
EBONE WP8

Design of a monitoring system and its cost-effectiveness

Report 8.1: Optimization of biodiversity monitoring through close collaboration of users and data providers

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1. Introduction

Biodiversity conservation is becoming an increasingly urgent matter in the face of accelerating degradation of natural ecosystems (e.g. Stem et al. 2005, Henry et al. 2008). Effective conservation however strongly depends on our ability to measure and monitor biodiversity change, and on the responses to biodiversity loss of a wide group of stakeholders and actors, including governments, local communities and the international community.

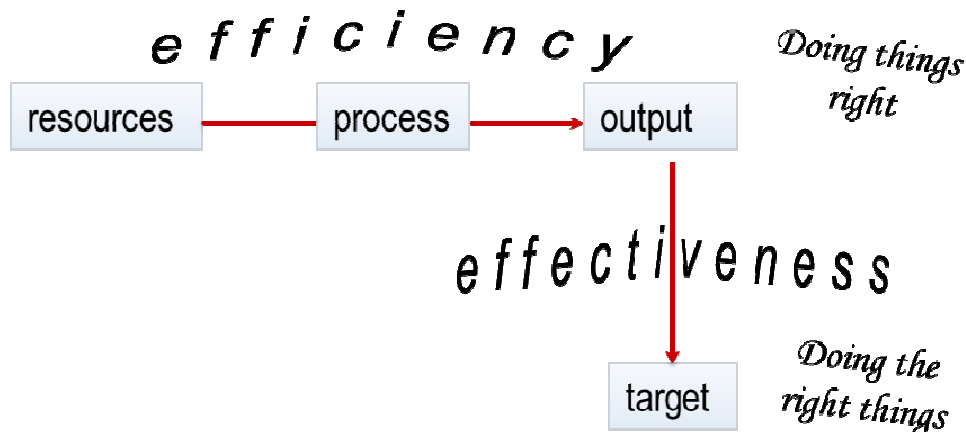
Biodiversity monitoring has taken on increased importance as the conservation community has faced a continuing struggle to demonstrate progress (or failure) made towards halting the loss of biodiversity (Stem et al. 2005) and countries are facing difficulties in meeting their reporting obligations under the Convention on Biological Diversity. At the European policy level there is a clear demand for an international framework for surveillance and monitoring of biodiversity (e.g. Bunce et al. 2008), indicating the growing need to quantify biodiversity composition and dynamics at large spatial and temporal scales (Henry et al. 2008). Such a framework is needed to bridge the gap between international commitments and national and local actions, and can help to ensure that biodiversity monitoring and responses to biodiversity change are incorporated effectively into policy and practice at international, national and local levels.

Over the years numerous efforts have been made to develop useful and practical monitoring and evaluation systems, often with mixed results. Existing monitoring schemes often differ in monitoring target and sampling design, cover only specific elements of biodiversity, have a restricted geographical coverage, and may have limitations regarding the potential for interpolation and generalisation (Schmeller 2008). Practical approaches to biodiversity monitoring vary widely among countries and the accumulating data are frequently not generalizable at the international scale (Teder et al. 2007). One problem seems to be that many new monitoring schemes are build up from scratch, overlooking lessons learned from the many existing monitoring approaches (Stem et al. 2005). Recent years have shown some cooperation and convergence of monitoring programs in Europe, but further efforts are required to work towards highly standardized, international monitoring networks. This includes integration and calibration of biodiversity information across existing schemes, which for the moment is still very poorly developed (Henry et al. 2008).

Clearly, further optimization of biodiversity monitoring is an important challenge as it forms the basis for improved decision making. To meet this challenge and in response to the widely recognised problem of limitation in the integration of existing monitoring systems, the EBONE project is aiming to develop a European-wide observation and monitoring framework that covers all aspects of biodiversity in one coherent system. In this report we describe the need for optimization of biodiversity monitoring, both from scientific and policy point of view, address the efforts that already have been made and set out the remaining issues that need improvement. One way forward is the optimization of biodiversity monitoring through close collaboration of users and data providers. In this respect optimization means that *the monitoring scheme fully meets the expectations and goals of the users in a scientifically and statistically sound and most cost-effective and durable way.*

Another key challenge that needs to be addressed is the cost-effectiveness of the monitoring system. Cost-effectiveness is a crucial consideration in the design of an efficient monitoring programme. A clear focus on crucial information needs in the conservation process is necessary and is generally believed to greatly increase conservation effectiveness (Nichols & Williams 2006).

Efficiency, the balance between the resources used and the output realized. Or, 'doing things right', 'the route to the goal'. **Effectiveness**, the capability of producing the desired results, the degree in which a target is achieved. Or, 'doing the right things', 'scoring a goal'. So, question is if the EBONE procedure is the most appropriate to yield data that produces the correct and relevant information, needed to answer the biodiversity policy and management questions. Is there alignment with the goals? In other words, *does the monitoring scheme fully meets the expectations and goals of the users in the most cost-efficient and durable way?*



In this report, we will analyze the conditions to meet efficiency and effectiveness. It is our conviction that this can only be achieved through close collaboration between the users and the developers of a monitoring scheme. Indeed, an optimal monitoring program starts from a precisely formulated information need. Only when this information desire is clarified, decisions can be made regarding

- 1) the proper selection of variables, the WHAT question;
- 2) the proper selection of methods for data collection, the HOW question;
- 3) the proper sample selection, the HOW MANY question;
- 4) the proper sampling strategy, the WHERE and WHEN question; and
- 5) the proper organization of the monitoring, the WHO question.

2. The demand for reliable, high-quality monitoring data

2.1. Current monitoring requirements of international biodiversity policy

A prerequisite for both EU and international biodiversity policy to be successful, is to have reliable measures on the status and trends in biodiversity. In this respect monitoring demands and obligations are mentioned in several policy documents at the **EUROPEAN AND PAN-EUROPEAN LEVEL**, as for example the Habitats Directive and several conventions and strategies dedicated to the former Countdown 2010 biodiversity target (http://www.countdown2010.net/archive/2010_target.html).

- Articles 11 and 17 of the **Habitats Directive** (92/43/EEC)

Article 11:

'Member States shall undertake surveillance of the conservation status of the natural habitats and species referred to in Article 2 with particular regard to priority habitat types and priority species.'

Article 17:

(1) *'Every six years from the date of expiry of the period laid down in Article 23, Member States shall draw up a report on the implementation of the measures taken under this Directive. This report shall include in particular information concerning the conservation measures referred to in Article 6(1) as well as the evaluation of the impact of those measures on the conservation status of the natural habitat types of Annex I and the species in Annex II. The report, in accordance with the format established by the committee, shall be forwarded to the Commission and made access to the public.'*

(2) *'The Commission shall prepare a composite report based on the reports referred to in paragraph 1. This report shall include an appropriate evaluation of the progress achieved and in particular, of the combination of Natura 2000 to the achievements of the objectives set out in Article 3 ...'*

- **Kyiv Resolution on Biodiversity** (2003)¹

Key target nr. 8:

'By 2008, a coherent European programme on biodiversity monitoring and reporting, facilitated by the European Biodiversity Monitoring and Indicator Framework, will be operational in the pan European region, in support of nature and biodiversity policies, including by 2006 an agreed core set of biodiversity indicators developed with the active participation of the relevant stakeholders.'

- **Message from Malahide** (2004); Halting the decline of biodiversity – Priority objectives and targets for 2010

¹ In 2002, the Parties to the Convention on Biological Diversity called for a significant reduction of the current rate of biodiversity loss by 2010. Europe has gone one step further: In 2003, 51 countries in the wider Europe adopted a target (the Kyiv Resolution on Biodiversity) to halt the loss of biodiversity by 2010. They aimed to achieve this through a set of policy actions identified in the European Biodiversity Strategy.

Objective 15:

'To implement an agreed set of biodiversity indicators to monitor and evaluate progress towards the 2010 targets, with the potential to communicate biodiversity problems effectively to the general public and to decision-makers and provoke appropriate policy responses.

...

15.2 Monitoring: use, and if necessary develop, monitoring frameworks (building on existing monitoring approaches and methods including those of civil society) in order to establish adequate harmonised data flows for the biodiversity headline and structural indicators to reveal and communicate key trends from 2006. ...'

Objective 16:

'To improve and apply the knowledge base for the conservation and sustainable use of biodiversity.

16.1 Status, trends and distribution of all habitats and species of Community Interest and of additional species of policy relevance known.

...

16.7 Common data standards and quality assurance procedures established and promoted to enable interoperability of key European and national biodiversity databases and inventories by 2008.'

- **EU Biodiversity Action Plan to 2010 and Beyond** (CEC 2006); the Action Plan specifies actions and targets for monitoring:

'C1.3 TARGET: Monitoring providing adequate data flow for implementation of indicator set, for reporting on favourable conservation status, and for broader assessment of effectiveness of this Action Plan by 2010.'

'C1.3.1 ACTION: Establish reference values for favourable conservation status for Habitats en Birds Directive habitats and species to achieve a consensus of definitions across Member States [2006/07]; monitor habitats and species in relation to these values [2007 onwards].'

'C1.3.2 ACTION: Use, and as necessary develop, monitoring tools, approaches and frameworks (building on those existing, including those of civil society) in order to establish and coordinate adequate harmonised data flows for the biodiversity indicators to reveal key trends [2007 onwards].'

'C1.3.3 ACTION: Develop shared information system for biodiversity monitoring and reporting in the EU, based on agreed biodiversity indicators, which makes data available to all interested users, streamlines reporting and supports policy evaluation and development at national, regional and global levels [2006 onwards].'

- The current **EU Biodiversity Strategy to 2020** (EC 2011a). Regarding the Habitats and Birds Directive, the Strategy proceeds with the former policy and hence requires appropriate assessment of conservation status (Target 1 of the Strategy). Target 2 however is new: *'By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15 % of degraded ecosystems'*. So, ideally a specific monitoring and reporting facility should be established in order to assess progress and the achievement of the set target. However, the action regarding monitoring and reporting the Strategy defines, only refers to target 1.:

Action 4 *Improve and streamline monitoring and reporting*

4a) *The Commission, together with Member States, will develop by 2012 a new EU bird reporting system, further develop the reporting system under Article 17 of the Habitats Directive and improve the flow, accessibility and relevance of Natura 2000 data.*

- The **Council Conclusions on the EU Biodiversity Strategy to 2020** (EC 2011b) however, adds a more general objective, stating that the Council
“AGREES that a coherent framework for monitoring, assessing and reporting on progress in implementing the Strategy is needed to link existing biodiversity data and knowledge systems with the Strategy and to streamline EU and global monitoring, reporting and review obligations under environmental and other relevant legislation as well as to avoid duplication and increase of reporting and administrative burden”

At the GLOBAL LEVEL the **Convention on Biological Diversity (CBD)** also implies a monitoring demand to support national reporting on the effectiveness of biodiversity measures taken:

CBD Article 26:

Each Contracting Party shall, at intervals to be determined by the Conference of the Parties, present to the Conference of the Parties, reports on measures which it has taken for the implementation of the provisions of this Convention and their effectiveness in meeting the objectives of this Convention.

Finally, also other policy domains require reliable data of biodiversity and environmental qualities in order to assess the impact of their policy on biodiversity and ecosystem functioning, and the achievement of agreed biodiversity related objectives. Important in this respect is the **Common Agricultural Policy after 2013** (http://ec.europa.eu/agriculture/cap-post-2013/index_en.htm). The European Commission proposes a **Regulation on the financing, management and monitoring of the common agricultural policy**, of which a specific article deals with monitoring:

Article 110: Monitoring and evaluation of common agricultural policy

Sub. 2: *The impact of the common agricultural policy measures referred to in paragraph 1 shall be measured in relation to the following objectives:*

- (a) *viable food production, with a focus on agricultural income, agricultural productivity and price stability;*
- (b) *sustainable management of natural resources and climate action, with a focus on greenhouse gas emissions, biodiversity, soil and water;*
- (c)

2.2. Some concerns on current availability, quality and use of monitoring schemes and monitoring data

Several recent and representative policy documents and processes indicate that there is a structural lack of reliable, harmonized monitoring data:

- Most recent, the **State and Outlook 2010** report on the European environment [SOER 2010] published by the European Environmental Agency (EEA 2010) reports that *“quantitative data on the status and trends of European biodiversity are sparse, both for conceptual and practical reasons. The spatial scale and level of detail at which ecosystems, habitats and plant*

communities are discerned is to a certain extent arbitrary. There are no harmonised European monitoring data for ecosystem and habitat quality, and the results of separate monitoring schemes are often difficult to combine because of low comparability.”

- The **Composite Report on the Conservation Status of Habitat Types and Species** (EC 2009) also reports that many Member States simply lacked comprehensive and reliable data for a large number of habitats and species reported in their territories, making conservation status assessments problematic. The monitoring and reporting obligations under Articles 11 & 17 of the Habitats Directive have recently improved the evidence base for conservation status assessments, but only for a number of habitats and species considered to be of Community interest. Still, overall some 13 % of regional habitat assessments and 27 % of regional species assessments were reported by Member States as ‘unknown’. The number of ‘unknown’ classifications was particularly high for species found in the countries of southern Europe, with Cyprus, Greece, Spain and Portugal all indicating ‘unknown’ for more than 50 % of the species reported in their territories. Missing information isn’t the only problem however. Even when information is available, problems often arise due to the different ways in which data is collected and presented. In addition, one of the main monitoring objectives is to be able to assess the effectivity of the Directive. Therefore monitoring data used to assess the conservation status of species or habitat, have to allow comparison between protected and unprotected areas, which in most cases is currently not possible (EEA 2010, EC 2009).
- Similar concerns were reported in the **Streamlining European Biodiversity Indicator (SEBI) process** that was established in 2005 to select and streamline a set of biodiversity indicators to monitor progress towards the 2010 target of halting biodiversity loss (Parr et al. 2010, EEA 2007, 2010). The lack of harmonized monitoring data that are comparable over space and time, is also probably the biggest constraint on the development and use of indicators for large-scale (European, global) biodiversity assessments. The SEBI process explored the availability of data in the indicator development process and the final choice of indicators was highly data constrained. But also many of the developed indicators need refinement that can only be achieved through harmonisation of monitoring schemes in order to improve comparability of data between countries and regions. Within the SEBI 2010 indicator set, the specific indicators on the components of biodiversity are most developed for species diversity and less developed for genetic and ecosystem diversity. The set is also strongest on compositional aspects (type, number, extent) and less well developed on structural and functional aspects (Groom *et al.*, 2005). Further research is needed on these relatively weaker aspects. However, whilst the 2010 targets provided an important mechanism by which to assess global biodiversity change, ensuring that appropriate monitoring systems are in place and translating monitoring results into effective conservation on the ground remain a major global challenge for a number of reasons, including financial and technical-capacity constraints, and policy and legal barriers.

The conclusions and comments (all from representative policy documents) that are mentioned above, are indicating that there is a clear demand for high quality, harmonized monitoring data. Better and more targeted information to underpin environmental analysis and assessment is needed. Having such data available would provide a long-term sustainable evidence base for biodiversity policy, allowing a more up-to-date reporting and finally enhancing effectiveness of environmental governance.

These observed problems are not new however and not restricted to biodiversity monitoring s.s.. E.g. a commission Working Document from 2002 on Rural Development Programming already signals the consequences of insufficient monitoring data: “*the quantity and quality of common monitoring data received for Year 2000, if repeated in future reporting exercises, would be unlikely to permit the*

Commission to effectively monitor the implementation of the RDPs or to provide reliable aggregation of monitoring data at Community level’.

The European Environmental Agency is aiming to strengthen existing environmental monitoring, data and information management. Improved collaboration and coordination between the vast array of actors and existing data and methodologies is one way forward (EEA 2007, 2010). Having a common approach towards habitats classification, for example, requires both a policy and scientific consensus across Europe. Many monitoring methodologies have been developed under national and EU research programmes but have still not reached their potential. Either they have not been applied fully due to shortcomings in data availability or there has been a lack of consensus on a particular method's application (EEA 2007). Anyway it is clear that international biodiversity urgently requires monitoring methods that can produce reliable data and indicators that are comparable over time and space.

Similar recommendations were obtained from a NGO consultation within the **European Habitats Forum** (EHF) (Walder et al. 2006): *“The EHF strongly advises Member States and the European Commission to develop common standards for monitoring, assessing and reporting. EHF recommends that databases must improve significantly before future reporting rounds and Member States must provide better data in comparable formats, using unified methodologies. Assessments should rely on sound scientific methods. The EHF also recommends that an integrated monitoring system is established to efficiently use biodiversity monitoring data in all areas of European policy, involving a broad spectrum of European civil society, such as scientists, NGOs and volunteers to assist with the implementation of monitoring obligations.”*

3. Efforts so far to improve international biodiversity monitoring

In response to the problems and issues mentioned in §2 there have been several efforts and initiatives on the (Pan-) European level to improve monitoring methods and activities. This includes initiatives that work towards the harmonization and inter-calibration of monitoring and reporting methodologies (§ 3.1), the development of common databases (§ 3.2), stimulation of international cooperation and partnerships in biodiversity and monitoring research (§ 3.3) and the development of common field protocols (§ 3.4)

3.1. Harmonization and intercalibration of methodologies

Important efforts have been done on the harmonization and inter-calibration of a number of monitoring methodologies existing in different European countries. Biodiversity monitoring can learn a lot from these on-going and completed initiatives. Some of them are part of a set policy implementation process, others were initiated by specific conservation NGO's or research projects.

Exemplary in this respect are the inter-calibration initiatives under the **Water Framework Directive** (WFD) (for more details, see also Box 1). The multitude of aquatic bioassessment methods used for European surface waters is perplexing. One is tempted to query if this methodological patchwork allows for comparable status classification across the continent. A Europe-wide monitoring of ecological status demands for harmonised assessment concepts and approaches (Birk et al. 2012). As a result a lot of effort is now being put in carrying out European benchmarking or intercalibration exercises to ensure that good ecological status represents the same level of ecological quality everywhere in Europe (e.g. Birk and Hering 2009, Borja et al. 2007). These inter-calibration exercises aim at consistency and comparability in the classification results of the monitoring systems operated by each Member State (EC, 2005). This means that the inter-calibration is not impacting on the monitoring systems themselves, nor on the biological methods, but focuses on the classification results and the commonly agreed standards and indicators.

Comparable harmonization efforts have been made within the development of a **Pan European Common Bird Monitoring Scheme** (PECBMS) under the coordination of the European Bird Census Council (EBCC) and BirdLife International (for more details, see Box 2). The PECBMS started in 2002 and aims to combine population trend data from annual national breeding bird surveys to produce policy-relevant indicators on the European level. More widely the project aims to improve the scientific standard of bird monitoring across Europe by fostering co-operation and the sharing of best practice and expertise (Klvanova et al. 2009, Voříšek et al. 2011).

Another good example showing European efforts on harmonization of monitoring methodologies can be found in the ongoing harmonization of **National Forest Inventories** in Europe. The main objectives are to improve and harmonise the existing national forest resource inventories in Europe and to support new inventories in such a way that inventories can meet national, European and global level requirements in supplying up-to-date, harmonised and transparent forest resource information for decision (policy) making, and to promote the use of scientifically sound and validated methods in forest inventory designs, data collection and data analysis. Several initiatives and action programmes were undertaken that focused mainly on harmonizing definitions and measuring practices as well as indicator and estimation procedures for assessing components of biodiversity (Tomppo et al. 2010, McRoberts et al. 2009, Winter et al. 2008).

Also in the context of European forest monitoring, the aim of the **BioSoil** project (2006 - 2009) was to demonstrate how a large-scale European study could provide harmonized soil and biodiversity data of forest plots and contribute to research and forest related policies. Already since the nineties a long transnational harmonization process for survey and sampling methods had been running and was consolidated in operational ICP-Forests submanuals. The soil submanual was the basis for the soil survey, unlike for the biodiversity assessment where a total new manual was developed but strongly based on the input of ICP-Forests experts. Moreover, most Member States regularly participated in interlaboratory soil ringtests and in intercalibration exercises as part of the ICP-Forests QA/QC programme. Essential preconditions and *very important lessons learned from the Biosoil project* are summarized in Box 3.

The same objectives are pursued in the framework of **monitoring initiatives and reporting obligations under the Habitats Directive**. Interpretation of habitat types (Annex I) and their conservation status differs between countries, indicating a need for an unequivocal description and definitions of the different Annex I habitat types and a common interpretation of favourable and unfavourable conservation status. The term 'favourable conservation status' is defined in the directive. However, each member state has to make its own interpretation which is expected to be based on scientific insights (Cantarello & Newton 2008, Mehtälä & Vuorisalo 2007). The terminology used in the directive is however often vague and susceptible for interpretation (Gaston et al. 2008). As a consequence, member states follow different trajectories towards its implementation (Bottin et al. 2005, LNV 2006), or use distinct definitions of habitats and evaluation methods (Cantarello & Newton 2008, Mehtälä & Vuorisalo 2007). This seriously hampers the process of streamlining results among member states and urges the need for calibration. Also it makes the implementation process less transparent and lowers the reliability of compiled data sets at the European level that should give an idea of the overall progress made so far (Louette et al. 2011). These problems encountered have been recognised by the European Commission and several efforts have been made to work towards similar and consistent interpretations between countries. Gaps in data have been identified and work is already underway to improve the reporting process and to improve data compatibility where this is needed (Sipkova et al. 2010) (See Box 4).

3.2. Common databases

The development of common databases aims to bring together biodiversity data in order to be able to make international biodiversity assessments and to collect the best available data on European biodiversity. One may distinguish between databases with the purpose to bring together and relate to each other metadata and a wide range of information on biodiversity from various sources, and transnational databases dedicated to specific taxa or components of biodiversity that thus provide standardized and reliable basic data for common research, assessment and reporting.

In the first category, the **Biodiversity Information System for Europe (BISE)** is the most important overarching database for data and information on biodiversity in the EU (<http://www.eea.europa.eu/themes/biodiversity/bise-2013-the-biodiversity-information>). Bringing together facts and figures on biodiversity and ecosystem services, it links to related policies, environmental data centers, assessments and research findings from various sources. It is being developed to strengthen the knowledge base and support decision-making on biodiversity. Priority objectives are the support of the implementation of the United Nations Convention on Biological Diversity (CBD) Strategic Plan 2010-2020, the support of the Integrated Platform for Biodiversity and Ecosystem Services (IpBes) and the support of the EU biodiversity Strategy 2011-2020. BISE is a partnership between the European Commission (DG Environment, Joint Research Centre and Eurostat) and the European Environment Agency (EEA). It incorporates the network of the European

Clearing House Mechanism within the context of the CBD. BISE showcases the best available information at European level, for example the findings of the EEA Biodiversity Baseline Report and the SEBI Indicator Assessments. Its content and services are being developed in collaboration with key users and information providers so that it meets the information needs of the new biodiversity vision and targets for the EU (<http://biodiversity.europa.eu/info>).

Within the framework of BISE, the **Biodiversity data centre (BDC)** is developed (<http://www.eea.europa.eu/themes/biodiversity/dc>). This database provides access to data and information on species, habitats and protected sites in Europe: species of the Nature Directives and Red List species, habitat types of the Nature Directives and the Natura2000 sites of the Nature Directives and nationally designated sites. Besides, related products for biodiversity indicators and assessment are available: quality-controlled spatial and tabular data, interactive maps statistics and indicators. For the European Environment Agency, the BDC is a core instrument to measure progress in implementing the EU 2020 Biodiversity Strategy. Mobilizing NGO data and citizen observations are essential in this respect (Rania Spyropoulou, EEA, 2011, Reporting – Data and indicators – Assessments (<http://glossary.en.eea.europa.eu/terminology/sitesearch?term=BISE+2011>)).

Also established within the BISE framework is the **Shared Environmental Information System (SEIS)** (<http://ec.europa.eu/environment/seis/>). This facility was developed because of the frustration that persists during the years. Indeed, despite the wealth of information that was collected, reporting systems remained fragmented, mainly because of shortcomings regarding timelines, availability, reliability and relevance of information, but also in relation to the ability to turn data into policy-relevant information. It was noticed that many initiatives and processes go in the right direction, but that coordination is still inadequate. The objectives are

- to improve the availability and quality of data and information needed to design and implement environment policy in the EU
- to reduce administrative burden on Member States and EU institutions and modernize reporting
- to foster the development of information services and applications that all stakeholders can use and profit from.

SEIS intends to be a truly integrated information system, sharing information across domains and scales and involving all components for effective knowledge building for reporting and policy implementation: Monitoring (= measure and record), Data (= integrate and manage), Information (= visualize and describe), Assessment (= analyze and understand), Reporting (= explain and communicate) (Franz Daffner, EEA, 2011, Organizing the reporting in a SEIS way. (<http://glossary.en.eea.europa.eu/terminology/sitesearch?term=BISE+2011>)).

Data in the **EUNIS database** (<http://eunis.eea.europa.eu/>) are collected and maintained by the European Topic Centre on Biological Diversity for the European Environment Agency (EEA) and the European Environmental Information Observation Network (Eionet; see also § 3.3) to be used for environmental reporting and for assistance to the NATURA2000 process and coordinated to the related EMERALD Network of the Bern Convention. EUNIS is part of the Biodiversity data centre. EUNIS consists of information on Species, Habitat types and Sites compiled in the framework of NATURA2000, but also data collected from literature, international conventions and EEA reporting activities, national Red Books, etc.. The EUNIS Habitat types classification is a comprehensive pan-European system to facilitate the harmonized description and collection of data across Europe through the use of criteria for habitat identification; it covers all types of habitats from natural to artificial, from terrestrial to freshwater and marine. The database includes EUNIS Habitats and Annex I Habitats of the EU Habitats Directive.

The **Eumon database** ('DaEuMon') developed within the EuMon project (EU-wide monitoring methods and systems of surveillance for species and habitats of Community interest – European Union's Framework VII program, <http://eumon.ckff.si/>) provides an overview of monitoring approaches and monitoring organizations in Europe. The EuMon consortium developed a comprehensive database (DaEuMon) on European biodiversity monitoring schemes. The database contains metadata that characterize monitoring schemes. DaEuMon will be maintained and expanded continuously and can be considered as a useful reference to monitoring schemes, bringing together information of more than 600 monitoring schemes across Europe (Framstad et al. 2008). EuMon evaluated existing monitoring methods for the design of monitoring schemes, for the analysis of monitoring data, and for the integration of collected information across schemes. Based on these evaluations, EuMon developed a primer for biodiversity monitoring that summarizes the most important recommendations and also compiled extended guidelines. Based on the central DaEuMon database EuMon continuously tries to contribute to common and harmonized monitoring protocols, which will provide more comparable data for agreed biodiversity indicators, thus allowing standardized reporting over time and across Europe.

SynBioSys Europe is an example of a pan-European thematic database (<http://www.synbiosys.alterra.nl/synbiosyseu/>). It is the initiative of the European Vegetation Survey to define and set up common standards and to establish a web-based facility to store information on vegetation and plant species of Europe that can be used for the evaluation and management of biodiversity among plant species, vegetation types and landscapes. It functions as a network of distributed national and regional databases related through a web-server. At the species level, more than 300,000 species and subspecies names from national species lists are imported into a common checklist in which the taxa are synonymized. This checklist allows the integration of species and vegetation databases from various countries (Schaminée et al, 2009). With the cross-reference to the EUNIS classification (Rodwell et al., 2002) and the European classification on climate, soils and anthropogenic pressures (Mücher et al, 2009), this database and web-facility bears great potentials for future pan-European biodiversity monitoring and reporting.

3.3. Collaboration & partnerships

Long before the end of the agreed time period for the set objectives of the European Biodiversity Strategy was reached, it was clear that achieving the 2010 Biodiversity target was seriously hampered by a lack of effective science to measure progress and to assess biodiversity status and change. In response to this problem, the European Commission established several initiatives to enhance collaboration amongst the (European) biodiversity research community.

ALTER-Net, Europe's biodiversity network, (<http://www.alter-net.info/>) for example is a Long-Term Biodiversity, Ecosystem and Awareness Research Network that constitutes of 26 partner institutes from 18 European countries and aims at integrating European biodiversity research in terrestrial and freshwater ecosystems: assessing changes in biodiversity, analysing the effect of those changes on ecosystem services and informing the public and policy makers about this at a European scale. As ALTER-Net stands for a better integrated and stronger European biodiversity research capacity, the essence of its Common Research Strategy is interdisciplinary and integrated ecosystem research combining ecological and socio-economic approaches. To ensure the relevance of its research activities for Europe's biodiversity policy, ALTER-Net equally wants to enhance and improve an effective and regular communication with various stakeholders and policy makers. Therefore, it wants to facilitate a two-way interaction between researchers and policymakers, focusing particularly on the European Commission and the International Panel for Biodiversity and Ecosystem Services (IPBES).

Originally funded by the European Union's Framework VI program to stimulate a collaborative approach, ALTER-Net is now operating independently. ALTER-Net is one of the *Networks of Excellence* established by the European Commission to achieve lasting integration of research capacity.

ALTER-Net is also a founder organisation of **LifeWatch** (<http://www.lifewatch.eu/>), (<http://portal.lifewatch.eu/>), a new and ambitious European project to create a state-of-the-art facility supporting biodiversity research. LifeWatch, currently in its building phase, will provide access to a wide range of biodiversity datasets, modelling and analysis tools. Its main objective is to link ecological monitoring data collected from marine and terrestrial environments with the vast amount of data in physical collections. LifeWatch is an integrated approach for developing an advanced infrastructure for biodiversity research using a wide range of techniques. It will be significant in tackling one of the major challenges facing modern society, bringing together many disciplines and investigative techniques. The new infrastructure will open up new areas of research and new services by providing access to the large data sets from different levels of biodiversity (genetic, population, species and ecosystem) together with a network of observatories, facilities for data integration and interoperability, virtual laboratories offering a range of analytical and modelling tools and a Service Centre providing special services for scientific and policy users, including training and research opportunities. It also intends to give access to scientists and policy makers comparing and supplementing these data with even more data obtained from weather stations, satellites, biological collections from all over Europe. The LifeWatch infrastructure is supported by all major European biodiversity research networks.

Initiated by the European Environment Agency (EEA) and its member and cooperating countries, **Eionet**, the European Environment Information and Observation Network, is the core partnership regarding environment information sharing (<http://www.eionet.europa.eu/about>) (see also EEA, 2012). It consists of the EEA itself, five European Topic Centres (ETCs) and a network of around 1000 experts from 39 countries in over 350 national environment agencies and other bodies dealing with environmental information. These are the National Focal Points (NFPs) and the National Reference Centres (NRCs). The mission of Eionet is "to provide timely and quality-assured data, information and expertise for assessing the state of the environment in Europe and the pressures acting upon it. This enables policy-makers to decide on appropriate measures for protecting the environment at national and European level and to monitor the effectiveness of policies and measures implemented".

The Eionet partnership is crucial to the EEA in supporting the collection and organisation of data and the development and dissemination of information. Through Eionet, the EEA coordinates the delivery of timely, nationally validated, high-quality environmental data from individual countries. This forms the basis of integrated environmental assessments and knowledge that is disseminated and made accessible through the EEA website. This information serves to support environmental management processes, environmental policy making and assessment, and public participation at national, European and global levels.

As Eionet's main objective is to optimize and organize the exchange and sharing of information, Reportnet is an important infrastructure for supporting and improving data and information flows (<http://www.eionet.europa.eu/reportnet>). The system integrates different web services and allows for distributed responsibilities.

3.4. Development of common field protocols

The potential to combine biodiversity monitoring data from different sources and localities in a common analysis and reporting, not only depends on agreed nomenclature and typologies, complete metadata and shared database, but also, and not the least, on comparable and reliable protocols for

the initial data collection. This concerns the sampling design (statistics!), the selection of the biodiversity variables, as well as the sampling technique (the precision and completeness of the collected data). In this respect, there have been several initiatives on the (pan-) European level to develop common field protocols for particular biodiversity components as the basis for biodiversity monitoring.

A well-known example on the species level, is that of **butterfly monitoring** in Europe. Since the first Butterfly Monitoring Scheme in the UK (<http://www.ukbms.org/>) started in the mid-1970s, butterfly monitoring in Europe has developed in more than ten European countries. These schemes are aimed to assess regional and national trends in butterfly abundance per species. A new development is to establish supra-national trends per species and multispecies indicators. Combining data from national recording sessions works very well, as all schemes apply the same method that was developed for the British Butterfly Monitoring Scheme (Pollard and Yates 1993, Van Swaay et al. 2002, Van Swaay et al. 2008).

More recently, also for habitat monitoring common field protocols were developed. Successful in this respect was the European Union's Framework V programme **BioHab**. In this project a habitat monitoring system has been developed that enables consistent recording and monitoring of habitats across Europe, and beyond (see for instance Jobse, 2009). The core of this habitat monitoring methodology is the system of *General Habitat Categories* (GHC's) derived from 16 easily identifiable plant life-forms and 18 non-life forms. Recording is based on strict rules and unequivocal definitions of terms and keys. To obtain total information on a habitat and its characteristics and qualities, indispensable to allow precise assessment and monitoring of the state of a habitat, the recording of the GHC's is completed with the description of qualifiers regarding for instance, environmental and management features. This GHC methodology provides an easily repeatable system for use in the field that can be cross-related to other habitat classification schemes such as Habitat Directive Annex I and EUNIS (Bunce et al. 2008). The GHC's may provide the lowest common denominator linking to other sources of data required for assessing biodiversity e.g. phytosociology, birds and butterflies. They may also be more easily discriminated from the air or space using remote sensing methods because the system is based on habitat structure. The approach provides an extremely powerful assessment tool for biodiversity, providing a missing link between detailed site-based species, population and community level measures and extensive assessments of habitats from remote sensing (Bunce et al. 2008, Parr et al. 2010). The approach that EBONE is using, is based on the methodology developed in BioHab.

Box 1: Intercalibration efforts under the Water Framework Directive

Over the last decade biomonitoring of European aquatic ecosystems has changed substantially (Birk et al. 2012). This development was driven by the EU Water Framework Directive 2000/60/EC (WFD), which required assessment methods for different aquatic ecosystem types (rivers, lakes, transitional waters, coastal waters) and different aquatic organism groups (phytoplankton, aquatic flora, benthic invertebrates, fish). The WFD has changed management objectives from merely pollution control to ensuring ecosystem integrity (Borja et al. 2008). Deterioration and improvement of “ecological status” is defined by the response of the biota, rather than by changes in environmental parameters. This response must be investigated at the level of the “water body” (e.g. a river stretch, a lake or a part of a coastal water), which represents the classification and management unit of the WFD. Water bodies of the same category are grouped into “water body types”, for each of which undisturbed reference states are defined. In biological assessment, the observed condition is compared against the reference status with the result given in five classes: high status (no differences to reference conditions), good status (slight differences), moderate status (moderate differences), poor and bad status (major differences). Good ecological status represents the target value that all surface water bodies have to achieve in the near future.

While the WFD indicates what characteristics of the aquatic organism groups should be assessed (e.g. “abundance”, “community composition”) it does not specify, however, which indices or metrics of these various elements should be used (Hering et al. 2010). This decision was left to the EU Member States. Just developing methods for the different combinations of organism groups and water categories, would have resulted in about 20 methods. However, many countries preferred developing country-specific methods, either to continue using existing time series by adapting their national methods to the WFD, or to regard for the specific ecoregional and biogeographic situation; therefore, a multitude of methods resulted (almost 300 according to Birk et al 2012) instead of a handful of methods applicable Europe-wide (e.g. Birk and Schmedtje 2005; Borja et al. 2009).

Differences among methods include e.g. metric selection, sampling strategy, taxonomic resolution (species or phylum level) and quality class boundary setting (Birk et al. 2012).

An important factor in the use of metrics seems to be monitoring tradition. Metric selections not only differs between countries, metric types used also differ considerably between different organism groups and different water categories. For example, in lakes, the metric chlorophyll-a, a measure of algae abundance, is present in all assessment methods. For rivers, there is a long tradition in using benthic invertebrates, for assessing the effects of organic matter. Further method development has built to a large extent on these traditions. For fish, assessment methods were developed more recently in the framework of scientific project specifically targeting the WFD (Schmuts et al. 2007), resulting in a more balanced, multimetric approach. In addition it is also remarkable that almost all methods are developed by countries of central and Western Europe, also reflecting the different monitoring traditions.

For effective evaluation representative sampling is fundamental. As sampling strongly determines the results of status classifications its strategy shall ideally be aligned to the individual biological indicator used in monitoring (de Jonge et al. 2006). But in practice many assessment methods were built on the basis of already existing data and practices (Beliaeff and Pelletier 2011). Sampling strategies range from small-scale sampling of the taxonomically diverse groups of benthic invertebrates and phytobenthos that demand elaborate processing, to very large-scale sampling of plants or the mobile fish fauna.

Boundary setting is also an important step in the design of assessment methods as it defines the target values for environmental management. Ecological thresholds, i.e. small changes in an environmental driver that trigger major changes in the ecosystem, should play a key role in boundary setting (Lyche Solheim et al. 2008). In practice however boundary setting often follows non-ecological principles. Statistical approaches, for instance, in which the gradient of biological condition is divided into equidistant classes allow for convenient mapping of the ecosystem status but lack biological significance. As a consequence there is no guarantee that ecological class boundaries correspond to meaningful changes in ecosystem functioning and biological communities (Birk et al. 2012).

At first glance the multitude of aquatic bioassessment methods used for European surface waters is perplexing. One is tempted to query if this methodological patchwork allows for comparable status classification across the continent. A Europe-wide monitoring of ecological status demands for harmonised assessment concepts and approaches (Birk et al. 2012). As a result a lot of effort is now being put in carrying out European benchmarking or intercalibration exercises to ensure that good ecological status represents the same level of ecological quality everywhere in Europe (e.g. Birk and Hering 2009, Borja et al. 2007). These inter-calibration exercises aim at consistency and comparability in the classification results of the monitoring systems operated by each Member State (EC, 2005). This means that the inter-calibration is not impacting on the monitoring systems themselves, nor on the biological methods, but focuses on the classification results. The inter-calibration exercises aim to ensure that the values assigned by each Member State to the good ecological class boundaries are consistent with the Directive’s generic description of these boundaries and comparable to the boundaries proposed by other Member States. This means that these boundaries need to correspond to comparable levels of ecosystem alteration in all Member States’ assessment methods for biological quality (EC, 2005). It is warned however that the process will only work if common boundary values

similar assessment methods or where the results for different assessment methods are normalised using appropriate transformation factors (EC 2005, van den Bund 2009, Poikane 2009, Carletti & Heiskanen 2009).

Next to comparability, geographical variation is also taken into account in the intercalibration process under the WFD. Expert groups have been established for lakes, rivers and coastal/ transitional waters, subdivided into 14 Geographical Inter-calibration Groups of MSs that share the same water body types in different sub-regions or ecoregions (van den Bund 2009, Poikane 2009, Carletti & Heiskanen 2009).

Box 2: Development of a Pan-European Common Bird Monitoring Scheme

Objectives, organisation and participation

The Pan-European Common Bird Monitoring Scheme (PECBMS) is a joint initiative led by the European Bird Census Council (EBBC) and BirdLife International. Other main partners include the RSPB and Statistics Netherlands. The PECBMS started in 2002 and aims to combine population trend data from annual national breeding bird surveys to produce policy-relevant indicators on the European level (Klvanova et al. 2009).

The PECBMS also aims to increase both the number of countries collecting and submitting data on trends, and the number of species covered, to help develop and promote biodiversity indicators in Europe. More widely the project aims to improve the scientific standard of bird monitoring across Europe by fostering co-operation and the sharing of best practice and expertise (Voříšek et al. 2008, Voříšek et al. 2011).

The project has established a large European network of collaborators, including coordinators of national or regional monitoring schemes, EBBC delegates and BirdLife partners. In countries where volunteers and their organizations are short of means, BirdLife International assists by establishing a regional coordination secretariat that, besides organizing the bird census, also engages in the conservation of important bird areas. The project is coordinated by a central coordination unit based at the Czech Society for Ornithology in Prague, supported by means provided by BirdLife International and a European grant. A Steering Group and Technical Advisory group oversee the work (Voříšek et al. 2008, Voříšek et al. 2011, Klvanova et al. 2009).

National monitoring schemes are organized mostly by NGOs with various involvements of other institutions (governmental agencies, universities, research institutes, etc). The majority of field work (i.e. bird counts) is done by volunteers and managed by coordinators. The national coordinators provide PECBMS with the results of their schemes according to agreed standards and formats. Because of lack of tradition, there is a need for continuous support and activation of the volunteer networks in the Eastern European countries. However, it can be expected that national authorities will increase their support as the Wild Birds Indicator is part of the SEBI 2010 indicators and hence oblige the member states to provide accurate data on a regular basis.

Harmonization of national methodologies

As bird monitoring in Europe is mainly organized on a national level; most countries have developed their monitoring schemes rather independently. As a result national bird monitoring schemes employ an array of different methods (Voříšek et al. 2008, Voříšek et al. 2011)

Birds are counted using standardised field methods. There is however no uniformly best field method to count birds. Three main standard types of methods are used by national schemes and are accepted within the PECBMS: territory mapping, line transect and point counts. As the PECBMS works with national indices that shed light on the degree of change in successive years, rather than with raw data that produce absolute values (for instance bird densities) (see further), the field method used to produce the national indices is of minor concern as long as this method is standardized through years and provides a reliable, representative picture of a species' national trend.

Representativity however is mainly determined by the selection of sampling plots (sites) across a territory or country. To improve representativity of national schemes and because of differences between them, harmonization efforts are mainly focusing on the sampling designs of national schemes. The most common methods to select sample plots in generic breeding bird monitoring schemes are *free choice*, *systematic selection*, *random selection* and *stratified random selection*. Free choice was the common method in older schemes, but is of course prone to bias and therefore not ideal in the context of PECBMS. Nowadays most of these schemes have been replaced with schemes with some element of randomization (stratified random or systematic choice of sampling sites). In the Netherlands post-stratification and weighting has been used as the method to reduce potential bias (Van Turnhout et al. 2008). All in all, thanks to the improvements in plot selection and increased rigour that have been applied, it is now believed that bias which could affect results at the European level is unlikely. Nonetheless, improvements in scheme design are ongoing in other countries. Stratified random selection is the prevalent method of newly established monitoring schemes in Europe (Voříšek et al. 2011).

Combining national data into supra national indices

Although field methods, selection of sample plots, and also number of years covered differ among European countries, statistical methods can handle this. These differences do not influence the supranational results, because the indices are standardised before being combined. National species indices are calculated using a statistical software package that was specifically designed for the project (Pannekoek and van Strien 2001), which allows for missing counts and produces unbiased yearly indices and standard errors using Poisson regression. The national indices are then combined by the central coordination unit and by Statistics Netherlands into supranational indices for species, weighted by estimates of national population sizes. Weighting allows for the fact that different countries hold different proportions of each species' European population, to make sure that a change in a larger national population has an accordingly greater impact on the overall trend than a change in a smaller population (Van Strien et al 2001). After these weightings and imputation steps, the national totals are summed up to European totals. Supranational indices are then combined into multispecies indices, the European "indicators".

Quality control

The overall aim of PECBMS is to use national data and to produce supranational indices and trends and multispecies indicators as precise and accurate as possible. Therefore, quality control is an extremely important part of the data flow process. Data quality control has been implemented at each level of data collation and analysis. Data quality of national schemes as well as PECBMS supranational outputs has been independently checked and approved by publications in peer-reviewed scientific journals (Voříšek et al 2011).

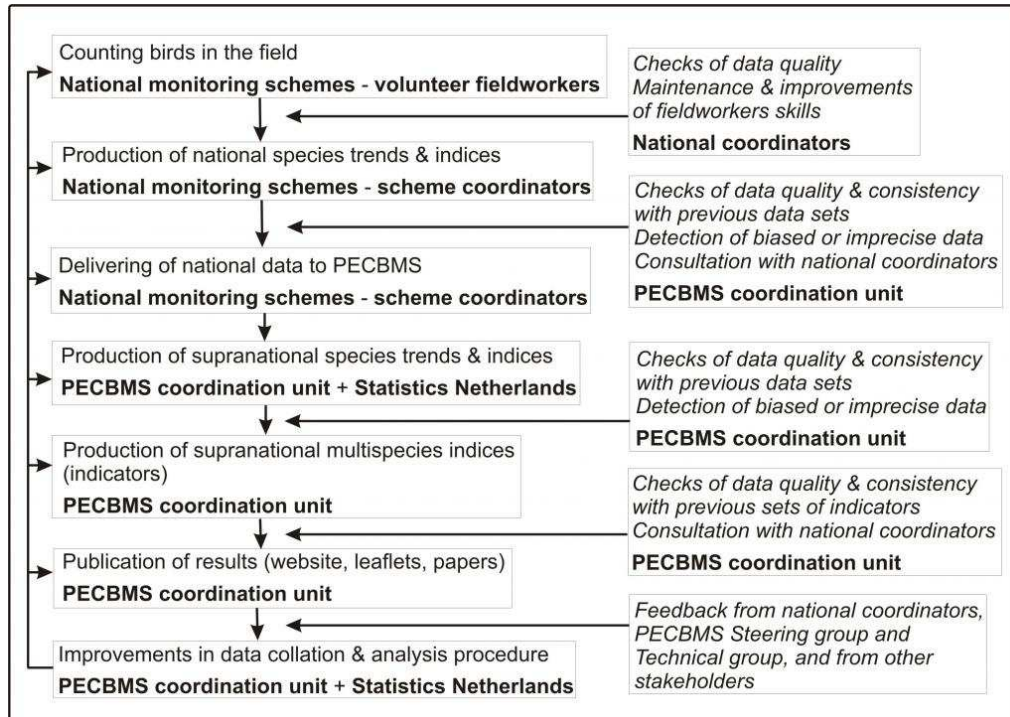


Figure 3.1. Organisation, coordination and quality control steps in the Pan-European Common Bird Monitoring Scheme (from Voříšek et al. 2011)

Involvement of volunteer fieldworkers

Extensive networks of professional and amateur naturalists and ornithologists are central to the PECBMS. Since bird watching is a widespread activity across Europe and elsewhere, it is often no problem to recruit a high number of volunteers for bird surveys, and this is relatively easy in comparison to other taxa. Just as professionals, volunteers must be able to identify the birds in the field properly, record field data accurately and in proper format and deliver them timely to the coordinators. Since the volunteers do their work for free, one might fear that their work suffers from this, but such fear is unnecessary. Coordinators use a wide array of methods to check the skills of the volunteers and to guarantee a high standard of the data delivered (Voříšek et al. 2011). A frequent feedback through reports with the results of the monitoring and a regular updating of the methods used, the evolutions and arrangements in other regions, etc., appear to be important to maintain the enthusiasm of the volunteers. The same counts for the common training sessions where new and experienced volunteers and professionals learn from each other and respectful exchange and pass experiences.

One possible problem connected to working with volunteers is the selection of sampling plots. Volunteers might prefer to count in areas that are rich in birds rather than to be directed to plots which have been selected randomly. To solve this problem several national monitoring schemes select sample plots in a stratified random manner. Another problem is that volunteer fieldworkers can leave a scheme at any moment, causing a turnover in the sites counted and missing values. This occurs in any long-term monitoring scheme and statistical techniques and software are widely available and used to solve this problem (Voříšek et al 2011). While the potential risks linked to the involvement of volunteer fieldworkers have been solved, several advantages remain: the running costs of a scheme are relatively low and large-scale schemes are feasible (Greenwood 2007).

Box 3: Lessons learned from the EU BioSoil demonstration project

Author: Bruno De Vos

BioSoil in a nutshell

The aim of the BioSoil project (2006 - 2009) was to demonstrate how a large-scale European study could provide harmonized soil and biodiversity data of forest plots and contribute to research and forest related policies. BioSoil was set up in response to the Forest Focus Regulation (EC) No. 2152/2003 of the Directorate-General for the Environment (call 2005 – 2006). Twenty-three member states all active within the UN/ECE ICP-Forests network participated in BioSoil and submitted their survey data to a central database. This resulted in one of the largest European environmental datasets potentially linking abiotic (soil) and biotic (forest biodiversity) components together and a valuable data source for years to come.

The spatial coverage of BioSoil within Europe was high: in addition to non-MS as Norway and Switzerland, no data were collected by Luxemburg and The Netherlands, and by the regions Wallonia (in Belgium) and the Azores (Portugal). This way, about 82 % of the total European forest was effectively surveyed. The temporal coverage of the study was relatively short: the survey was conducted within 3 years, but is embedded in a forest monitoring process of over 25 years by ICP-Forests. The taxonomic coverage is restricted to forests, roughly representing one third of the European land area occupied by terrestrial ecosystems.

The surveys were conducted on predominantly existing ICP-Forests plots: the soil component on 4928 systematic 16 x 16 km level I plots and on 127 intensive level II plots whereas the biodiversity component on 3379 level I plots (Hiederer and Durrant 2010). Hence, common data is available on max. 69 % of the level I plots. By its systematic plot design, the plots are quite evenly represented in all parts of Europe, covering representatively all kinds of forest and soil types.

Essential preconditions

BioSoil was generally evaluated as a successful project, mainly due to some preconditions. Without these BioSoil would probably not have been proposed or realized at all.

Firstly, fertile ground for BioSoil was the ongoing UN/ECE ICP Forests programme, financially supported by EC and individual Member States, and evidenced by a strong network of forest institutes and their experts across Europe all strongly believing in a collaborative approach of forest related research.

This network holds both excellent expertise and long term experience, especially in field assessments, and an operational research infrastructure (documented Level I and II plots, sampling equipment, trained laboratories).

Secondly, since the nineties a long transnational harmonization process for survey and sampling methods had been running and was consolidated in operational ICP-Forests sub-manuals. The soil submanual was the basis for the soil survey, unlike for the biodiversity assessment where a total new manual was developed but strongly based on the input of ICP-Forests experts. Moreover, most Member States regularly participated in interlaboratory soil ringtests and in intercalibration exercises as part of the ICP-Forests QA/QC programme.

The third precondition was that the time was right for a new forest (soil) survey. In most countries the previous forest soil survey ran 10-15 years before and some new national surveys were already planned. Since many EU plots coincide with the national plot network (interaction EU level – national/regional level) it seemed beneficial to survey both for European and national/regional interests.

The fourth precondition was that financial support from Europe was envisaged, up to 75% of the total eligible costs, which was stimulating both for large and small countries to get on board of this European research initiative.

Lessons learned from the BioSoil experience

More than two years after closing the BioSoil project, we dare to formulate following general evaluations and recommendations:

1. BioSoil demonstrated that a harmonized, large-scale soil and biodiversity survey is feasible across Europe, even within a relatively short period of 3 years, but only under the four above mentioned preconditions;
2. Within this period however, time was definitely too short for proper data validation, robust data storage and even preliminary data elaboration and evaluation. The latter processes need at least another 3 years by experienced staff.
3. It is of utmost importance to maximally use the experiences from the past (in this case: the first European Soil Survey; Vanmechelen et al. 1997) to avoid or circumvent similar monitoring problems encountered in the past. Building on experienced people (expert groups) and/or running a substantial pre-study based on previous endeavors is a prerequisite.

4. Clear instructions are essential, especially for transnational surveys. Even with perfectly clear manuals, survey and analysis methods applied within each country will always slightly differ due to other environmental and climatic conditions, variable equipment, background and experience of the field staff, etc. If starting manuals are less clear from the beginning, transnational harmonization is highly tentative. For instance: in BioSoil the documents with file format specifications were not always in line with the BioSoil manual, which caused already substantial confusion, which became apparent in the resulting database.
5. The quality of manuals may always be improved, and updates should be released also after the end of the survey or monitoring projects, so that future surveys may benefit from them.
6. Before and during the survey, intercalibration exercises (field training) and parallel analytical ringtests are essential to control and quantify all sources of variation. These data should be readily available to all partners, and are especially useful when the data is validated and evaluated.
7. Critically considered, the BioSoil project was probably too ambitious (too much layer depths; too much variables; highly intensive field work (profile description); different sets of soil variables for chemical analysis of layers and horizons within the same soil profile). Although all 23 countries submitted soil data, some of them only reported a subset of the required data causing imbalances in data availability of the resulting database. It seems better to restrict the requested variables but make them absolutely mandatory to report.
8. During the BioSoil survey, international reference methods were prone to change (FAO guidelines for soil description (2006), update 2007 of World Reference Base classification) which caused some bias in the reported soil classifications. It is recommended to determine which version to use and stick to that version during the survey, even when new updates are released.
9. A wealth of data was generated within BioSoil, which beautifully filled the existing knowledge and data gaps, but no comprehensive plans (nor money) were foreseen for thorough data evaluation and publication. It might be argued that this is mainly the task of the scientific community, but even in that case one should expect plans for data(base) access or dissemination, which were not developed either. This should be clear from the beginning.
10. Many countries encountered problems by using the BioSoil online data submission module. Difficulties when entering data and/or extensive 'error reports' are not particularly stimulating for Member States when entering large datasets. Submission modules should be robust and user friendly and thoroughly tested and instructed prior to initialization. It appeared that large and valuable data were missing in the final database because a country was unable to finish the whole input procedure. This might lead to frustration and should be avoided.
11. Due to unclear ownership and distribution rights of BioSoil data, the gathered data ended up in various databases: the soil data is (still) separated from the biodiversity data in DG JRC, where the soil data is probably integrated within the Soil Geographical Database of Europe and/or SPADE databases. Since the original BioSoil soil database was unavailable during the EU Life+ FutMon project, all soil data were reassembled in the BioSoil⁺ database, now used for scientific exploration and linking with the Forest Soil survey data (till 1997) and all other ICP Forests datasets. Prior to conducting a next EU wide soil survey, a transparent and binding agreement dealing with data ownership and database availability should be established between all actors. It is crucial that the scientific community is granted full access to the validated data after a predefined period.
12. Despite minor problems that occurred during the end phase of the BioSoil demonstration project, this project showed that transnational collaboration of hundreds of people across Europe could lead to a substantial soil database. Information has already been derived and published for several Forest and Soil condition reports (De Vos & Cools 2011) and the report on the State of European Forests 2011, among others. Part of the data is currently implemented for calibration and validation of various models and will definitely lead to numerous scientific publications.

Box 4: Harmonization of reporting under the Habitats Directive

An Interpretation Manual describes the habitats considered under the Habitats Directive but the descriptions given are mostly very short, only available in English and often have to cover a wide range of variation (Evans 2010). The terminology used is therefore often susceptible for interpretation (Gaston et al. 2008) and as a consequence Member States use different definitions of habitats and evaluation methods (Cantarello & Newton 2008, Mehtälä & Vuorisalo 2007). This seriously hampers the process of streamlining results among Member States and urges the need for calibration. Also it makes the implementation process less transparent and lowers the reliability of compiled data sets at the European level that should give an idea of the overall progress made so far (Louette et al. 2011).

Interpretation of habitat types

Comparison of published interpretations by Member States show that there can be marked differences in interpretation of habitats. For instance interpretation of several grassland habitats clearly differs between France, Germany and the UK (e.g. “6510 Lowland hay meadow”, “6230 Species-rich *Nardus* grasslands”) (Evans 2010). Sometimes there are even differences between regional administrations in the same country, especially where nature conservation is a regional responsibility as in many federal countries. For example in Belgium, woodland with *Quercus* but without *Fagus sylvatica* can be considered to be “9120 Atlantic acidophilous beech forest with *Ilex* and *Taxus* in the shrub layer” in Flanders, if the soil and herb layer correspond to this habitat, whereas in Wallonia the presence of *F. sylvatica* is considered essential, regardless the soil characteristics. Many habitats also have names which combine a physical description, sometimes with a geographical element, with a taxon (e.g. “6410 *Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (*Molinion caerulea*)” and it appears that some countries give more importance to one or other component of the name (Evans 2010). Also some member states have identified plant communities of nature conservation interest (‘regional important habitats’) and proposed them as the nearest Annex I habitat type even though they do not fully fit the definition (Evans 2006).

Several efforts have been made to work towards similar interpretations between countries. The principal forum for discussing varying interpretations has been the series of biogeographical seminars organised by the European Commission with support from the European Topic Center on Biodiversity (ETC/BD) (Evans 2006, 2010). The focus of these meetings is on the sufficiency of the Natura2000 network to ensure the long-term survival of the habitats and species of annexes I and II. However, this often involves discussion of how to interpret the habitats. The Boreal region has probably made the largest effort to have consistent interpretations with regular meetings of experts, often organised by the Baltic Environmental Forum with funding from the European Commission to discuss the implementation of the Habitats Directive (Evans 2010).

National assessments of conservation status

The EU Habitats Directive requires all member states to report every 6 years on the implementation of the Directive (Article 17). The national reports for 2001-2006 included assessments of the conservation status of each annex I habitat present in the country following an agreed methodology. This was the first time that the conservation status of habitats has been assessed across such a large area encompassing so many countries (Sipkova et al. 2010). Earlier evaluations of similar areas in the USA and Australia (Nicholson *et al.* 2009) only cover single countries. Standardized assessments of conservation status require the use of well-defined methods and comparable concepts in all Member States. The definition of ‘Favourable Conservation Status’ in Article 1 forms the legal background for assessment matrices (one for each species and habitat) that have been developed as part of the agreed reporting format (EC 2005).

Although guidelines and a standard method were provided, it is clear that this was not sufficient to ensure that the data reported in national assessments was compatible across the European Union (Sipkova et al. 2010). Assessment of the actual conservation status remains a challenging exercise that often leads to different approaches across Member States (Opdam et al. 2009). There are differences between countries, both in the precision of data reported (e.g. area of habitat), the criteria used for assessments and the use of expert judgment. For example almost all forest types in Denmark have been reported as ‘favourable’ whereas very few other countries in the Atlantic and Continental regions have assessed more than a few of their forests as favourable. It appears this is due to the criteria used to assess ‘structure and function’ for forests in Denmark being different from those used elsewhere. For some Mediterranean countries such as Italy and Greece, data was only available from protected sites for many habitats and this may be partly responsible for the high proportion of assessments as ‘favourable’ in these countries (Sipkova et al. 2010, Evans 2010).

The problems encountered, both in sourcing the required data and in the method used for assessments, have been recognized by the European Commission and a working group on reporting has been established which, among other topics, will provide improved guidelines for future reports, to facilitate aggregation and comparisons between Member States and biogeographical regions. Gaps in data have been identified and work is underway to improve the reporting process and to improve data compatibility where this is clearly needed (Sipkova et al. 2010). Standardization is particularly needed for the definition of range of habitats, the reporting of ‘structure & function’, including the treatment of typical species and the format for spatial data (Sipkova et al. 2010).

4. Remaining issues of improvement as dealt with in EBONE

4.1. Developing a monitoring programme: collaboration in an interactive process

Although the different initiatives mentioned above have considerably improved the conditions to develop and implement a pan-European biodiversity monitoring programme, a lot of issues remain to be optimized. To discuss them and assess the way EBONE can contribute to meet the necessary requirements, we will start from the successive phases of the design of a well-considered and purposeful monitoring scheme. The following chapters are to a great extent based on Wouters et al. (2008) and Onkelinx et al. (2008) and are partly a translation of their report.

Different frameworks for the development of monitoring programmes have been proposed. An example of such a framework is given by Vos et al. (2000). In this framework, the authors distinguish 7 components and some 'external constraints' that have to be discussed, decided and dealt with, in order to establish an effective monitoring scheme :

- Monitoring objectives
- Objects and variables
- Sampling strategy
- Data collection
- Data handling
- Maintenance
- Organization

In a recent report of the Research Institute of Nature and Forest - INBO, commissioned by the Flemish Environment Agency, a comprehensive methodology to develop appropriate monitoring schemes is presented (Wouters et al. 2008, Onkelinx et al. 2008). These authors discuss the different stages of the design and implementation of a monitoring programme related to the role of the different parties involved. Their framework consists of 5 phases:

- **Phase I. Prioritization of the information needs.** Through close collaboration between the user, client or sponsor and the designer of the monitoring scheme, the key questions that have to be answered by the monitoring and the specifications for the different components that follow from that, are elucidated.
- **Phase II. Development of the sampling design and the data collection.** Detailed specification of the required data collection and the associated costs in accordance with the information needs and the available budget.
- **Phase III. Planning of the data analysis.** Detailed elaboration of the management, analysis and interpretation of data and expected results as a reflection on the future monitoring results.
- **Phase IV. Planning of the reporting and communication.** Elaboration of a functional communication strategy with regard to the information needs of the sponsor and other stakeholders.
- **Phase V. Implementation and quality assurance.** Arrangement of operational aspects of budget, personnel, quality check, prior to the start of the proper monitoring. This phase only starts when mutual agreement in the former stages is reached.

The design of a monitoring programme is characteristically an **iterative process**, with feedback loops between the components of the framework. The elaboration of a particular phase however, can be done without considering the following ones. Indeed, while selecting and defining the key questions

(phase I of the above framework), it is not yet necessary to think of the way the data will be collected (phase II) or analysed (phase III). So, every step builds on the results of the previous one. However, the interaction and interdependency between the phases may equally require that previous options are adapted accordingly. For example, considerations regarding the sampling design may lead to the revision or even rejection of a set priority question and requirements regarding the data analysis may influence the agreed data collection and even the initial key question. This mutual interdependency has to be understood and accepted by everybody involved in the design of a monitoring programme.

Position of EBONE

For the monitoring procedure developed in EBONE, attention was paid to the different components that compose an applicable monitoring system. The organization and the tasks of the work packages guaranteed this. Also the development through a series of successive phases was partly respected. During the course of the project, the different phases of the framework were discussed with the partners to ensure that EBONE would deliver a transparent and appropriate methodology. We deviated regarding the active collaboration of users and other stakeholders; it's to say, the initial determination of the information needs, the users' requirements and interests, the definition of the monitoring conditions, etc., was done on the basis of a literature and document analysis, rather than by regular discussions and consultation with the users community (Henle et al., 2010; Parr et al., 2010). However, the annual meetings with the Advisory Board provided opportunities to discuss ideas and intermediate results, while the targeted workshops with stakeholders during the last year of the EBONE project, yielded very valuable feedback on the proposed monitoring starting points, methodologies and arrangements.

4.2. Partners in monitoring

The preparation of a successful monitoring programme demands for a close collaboration between clients and developers. The **clients**, for instance policy makers, site managers, and other users, define in the first place the demand and the context. It is their responsibility to be more explicit about their information needs and use of data. They should realize that not everything is possible, that they have to set priorities. Too often, demands are vague and not focused, and users expect an all-in-one methodology with results that can be used for a variety of questions. The **developers** at the supply side are monitoring scheme designers, data collectors, data analysts and data holders. They should be aware to start from the demand side and realize an effective scientific underpinning of an appropriate methodology. With the joint completion of the framework, the clients and suppliers ensure that agreements and decisions made are clear and unequivocal and the monitoring programme may be conducted successfully.

Position of EBONE

As EBONE focuses on biodiversity monitoring on a pan-European scale, the 'clients' of the EBONE monitoring methodology are above all the European Environment Agency and the European Topic Centre on Biological Diversity. They are in charge of compiling data and reporting on the 'state of nature' on the EU level. It is an objective of EBONE to deliver a European contribution to the development of a global biodiversity observation system, and thus the CBD and GEOSS can also be considered as users or stakeholders. The international Panel on Biodiversity and Ecosystem Services, IPBES, recently established by the UN, will equally become an important user and initiator of global harmonized biodiversity monitoring. Last but not least, most biodiversity monitoring in Europe is organized by the countries and regions. In this respect, the ENCA-network, the Heads of European Nature Conservation Agencies, should also be considered as priority clients.

4.3. Objectives, function and context of monitoring

Objectives of monitoring activities

Although the term monitoring is widely used, its meaning is less unequivocal. This is because objectives and functions of 'monitoring' activities and programmes may vary considerably. The objective can be the periodical and standardized recording of variables in order to assess and describe the state and evolution of a 'target population', *without or with comparing* it with the initial situation or a priori set standards or reference values. *Standardized data collection without any comparison*, is referred to as **surveillance**, *with comparison*, it is proper **monitoring**. The latter implies that the design of the data collection is such that it meets the statistical requirements to detect divergence from the set standards with a reasonable probability. From the foregoing, we can conclude that a number of data collection programmes for biodiversity policy are wrongly defined as 'monitoring'. Indeed, often standards or clear targets are not explicitly defined, and hence a direct link with biodiversity policy is not always obvious. Comparison against predefined standards, targets or reference values, or against the initial or previous situation in the case of change detection, allows evaluations to be made. If needed and depending on the evaluation result, actions can then be taken by policy to optimise the effect of policy measures (through legislation, incentives, etc.). The crux of policy-relevant monitoring is thus to provide policy makers with a reliable tool for evaluation and eventually adaptation.

Position of EBONE

EBONE is designed to function as a proper **monitoring** scheme. Indeed, the statistical underpinning of the sampling strategy and the rule-based data collection protocols, allow for reliable change detection and comparison against targets and reference values. Moreover, the explicit reference to the Annex I habitats of the Habitats Directive (Bunce et al., 2010), makes EBONE a policy-relevant monitoring.

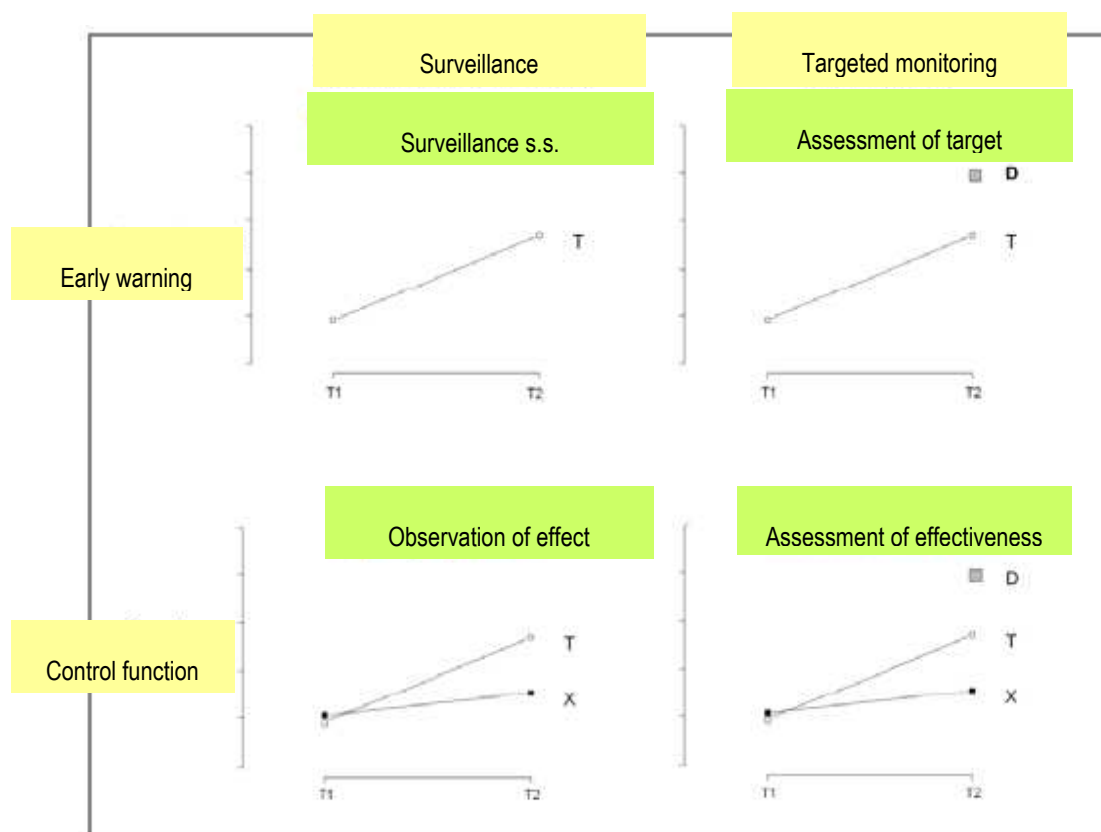
Functions of monitoring activities

Regarding the function of data collection programmes for biodiversity policy, Onkelinx et al. (2008) distinguish between a **signaling function** and a **control function**. With a signaling function, 'early warning' is pursued. To yield reliable results, changes in the performance of a variable (for instance decline of a population) should be identified with a statistical test. That means that an appropriate sampling design is necessary. Evidence of change however, is not sufficient to decide about the next steps, 'what to do?'. Nevertheless, active conservation (for instance implementation of protection plans, etc.) is often recommended. A second option is to wait and to initiate proper research to elucidate the causes and mechanisms of the observed change. These will then provide the knowledge base for further tailored conservation action (Nichols & Williams 2006).

With the control function, the assessment of the effect of particular policy or management measures, societal activities and developments, is the purpose. Of course, this approach starts from the assumption that there is a direct relationship between the observed changes and the relevant measures. However, this is not self-evident. Targets can be achieved due to favourable conditions that are independent from the implemented policy measures (for instance good weather conditions), or they gain from measures taken in another policy sector than the one that is analysed. Monitoring for a control function thus requires in principle a strict design based on random sampling of variables that are affected (the group where the measures are applied) and unaffected (the control group) by the policy measure. In this case, the monitoring focuses precisely on the information needed to make policy decisions.

Monitoring with a control function is also referred to as **targeted monitoring** (Nichols & Williams 2006). It is integrated into conservation practice and its design and implementation are based on a priori hypotheses and models of how the ecosystem will respond to management or policy measures. It is this approach which makes targeted monitoring especially suitable for *adaptive management*: it provides precisely the information needed to make conservation decisions.

Combining the two objectives and two functions of repeated biodiversity data collection, yields four types of data collection strategies.



D = quantified target / objective; X = state without policy / management measures

Although working with neatly formulated hypotheses is broadly accepted good practice in ecological research and conservation biology, surveillance monitoring is still widely used as a means to gain useful knowledge for biodiversity conservation. It is frequently characterized as ‘*omnibus*’ *surveillance monitoring* because of its supposed potential use for a diversity of conservation purposes and its inclusion of many different species and ecosystem components (Nichols & Williams 2006; Yoccoz et al., 2001). Although information can be obtained indeed, it is far from efficient, since the information content highly depends on chance, which is, given the limited resources for monitoring, not the best strategy.

Position of EBONE

Above all, EBONE is designed to serve the **signalizing** function. However, when the sampling scheme is supplementary stratified according to protected / non-protected sites and the number of sampling sites is increased proportionally, EBONE can equally well have a control function.

Context of monitoring

The purpose of a monitoring programme will also differ dependent on its context. This refers to the level of decision making the monitoring relates to.

In a **strategic** or **programme context** the monitoring generates general information on state and evolution of biodiversity to support strategic decisions, normally embedded in high(er) level policy and management. This concerns estimates of needs and priorities (corresponding with the signaling function), as well as assessments of a series of measures (corresponding with the control function).

In a **project context** the monitoring focuses on the state and evolution of specific components in order to address concrete problems or measures, often associated with the level of site manager. That concerns the detailed monitoring of where, how and to what an extent biodiversity evolves (corresponding with the signaling function), as well as the assessment of the effect of a particular measure (the control function).

Position of EBONE

Meant to yield information on stock and trends on the European level, EBONE operates clearly in a **strategic context**.

To discriminate between strategic and project monitoring is important as it is seldom possible to serve both aims with one monitoring scheme. This can be illustrated with the *monitoring of management results* in nature reserves in Flanders (Belgium) (De Cock et al 2008). The purpose of this monitoring was “to give, in an objective way, a general valuation of the evolution of properly managed nature reserves in Flanders. The valuation of the management is done with regard to the set ‘nature target types’, for instance Annex I habitats, as defined in the management plan of each individual nature reserve”. This purpose description is a combination of both strategic and project contexts. Assessment of particular nature target types at a regional level (Flanders), is of interest to policy makers and authorities responsible for the reporting of the conservation status of habitats (in the context of Art. 17 of the habitats Directive), and refers to the strategic context. The information on the effectiveness of management in the different nature reserves supports local operational decisions of the local manager and refers to the project context. Mixing both contexts is not evident because they deal with different ‘target populations’: the total set of an Annex I habitat type in Flanders for the strategic context and the variety of that habitat type in the individual nature reserves for the project context. A representative random sample of the ‘Flemish target population’ will yield statistical reliable conclusions for the regional level. However, for each individual reserve involved, the number of sampling units from that reserve, if any, will be too small to decide about the state of its habitats in relation to the management applied. Indeed, the potential variety of the habitats and their response to management measures will be insufficiently covered, and thus local managers will gain unreliable conclusions regarding their management effectiveness.

The admission of the differences between the monitoring types and their associated requirements regarding the sampling design, and the choice for a defined purpose when a monitoring programme is agreed upon, is often lost during the course of its development. Quite often, it is pursued to combine individual project context monitoring schemes of particular elements (for instance habitats) in order to gain information of the whole of the population of these elements. From a statistical point of view, this is not justified because the individual elements are not a totally random sample of the a priori defined target population.

The evolution of the content of the monitoring programme of nature management in Flanders (De Cock et al. 2008) is illustrative in this respect. The programme started in a strategic context with question such as “Does the management in nature reserves add to the maintenance and restoration of biodiversity in the region?” and “Is the management applied in general efficient to achieve the set nature targets?”. During discussions of the steering committee, the original purpose widened as it was the request to increase the usefulness of the monitoring results (“make it more concrete”). And thus new tasks were added, such as “Develop a protocol for data collection, suitable for a quality assessment at a regional scale, as well as for the local management monitoring”, “Compile multispecies lists for the nature target types to be monitored” and “Monitor the effect of management measures. So, standardize and classify the measures in order to obtain uniform management types as a basis for a random selection of sample sites”. These are questions typical for a project context. At the end, a disappointing monitoring programme was presented. Indeed, to respond to the increased expectations, a long list of variables (species, habitat and vegetation types and characteristics) had to be monitored, each according to a detailed and elaborated protocol. Through that, the programme risked to exceed the available resources and thus the repeated monitoring in the future. Detailed questions (project context) also demands for an adapted sampling design with a sufficient number of replica sampling units. However, the total number of local management variants related to the set nature target types yielded a large number of combinations each however with too few elements to allow statistically rigid random sampling. Then, after a test monitoring and ample consideration it was finally decided to handle the strategic and the project oriented questions separately and develop a specific monitoring strategy for each of them.

Taking similar considerations as the above into account, Lindenmayer & Likens (2010) distinguish between three types of (long-term) monitoring:

- **Curiosity driven or passive monitoring.** This is a type of monitoring without a specific and pre-defined question. Because hypotheses are lacking and assessment of management measures or experiments make no part of the monitoring, little if any new understanding of the target ecosystems is gained. Nevertheless, sustained and rigorous but mere data collection can sometimes, by accident, yield information that is useful to elucidate ecological phenomena. But because this is purely a matter of chance, it is a very un-efficient way to spend the limited resources for biodiversity research and conservation.
- **Mandated monitoring.** This is monitoring carried out according a governmental obligation, a political directive, etc., and thus is not without engagement. At its base is a broad policy relevant question. The focus is usually to identify trends and, more extended, to assess the effectiveness of policy and management measures. The obligation to report every 6 years on the implementation of policy and management measures and the conservation status of species and habitats under the Habitats Directive (art. 17), is a good example of this type of monitoring.
- **Question-driven monitoring.** This type of monitoring is carried out as an integral part of a scientific research. It is based on a conceptual model of the studied system and guided by a clear experimental design. The monitoring yields data to be used in the testing of a priori prediction, the hypotheses. The big advantage of this type of monitoring is its predictive capacity and contribution to the real understanding of cause - effect relationships in the studied ecosystems and other entities. Knowledge increases and can lead to new questions, in its turn pushing the monitoring to develop into **Adaptive Monitoring** (Lindenmayer & Likens, 2010). Decision makers and managers may also benefit from the results through targeted improvement of the measures they apply. Monitoring activities carried out in the framework of the Long-Term Ecological Research (LTER) are excellent examples in this respect.

Lindenmayer & Likens (2010) recognize that these categories are not mutual exclusive. There can be overlap. A statistical framework can be developed for both a mandated and a question-driven monitoring programme. And one monitoring programme can be built of both strategies, as is the case for the EBONE programme we propose. Finally, mandated monitoring is often coarse-scale, ideal to assess the state of biodiversity, but weak to elucidate the mechanisms giving rise to this condition. Question-driven monitoring on the contrary, is normally fine-scaled and process-based, but hardly suitable to make valid spatial extrapolations (Lindenmayer & Likens (2010)). The difference between the Countryside Survey (more or less mandated monitoring) <http://www.countrysidesurvey.org.uk/> and the Environmental Change Network (question-driven monitoring) <http://www.ecn.ac.uk/>, is obvious in this respect.

Position of EBONE

Although there is no official assignment for EBONE to deliver data and information for a specific policy purpose (EBONE is a project under EU DG Research FP7 programme, and not commissioned by DG Environment), the decision to start from the SEBI 2010 indicator needs and the explicit reference to the Habitats Directive reporting obligations, makes EBONE a **mandated monitoring**. The choice to focus on habitats and their characteristics as the variables of monitoring, is based among others, on the assumption that there is a relation between habitat type and species composition. As this is a hypothesis that can be tested with the EBONE approach if collected habitat data are completed with species data, EBONE can be considered to be a question-driven monitoring just as well.

It is clear that objective, function and context of a monitoring may differ considerably. To develop an appropriate and efficient biodiversity monitoring programme or to optimize an existing one, it is therefore necessary that users / stakeholders and designers / scientists discuss these issues together and reach mutual agreement. The idea must be accepted that effectiveness and efficiency of a monitoring programme depend to a very large degree on the will to clarify and define precisely the information needs that underlie the development of the monitoring programme.

4.4. Defining priority questions

Quite often, the information need of the demand side is only formulated in general terms. At first sight, this may seem reasonable, as policy or management objectives are often only qualitatively defined as well. This poses problems however, the moment it has to be decided when and where to start action to control or counteract an observed change. Therefore, one needs predefined threshold values, norms or standards, or at least a baseline to compare with. So, users together with designers of monitoring programmes should discuss the information need in detail in order to define priority questions which ensure that a monitoring programme is set up that will yield relevant information. Clearness about the information need will also clarify the objective, function and context of the monitoring that is required, and hence the conditions for its development. It may also become clear that a former general question breaks up in a series of successive questions of which only part can be dealt with in the monitoring programme under construction.

To be able to construct a scientifically sound monitoring scheme that yields reliable and consistent data, the priority questions have to be reformulated in 'measurable' questions, problem statements or units. For example, the priority question "*Is, in a particular region, the stand still principle respected with regard to nature quality? Are areas with good nature quality maintained?*" gives no clues how to measure and assess a situation. Therefore, the question should be clarified. So, "*Have maximum 10% of the areas with a good quality in the former monitoring round now a lower quality?*" and "*Is the*

decrease of the number of areas with a bad quality sufficient to reach the target of 0% areas with bad quality in 2015? are measurable questions and thus good alternatives. Concluding, in general priority questions should be reformulated so that measurable estimates can be derived or they should be defined as hypotheses that can be tested statistically.

From the former, it will be clear that clarification of the priority questions is also needed to decide upon the technical characteristics and requirements of the monitoring methodology, for instance selection of the proper variables, the sampling intensity, etc. Ideally, the objectives for the monitoring should be specified in terms of

- a target or threshold value (number/area, etc.)
- a statistical measure (mean, percentage, proportion, etc.)
- a definition of the desired statistical confidence level (concentration $a \pm b$)
- a spatial and/or temporal scale (rural area, current year, etc.)
- a period within which the objective has to be achieved (by 2020, etc.).

The policy objectives regarding forest composition in Flanders (B) are illustrative in this respect. The objectives of the Nature and Forestry Agency are indeed:

- *“To reduce, on the long term (this is still to be specified), the average volume of exotic tree species to maximal 20%”;*
- *“Homogeneous exotic species plantations have to be changed into mixed stands with at least 30% native species”;*
- *“In public forest the amount of dead wood (relative to the total wood volume) has to be 4% on average”;*
- *“By ?? (still to be specified), 5 to 15% of the forest area has to be ‘open patch’, with each patch being 3ha at the maximum”.*

Such explicit and measurable objectives are the exception rather than the rule. On the European level, the majority of the former biodiversity policy objectives are formulated qualitatively. However, in the EU Biodiversity Strategy to 2020, a few objectives are formulated more precisely (European Parliament resolution of 20 April 2012 on our life insurance, our natural capital: an EU biodiversity strategy to 2020 (2011/2307(INI))):

- *“in order to establish a clear pathway to achieving the 2050 vision, at least 40% of all habitats and species must have a favourable conservation status by 2020”, while*
- *“by 2050, 100% (or almost 100%) of habitats and species must have a favourable conservation status”.*

Regarding the maintenance and restoration of ecosystems and their services, the Strategy is equally clear when referring to the CBD requirement:

- *“...to restore 15% of degraded ecosystems by 2020 (as a minimum)”.*

Position of EBONE

Priority questions for EBONE were not defined a priori. It was however one of the tasks of work package 1 and 2 to analyse the information needs and priorities related to the European and global (CBD) biodiversity policy. This resulted in the selection of three main headline indicators from the set of 26 of the SEBI (“Streamlining European 2010 Biodiversity Indicators”) process (Henle et al., 2010; Parr et al., 2010):

- extent and change of habitats of European interest in the context of a general habitat assessment
- abundance and distribution of selected species (birds, butterflies and plants)
- fragmentation of natural and semi-natural areas

As these indicators are widely accepted by the EEA partner countries and form part of any standard set that most probably become part of a Global or European Biodiversity Observation Network (GEO BON), and given the current lack of data, this selection will meet the existing information needs. It must be noticed however, that these indicators do not relate to quantitatively formulated objectives. Therefore, EBONE itself formulated some quantitative objectives to be met by the monitoring scheme (see chapter 7 of this report). So, the sampling design should be able to precisely deliver stock estimates for habitat types that are present in 5% or more of the sampling units, and detection of an average of 1% per year decline for a habitat type must be possible. The latter coincides with the requirements from the Habitats Directive.

5. Monitoring optimization: choosing the right variables

This chapter focuses on the selection of variables in relation to the objectives of a monitoring program and its effectiveness. Special attention is given to (1) the selection of appropriate variables to be measured in EBONE, (2) the use of the 'habitat' concept as a conservation umbrella, (3) accounting for 'cause-effect' relationships, (4) the early-warning function of variables, and (5) the need for an adaptive monitoring.

5.1. Selection of appropriate variables to measure

Progress in many monitoring programs is hampered by wrangles over what to monitor. Lindenmayer & Likens (2010) see two approaches that are commonly but often unsuccessfully employed to tackle this problem are:

- (1) Monitoring too many variables: a common response to unresolved arguments about what to monitor has been to monitor a large number of variables (the so-called 'laundry list'). However, such a 'laundry-list' approach can have a range of problems. First, it can divert attention from posing well-crafted and tractable questions. Second, resource and time constraints frequently mean that a poorly focused 'laundry list' will result in many things being monitored badly. Third, a 'laundry list' may make a monitoring program too expensive to be sustained financially beyond the short-term and may ultimately lead to its collapse. Finally, a 'laundry-list' approach can create problems with the statistical design of a monitoring program. Monitoring a long list of entities can only ever realistically be done on a few sites, but often a robust statistical design for a monitoring program will call for many sites to be subject to repeated survey, such as to provide sufficient statistical power to detect an effect or identify a trend.
- (2) Monitoring indicator species: An alternative approach focuses on 'indicator species' or 'indicator groups' as the targets of monitoring programs (Cantarello and Newton 2008; Dung and Webb 2008). The application of indicator species however has to be treated with extreme care. Indeed, some problematic issues can arise when using 'indicator species'. E.g. it is rarely explicitly stated what the indicator species or groups are indicative of, particularly at the ecosystem level. Also there is often a general lack of understanding of causal relationships between indicator species and the entities they are assumed to indicate (Lindenmayer et al. 2000; Wright-Stow and Winterbourn 2003). Indeed, in several cases where causal relationships have been carefully examined, a proposed indicator species subsequently has proven to be a misleading surrogate of environmental conditions or the occurrence of other biota. Scepticism on the indicator species and other surrogate approaches is important because it is critical for resource managers, policy-makers and scientists to be made aware that a particular species may not be a good indicator of the presence or abundance of other species, of the suitability of the ecosystem to support species assemblages, or of ecosystem processes. Otherwise they may believe that by conserving a so-called 'indicator species' they have effectively conserved all other biota when in fact they have been deceived by a false sense of security (Lindenmayer et al. 2009).

It has been suggested by Lindenmayer and Likens (2010) that the problems with 'laundry lists' and indicator species can be avoided by carefully crafting questions at the onset of a monitoring program. Selecting the right variables inevitably begins with asking the right questions, using a well-conceived model to help conceptualize the goals of the monitoring system. This key step will help identify those

entities most appropriate for monitoring. Such a focused approach obviates the need to measure a vast array of things and ensures that a subset of entities can be monitored well, rather than vice-versa. In addition, making direct measures of targeted entities ('target variables') avoids the need to assume that those entities are surrogates or indicators of other entities.

One of the main goals of the European Biodiversity Observation Network is to provide data to support the development and reporting of biodiversity indicators. Of course, biodiversity indicators span broad levels of biological, spatial and temporal organisation within ecosystems and the options for choosing variables are still almost infinite. As outlined and discussed in deliverable 1.1 (Parr et al., 2010), the EBONE methodology focuses on contributing to three main pan-European SEBI biodiversity indicators covering:

- (1) The extent and change of habitats of European interest (in the context of a general habitat assessment)
- (2) Abundance and distribution of selected species (birds, butterflies and plants)
- (3) Fragmentation of natural and semi-natural areas.

A common denominator that is able to functionally address these targets seems to be the concept of 'habitat'. Choosing **habitat as the key variable** has a number of advantages. Firstly, it directly relates to General Habitat Categories and to the Habitat types of European interest (Annex I of the Habitats Directive). For EBONE, a major goal is to be able to make stock and change estimates for the general habitat categories defined in the Handbook (Bunce et al. 2011), as well as for Habitat types of European interest (Annex I). As there is a clear relationship between habitat structure and the presence of species, the concept also indirectly relates to species distribution and abundance. It has been shown that general habitat categories relate well with species composition, not only for plants, but across a range of different taxonomic groups (Olsvig-Whittaker et al. 2010). There is no doubt that particular species and sets of species are of high priority in terms of biodiversity conservation and therefore should be the focus of European wide long-term monitoring programs. Building monitoring schemes for particular species groups however, is not an objective of EBONE, as their monitoring is already well covered and organized, especially for birds and butterflies. Birdlife International, the European Bird Census Council, and Butterflies International integrate various schemes existing in Europe, which is possible because the methodology is comparably well standardized, and data storage and sharing are well organised. Any further improvements should run under the lead of these existing networks. Thus, despite being high priority, it will make no sense for EBONE to include species monitoring into its core procedure, unless these networks signal specific needs of support (Henle et al. 2010). Finally, the configuration of 'habitats' on a landscape scale directly relates to fragmentation of natural and semi-natural areas and hence is well fitted to serve the third indicator.

Position of EBONE

As appropriate variables for monitoring, EBONE choose to develop and test an elaborated and extensive list of '*General Habitat Categories*' based on plant life-forms, together with qualifiers for additional, but essential information on habitat characteristics and decision rules for reliable recording. The fieldwork in different countries proved its applicability in a wide range of environments and landscapes. However, adaptation to local conditions and biodiversity characteristics can be necessary and is even recommended as it increases the local usefulness of the system.

In Israel for example, because of the deviant nature of desert and steppe habitats, 'tree, shrub and low-shrub density' was added as a classification factor, where this is not included in the procedure described in the common EBONE Handbook. Also, dominant species define subcategories in Israel, while they are not used for classification in the original method. A detailed structure description is

recommended in the desert areas, elsewhere this is only mandatory. And the precedence rule is based on 10% cover in the sparsely vegetated regions of Israel, while in Europe the precedence rule is based on 30% cover (Margareta Walczak, 2011, communication on the EBONE WP9 workshop, Ein Gedi, 10-12 January 2011).

As long as the core of the General Habitat Categories is maintained, adding more details to the recording procedure poses no problems. On the contrary, such a flexibility allows to respond to local information needs and thus helps to meet, to a certain extent, the requirements of a variety of stakeholders and users. In practice, it is very likely that this ability of the EBONE methodology to be applicable for local purposes will be the driver for regional and national authorities to engage in a monitoring project that is part of an overarching pan-European biodiversity monitoring facility. If this could be achieved, then European as well as regional and national users could fulfil their monitoring requirements and obligations with the same monitoring programme.

5.2. Habitat as a conservation umbrella

A key element in the EBONE approach to biodiversity monitoring is that it considers 'habitat' to be a conservation umbrella for biological diversity. Although widely used, the content of the concept 'habitat' remains diverse, ambiguous, and difficult to be used consistently in monitoring (see Box 5). For a clear understanding of the use of the concept in EBONE, 'habitat' is defined as "*an element of land that can be consistently defined spatially in the field in order to define the principal environments in which organisms live*" (Bunce et al. 2011). Often 'habitat' is even more narrowly defined in terms of species requirements (e.g. the spatial extent of a resource for a particular species), indicating that habitat characteristics can be explicitly linked to individual species and their ecological requirements.

Not surprisingly, basic ecological premises indicate a clear relationship between biological diversity (species richness) and habitat structure, both at a local and landscape scale. Numerous studies have found that the composition and complexity of habitats could be a good indicator of overall biodiversity (Cornelis and Hermy 2004; Honnay et al. 2003; Hermy and Cornelis 2000; Wessels et al. 1999). In this respect habitat structure mainly refers to the vertical and horizontal structure of the vegetation (vegetation layers; composition and height variability of trees, shrubs, forbs and grasses, spatial patterns).

In EBONE plant life-forms are used to describe habitat structure. The central pillar of the General Habitat Categories (GHC) is that there is a relationship between plant life-forms, as described by Raunkiaer (1934), and the environment. The GHC surveillance and monitoring approach provides rules for consistent recording in the field of life-forms that make up habitats. Habitats are important for biodiversity in their own right, as described in the Habitats and Species Directive of the European Union, but can also be used as a framework for sampling other aspects of biodiversity, e.g. vegetation and spider assemblages (Bunce et al. 2006). Habitats are linked to species occurrence. For instance, birds as the bittern (*Botaurus stellaris*) are only found in reed beds and the large blue butterfly (*Phengaris arion*) only in calcareous grasslands. This is reflected in landscape level models such as LARCH (Opdam et al., 2003; Verboom and Pouwels, 2004). A further essential reason for using life-forms is that many animal species (e.g. birds and butterflies) respond to habitat structure rather than to individual plant species (Bunce et al. 2006). Plant life-forms, as described by Raunkiaer (1943) define to a high degree the structure of the habitat and thereby the habitat quality for species (Bunce et al. 2008). In this respect, it is encouraging to note that, field tests in Israel with the General Habitat Categories have shown good predictive power for species composition of both plant and animal taxonomic groups (Olsvig-Whittaker et al. 2010).

Furthermore, the vegetation structure of habitats can link biodiversity with other environmental or ecological indicators, thus integrating the form and function of biodiversity (Whitford et al. 2001). Therefore, vegetation structure, as an indicator of biodiversity, has both ecological credibility and the potential to accurately simplify complex interactions between species and habitats (Tzoulas & James 2010). Monitoring procedures that use vegetation structure as an indicator of biodiversity thus have the potential to bridge difficulties of understanding between disciplines (Whitford et al. 2001; Hercoc 1997; Tzoulas & James 2010).

The concept of general habitat categories, which has its roots in phytogeography, seems to be closely related to the concept of plant functional traits (e.g. McGill et al. 2006). The links between both warrant further investigation.

Box 5: Habitat definitions

Although widely used, the content of the concept 'habitat' remains diverse, ambiguous, and difficult to be used consistently in monitoring. There are various reasons for that, not at least because 'habitat' is used in very different contexts. Three are obvious (Hall et al. 1997; Dennis, 2010; Dennis et al. 2003):

- The *legal, juridical* context. Habitats are referred to in for instance the EU Habitat Directive with the Annex I habitats. Here, a typification is presented of habitats described in general, with minor reference to scale issues or diagnostic features. Criteria to identify a particular habitat or to distinguish between them are not included.
- The *nature conservation policy* context. This is the domain of the 'favourable conservation status' of habitats, of the 'significant impact' on habitats, of the EUNIS classification, etc. There is at one hand a link to the typology presented in the legal documents, while on the other hand the habitat types have to be interpreted to concrete situations in order to assess their characteristics and functioning.
- The *scientific, ecological research* context. In this sense, the habitat is defined as the spatial extent of a resource for a particular species. Habitat is thus explicitly linked to a species or species group sharing the same ecological requirements. The 'habitat patch', 'micro-habitats', 'temporary habitat', are notions used in this respect.

The last scientific definition also shows an evolution in the meaning of the term habitat from vague and broad to narrow and precise. The following definitions illustrate this:

- "*Place, living space, where an organism lives*" (Odum 1963).
- "*Place where a species normally lives, often described in terms of physical factors such as topography and soil moisture and by associated dominant forms (e.g. intertidal rock pools or mesquite woodland)*" (Calow 1999).
- "*Habitat is a zone (area) comprising a set of resources, consumables and utilities, for the maintenance of an organism. The resources occur in union and/or intersect and may also be equivalent; links between resource outlets are established by individual searching movements of the organism*" (Dennis and Shreeve 1996).

5.3. Variables accounting for causal effects

Traditionally, many long-term monitoring surveys have been initially planned and designed only to provide estimates of state and trends in biodiversity. However, with successful surveys, additional objectives become more important over time, as interesting changes occur in the sampled population (Olson & Schreuder 1997). An additional monitoring concern will be to relate observed changes to underlying causes. Indeed a monitoring programme can be developed to report exclusively on the state and change of biodiversity and can yield relevant results, but for policy and management it often remains unclear what measures could be taken to stop the observed evolution, if this was assessed as negative (De Blust & Van Olmen 2004). Therefore it is recommended that a European wide monitoring programme should not simply aim to document status and trends in biodiversity components (habitats or species) but also to link these changes to possible causal effects, thus facilitating identification and establishment of possible causality. This is mainly the case where we have reasonable a priori hypotheses of what may cause observed changes (Lindenmayer & Likens

2010). To detect and understand underlying mechanisms of biodiversity change, a smart selection and combination of variables is necessary and requires combined monitoring of target and background variables.

Linking observations directly to specific causes will require that potential causal factors are explicitly incorporated in the design on the basis of a predefined conceptual model of the system. This is however only achievable in controlled, random experiments. In large scale monitoring projects it is impractical or impossible to implement experimental designs, which is the most robust way to make clear inferences about causal factors, into the core procedure (due to spatial, temporal or resource limitations). An integrated monitoring design however, including a smart selection of 'stressor' related or 'background' variables can enable preliminary investigations into potentially causal relationships (Olson & Schreuder 1997, De Blust & Van Olmen 2004, Lindenmayer & Likens 2010). Such a design will not allow any definitive conclusion on causal factors, but strong correlations may strengthen the hypothesis of such a relationship and can initiate further experimental design-based studies or monitoring to further investigate possible causal factors, for example within the Long Term Ecological Research Network (LTER) or the Long Term Socio-Ecological Research Network (LTSER).

Integrated, nested monitoring

The task to regularly document on the state and change of biodiversity, and at the same time shed light on the processes involved in its change, means that a comprehensive and integrated monitoring scheme has to be elaborated. Therefore additional variables may be necessary. These additional variables however need to address plausible hypotheses developed from careful assessments of potential causes of change. One way forward is that the monitoring program should address a framework or conceptual model that explicitly links the different factors involved in cause-effect chains, e.g. the *Pressure – State – Response (PSR)* Model of the OECD or the *Driving forces – Pressures – State – Impact – Responses (DPSIR)* Model used by the European Environment Agency (Antrop et al. 2000, Van Olmen 2000, De Blust & Van Olmen 2004). As these cause-effect chains are also a common rationale used in environmental management and policy, monitoring gains credibility and utility the more it coincides with these different phases of environmental and nature policy. To be able to address the successive factors of the ecological disturbance chains, a monitoring programme should include a set of variables and indicators that relate to the different phases. Variables are then selected according to their role in the DPSIR-conceptual model and integration in the monitoring procedure follows the functional interrelations between these variables (De Blust & Van Olmen 2004).

In terms of accounting for causal relationships, a fundamental character of such an integrated monitoring is that surveillance of the series of linked variables is spatially nested. Although variables in an integrated monitoring are selected because of their mutual interdependencies, a spatially nested data sampling alone does not cover these interrelations automatically. Data recording has also to be done in a proper sequence and within the appropriate period and time interval. And then again, it is clear that there can be an important time lag between driving forces, the changes these induce in the environmental and landscape characteristics and the ultimate impact they have on biodiversity change. Consequently, integrated monitoring of biodiversity at the landscape scale should be a dynamic process that combines interrelated data collected according to their appropriate spatial and time scale.

In general, the implementation of this integrated approach starts with an analysis of the possible environmental disturbance chains occurring. This must yield the variables that are important to assess. The table 5.1 gives an example of an integrated monitoring programme for landscape and biodiversity of the Flemish countryside (Antrop et al. 2000, Van Olmen 2000, De Blust & Van Olmen 2004) that works with a series of linked variables and indicators that addresses pressure (e.g. land

use, emission, ...), state (e.g. nutrient concentrations, pollutants loads, habitat structure, ...) and impact indicators (consequences to biodiversity).

Position of EBONE

The EBONE monitoring design, as we envision it, can facilitate further analyses by partially anticipating probable causal factors for inclusion in the procedure. Although it is not possible to anticipate all variables related to potential causal effects of biodiversity change, we believe it is possible to include a subset of variables in the core procedure that enable preliminary causal analytic investigations to be conducted. In the current EBONE monitoring design, possible causal relationships are accounted for by including a selection of environmental and management qualifiers into the procedure (“background variables”). This additional background information is collected to improve the chances of identifying possible hypotheses. Also the recording of General Habitat Categories, directly relates to land use and landscape patterns. As changes in land use and landscape patterns are an important driving factor for biodiversity change, this information is also very important in establishing possible causal relationships. In order to further optimize EBONE monitoring in terms of detecting possible causal relationships, new selections of variables or indicators can be added.

Long Term Ecological Research Network (LTER)

An integrated monitoring design will probably not allow any definitive conclusions on causal factors, but strong correlations may strengthen the hypothesis of such a relationship and can initiate further experimental design-based studies or monitoring, to further investigate and document possible causal factors, for example within the Ecological Change Network (ECN - UK) or the Long Term Ecological Research Network (LTER). The LTER Network is a collaborative project investigating ecological processes over long temporal and broad spatial scales. The Network promotes synthesis and comparative research across sites and ecosystems and among other related national and international research programs to support research on long-term ecological phenomena. The LTER network is essential in explaining the mechanisms behind widely occurring and extensive changes. For effective European biodiversity monitoring we thus propose a parallel design consisting of a large-scale, long term ongoing survey as the core of the monitoring (EBONE), accompanied by supplemental analytical design studies or surveys (ECN, LTER, i.e. intensive monitoring on few, purposively chosen sentinel sites with the aim of uncovering causal effects or trends in variables with low spatial variability through analysis of long time series).

	Pressure	State indicator	Impact indicator
Desiccation	Area of parcels with subterranean drains. Total volume of permitted groundwater extraction.	Water level in gauges and ditches. Groundwater quality expressed as conductivity and ion ratio.	Number of obligate phreatophyte plant species. Share of the different moisture plant indicator classes (sensu Ellenberg*) in the total flora.
Eutrofication	N and P emission from local sources (e.g. total number of cattle and pigs).	N deposition (wet and dry) measured in (semi-) natural vegetation to allow comparison with critical loads **. Soil P saturation in representative parcels.	The proportion of clearly dominant plant species in the herb layer . Share of plant species characteristic for oligo- to mesotrophic conditions (sensu Ellenberg*) in the total flora.

Acidification	Potential acidifying emission expressed as total acid equivalents.	Real deposition (wet and dry) as total acid equivalents in (semi-) natural vegetation to allow comparison with critical loads ***. pH of phreatic water.	Forest vitality, degree of leaf damage. Share of the different acidity plant indicator classes (sensu Ellenberg*) in the total flora.
Fragmentation	Increase /decrease of hard barriers (length / area). Presence of mitigating infrastructure (ecoduct, etc.).	Landscape metrics.	<i>Difficult to define in general..</i> Presence / absence of species functional groups according their dispersion strategies and capacities.
Erosion	Total area of land without vegetation cover in winter related to terrain slope. Presence of permanent vegetated talus and verges in raised areas.	Presence of eroded ground and gullies. Length of roads covered with mud. Organic matter content of the topsoil of arable land.	Area of un-vegetated patches in small landscape elements. Number of pioneer plant species in the total flora of small landscape elements.

*Ellenberg et al. 1992 ** Bobbink et al. 1998 *** De Vries 1998

Table 5.1. Examples of series of pressure, state and impact variables/indicators as used in the integrated monitoring programme for landscape and biodiversity of the Flemish countryside (Antrop et al. 2000, De Blust & Van Olmen 2004).

5.4. Variables and early warning potential

Next to delivering information on trends and possible causal relationships on key aspects of biodiversity, an effective biodiversity monitoring program should also provide an early warning system. This may allow to quickly tackle problems that might otherwise be difficult or expensive to reverse. In order to have a reliable system, the following conditions have to be fulfilled:

- (1) The selection of samples is random, representative for the target population and sufficiently replicated to allow strong inferences on the key objectives which are for instance state and trend estimation.
- (2) The observations made in situ have some advantages that allow to detect change in an early state which cannot be obtained from current remote sensing data:
 - o The use of life-forms appears to detect small but significant changes in habitat structure. As many animal species respond to (changes in) habitat structure, changes in the proportion of life-forms can be used to quickly signal possible changes in species composition on the landscape level, before major changes in General Habitat Categories, Habitat types (e.g. Annex I), or in landscape patterns occur.
 - o Additional information on environmental, site and management, collected as consistent qualifiers, can indicate changes in underlying ecological factors, before this is even noticeable in changes in life-forms, General Habitat Categories or Annex I habitat types.

The first prerequisite is obvious and should be a qualification of each appropriately designed monitoring programme. The second depends on the variables that are monitored and could thus be selected according the early warning purpose of the programme.

Position of EBONE

EBONE is designed to be able to provide an early warning system for relatively small, slow changes in the area of general habitat categories (provided they are not too rare), which would remain hidden without the monitoring network. Stronger changes such as large-scale environmental disturbances and major changes in habitat or species diversity (catastrophic change) will also be picked up, but such changes will probably have been noticed also without the monitoring design. Both aspects allow for improved and quicker decision making. This early warning and signalling function is addressed by two key elements of the monitoring framework. One relates to the sampling design (i.e. the selection of samples) (see chapter 7), the other relates to the observation methodology within sampling units.

5.5. Changing objectives... changing variables

As mentioned above, setting clear objectives and framing questions will help to avoid discussions on what to monitor, because that will then be based on the (priority) questions asked. Fundamental in this respect is that the question setting, monitoring design, data collection, data analysis, and data interpretation are iterative steps. This also means that a monitoring programme can evolve and develop in response to new information or new questions being posed; an adaptive monitoring scheme is appropriate in this respect (Lindenmayer & Likens 2009, 2010).

Indeed, a large-scale, long-term monitoring programme must also be able to address issues of changing objectives and analytic inference. Descriptive objectives (e.g. status and trend of habitats) may remain the central core of a programme, but the variables of interest may change over time. Ecological objectives will also continue to evolve for the foreseeable future as new methods for measuring biodiversity will be developed (Olson & Schreuder 1997, Lindenmayer & Likens 2010).

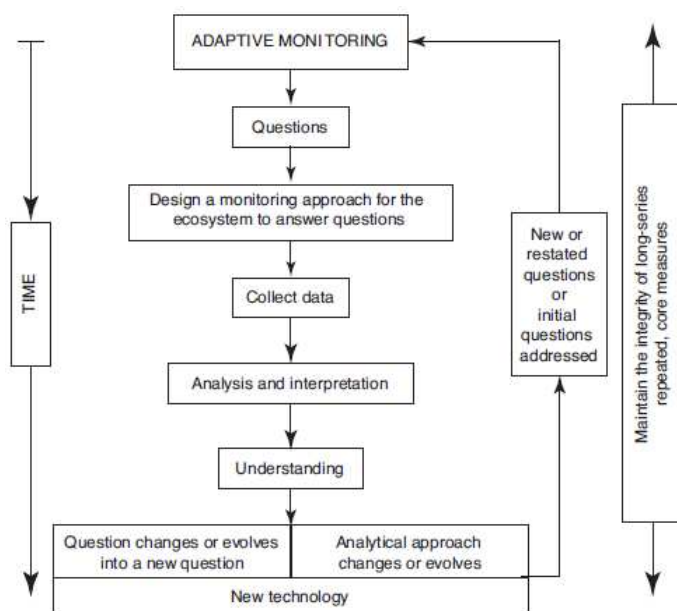


Figure 3.5 The Adaptive Monitoring framework. (Redrawn from Lindenmayer and Likens 2009, with permission from Elsevier)

Results from descriptive analyses lead naturally to analytical questions, including investigations of potential causes that can 'explain' the results, i.e. cause-effect studies (see also §5.3). As it is not possible to anticipate all variables related to potential causes, new insights might increase the interest in adding new variables. Several authors have highlighted the importance of evolving questions as part of a robust approach to monitoring programs (Hicks and Brydes 1994, Ringold *et al.* 1996). Clearly a way forward is to embrace an adaptive monitoring approach representing an iterative and linked framework that allows this evolution to take place in a logical way (Lindenmayer & Likens 2009, 2010).

6. Time and data management requirements related to the effectiveness of a monitoring programme

6.1. Time requirements of the EBONE protocol

Next to sampling design another crucial aspect in terms of cost-effectiveness of the proposed monitoring procedure, is the time required to carry it out. In this respect the field trials of habitat mapping and recording that have been organized in the context of WP6 of EBONE can be used as a specific case study on the efficiency of the methodological framework.

The EBONE protocol was tested in 9 countries² across the whole of Europe, covering 8 different environmental zones. Furthermore the protocol was also applied outside Europe, i.e. in Israel and South Africa. This ensures that the EBONE methodology was tested in many different situations and across a wide range of landscape and vegetation types. To assess the time requirements of the proposed protocol, a questionnaire was sent out to all partners involved in WP6. Data was collected on preparation time (field site selection, preparing field maps, ortho-photos, etc.), travel time to the field sites (back and forth), time spent to do the actual fieldwork (mapping and recording), recording effort (basic, intermediate or complete recording) and time needed for data input and data control. In total, data from 75 1km² “test sites” was received to perform this assessment.

Recording Effort <i>Basic</i> (n=11) <i>Intermediate</i> (n=25) <i>Full</i> (n=39)	<u>Basic</u> Habitat Mapping and Recording (Areal Elements + qualifiers) [person days/km ²] Average [Min – Max]	<u>Intermediate</u> Habitat Mapping and Recording (+ line/point elements and/or Annex I) [person days/km ²] Average [Min – Max]	<u>Full</u> Habitat Mapping and Recording (+ vegetation plots) [person days/km ²] Average [Min – Max]
Preparation	0,8 [0,5 – 1,0]	0,8 [0,4 – 2,0]	0,9 [0,3 – 2,0]
Travel to field site ¹	0,8 [0,3 – 1,0]	1,0 [0,1 – 3,4]	1,1 [0,1 – 5,0]
Field work ¹	4,1 [0,8 – 7,0]	5,2 [1,5 – 14,3]	6,3 [1,3 – 16,5]
Data input	2,0 [1,5 – 2,0]	3,0 [1,5 – 6,0]	2,8 [0,4 – 5,0]
TOTAL	7,7 [3,5 – 11,0]	10,1 [5,5 – 19,1]	11,1 [3,8 – 25,0]

Table 6.1: Estimates of time required for sampling and processing of EBONE habitat data. Time requirements are expressed in person days and per kilometre square; Average, minimum and maximum time are given for different parts of the protocol and for different ‘recording efforts’. The estimates are based on field trials executed within the EBONE project across Europe.

Basic= recording of areal elements + qualifiers

Intermediate= recording of areal elements + qualifiers + line/point elements and/or Annex I habitats

Full= recording of areal elements + qualifiers + line/point elements and/or Annex I habitats + vegetation relevés

On average 10,3 person days were spent on a single kilometre square, with a minimum of 3,5 person days and a maximum of 25 person days per square kilometre. This includes preparation time, travel time, fieldwork and data input. Time for the training of surveyors is not included. As could be expected, there was a very large variation in time requirements between countries and between visited sites (3,5 – 25 days), reflecting the variety in conditions and landscape heterogeneity on the one hand, and the recording effort on the other. Recording effort differed among squares from **basic** (only areal elements and qualifiers) over **intermediate** (including line and point elements and/or Annex I habitats) to **full** habitat mapping and recording (areal elements plus qualifiers, plus, linear and point elements and/or

² Austria, Greece, Belgium, Spain, France, The Netherlands, Great-Britain, Slovakia, Norway

Annex I habitats, plus vegetation relevés). Table 7.1 and figure 7.1 and 7.2 give an overview of time requirements for different “recording efforts” and for the different parts of the protocol (preparation – travelling – fieldwork – data input).

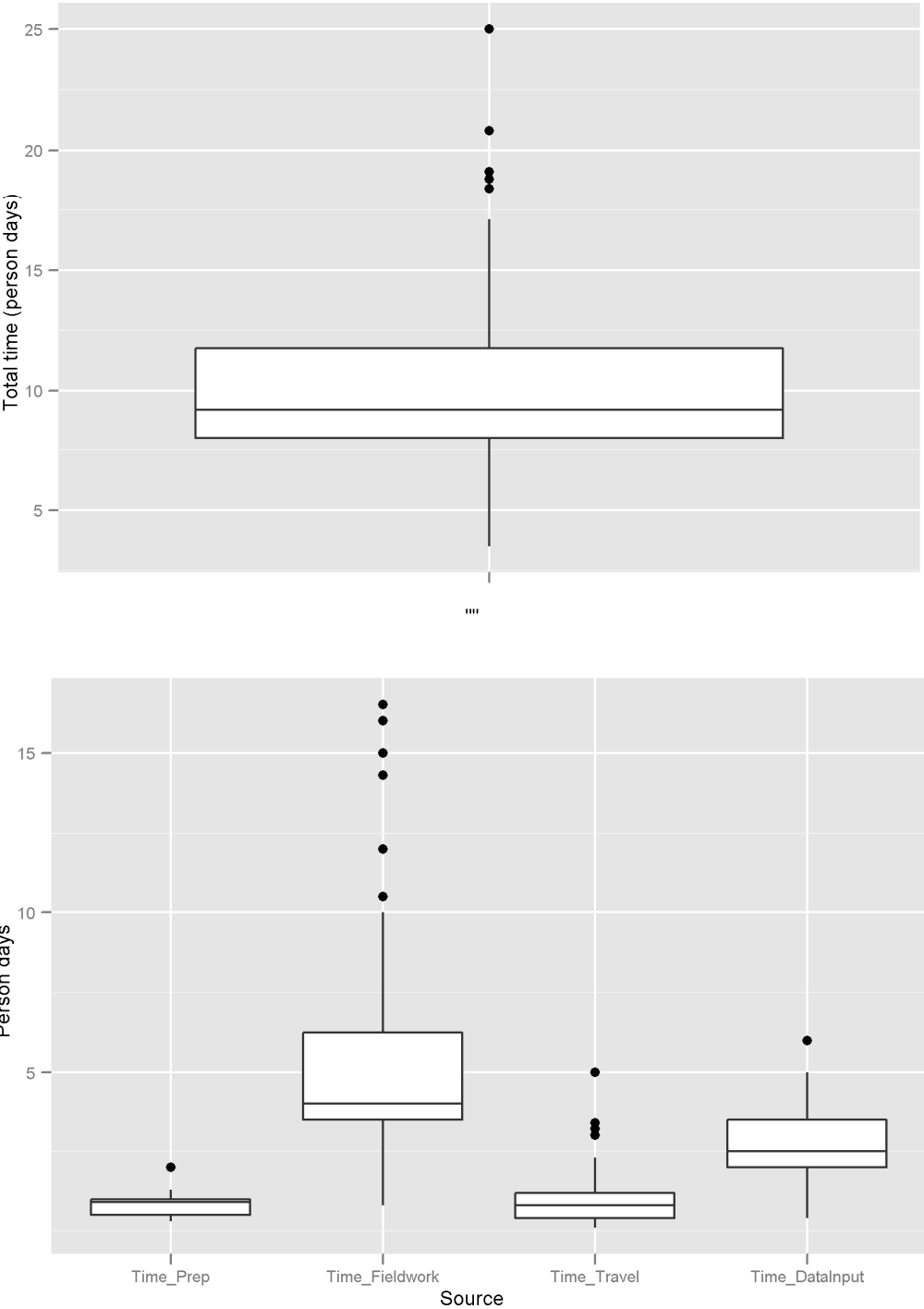


Figure 6.1: Variation in time requirements (in person days) for EBONE habitat mapping and recording. Total time use averages 10.3 person days (top) and total time partitioned into time for preparation, fieldwork, travel time and data input (bottom).;

Preparation time

Preparation time varied from a few hours to as much as two days. This includes field site selection and the preparation of field maps and ortho-photos. Preparatory work on delineation and interpretation of the major elements within the survey area based on aerial photographs, maps or satellite images is strongly recommended. A solid preparation beforehand can facilitate the fieldwork and thereby greatly enhance its efficiency (Bunce et al. 2011).

Travel to field sites

As one might expect, also travelling time varied greatly among countries and visited sites, mainly reflecting different country sizes and corresponding travel distances to get to the (random) selected field sites. Travel time varied from less than one hour to more than one day, with a maximum of 5 person days.

Fieldwork

The largest variation was found in the time requirements to do the actual mapping and recording. On average fieldwork required 5.6 person days per kilometre square, but this varied from approximately 1 day to as much as 16,5 person days, mainly reflecting the variety in conditions, habitats and landscape features among sites and the recording effort (basic, intermediate or full mapping and recording). Visited field sites seemed to vary from fairly homogeneous with only very little variation in General Habitat Categories to very complex landscape and vegetation patterns with many different General Habitat Categories. Complex landscape squares of course require more time to map and record. Mind that time is expressed in person days, as the EBONE procedure prescribes that the fieldwork is conducted by a field team that consists of at least 2 persons (preferably a botanist and an experienced mapper to ensure that the team is balanced), for safety and consultation reasons.

This large variation in time requirements also partly reflects the different recording efforts (basic, intermediate or full recording), as indicated in table 71 and figure 7.2.

Weather conditions and accessibility on the other hand, do have an impact on the time required to map and record a kilometre square, but could not explain the large differences in time requirements between visited sites (Figure 7.3).

Data input

Data input required on average 2,7 days (minimum 0,4 to maximum 6 days), with input time generally increasing with the amount of collected data (basic vs. intermediate and full recording). It was experienced however that the application of field computers was able to greatly reduce input time of data collected in the field. Of course, this requires more resources in terms of software and hardware. Several systems were tested in the EBONE project and were considered to be very useful. The use of field computers is also default in other major national biodiversity and landscape monitoring programs (e.g. NILS, Countryside Survey). Further development in technology is expected to even more facilitate habitat mapping and data input in the field.

Training of surveyors

All EBONE fieldwork participants participated in at least one week of specific training. It was recognized however that this was not enough to get completely familiar with all different aspects of the recording protocol. Based on experience in the BioHab project (Jongman et al. 2005), it is expected that at least 2,5 weeks of training are needed to become experienced with the complete procedure of mapping and recording. For basic habitat mapping and recording (elements and qualifiers) one to two weeks of training will probably be sufficient. In the Swedish NILS programme, two weeks of training and one week of fieldwork with guidance of an experienced fieldworker, (so in total three weeks of training) is the standard (Gonzalez 2005).

Overall efficiency

Despite the great variation in time requirements, the average time requirements of the EBONE procedure seems to correspond very well to time requirements of other large-scale national monitoring programs. For instance, in the Countryside Survey (UK), 12,3 days are spent to map and record one kilometre square on average (personal comm. Bob Bunce), whereas the complete EBONE procedure of habitat mapping and recording is, with 11,1 person days per km² on average, slightly less time consuming. Basic and intermediate application of the EBONE procedure seems to be considerably quicker (on average 8-10 person days per km²).

Time requirements for the fieldwork in the National Inventory of Landscapes in Sweden (NILS) however appears to be slightly less time consuming than EBONE (e.g. Gonzalez 2005). Aerial photo interpretation (inventory by infrared false colour aerial photos) takes on average 3,5-4 person days per kilometre square and fieldwork (plots and line segments, including travelling time) takes on average 3 days per km² and per field team (2 persons). So, approximately 10 person days are spent per km². Data control after the fieldwork however is not included in this figure and can take quite some extra time. This makes that the total time spent per square is not exactly known (pers. comm. A. Allard, SLU).

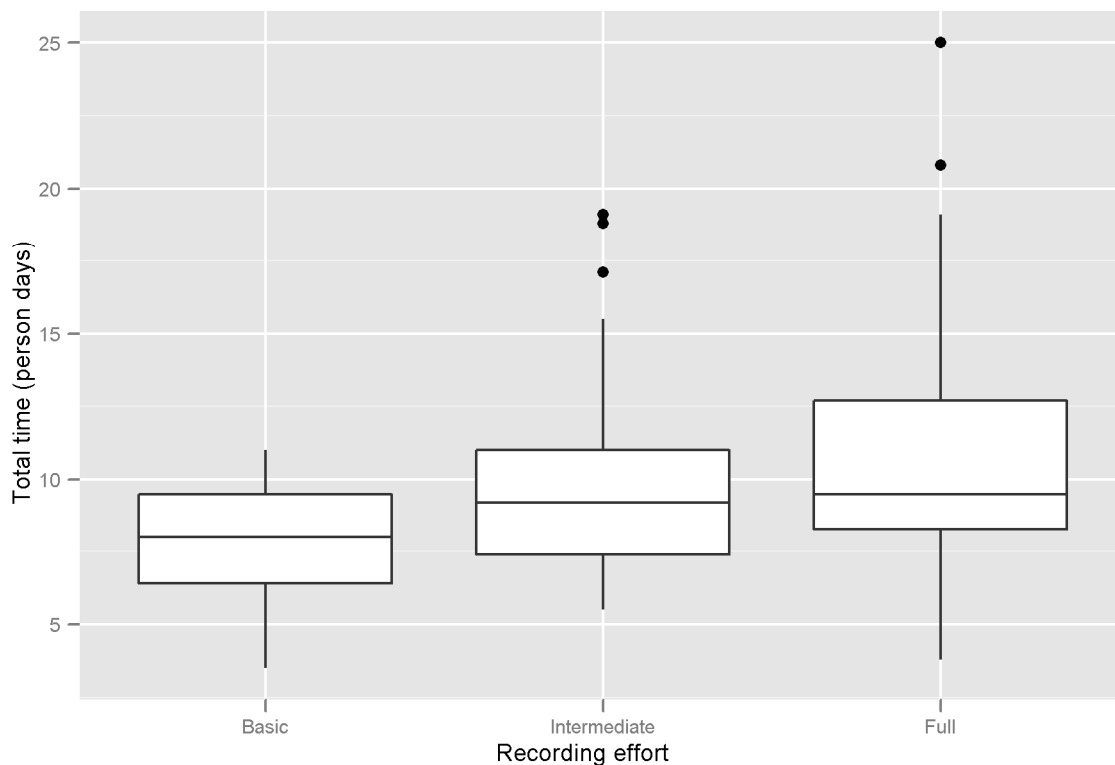


Figure 6.2: Time requirements (in person days) for conducting basic, intermediate and full EBONE habitat mapping and recording. Time use partly depends on the recording effort:

Basic= areal elements + qualifiers

Intermediate= areal elements + qualifiers + line/point elements and/or Annex I habitats

Full= areal elements + qualifiers + line/point elements and/or Annex I habitats + vegetation relevés

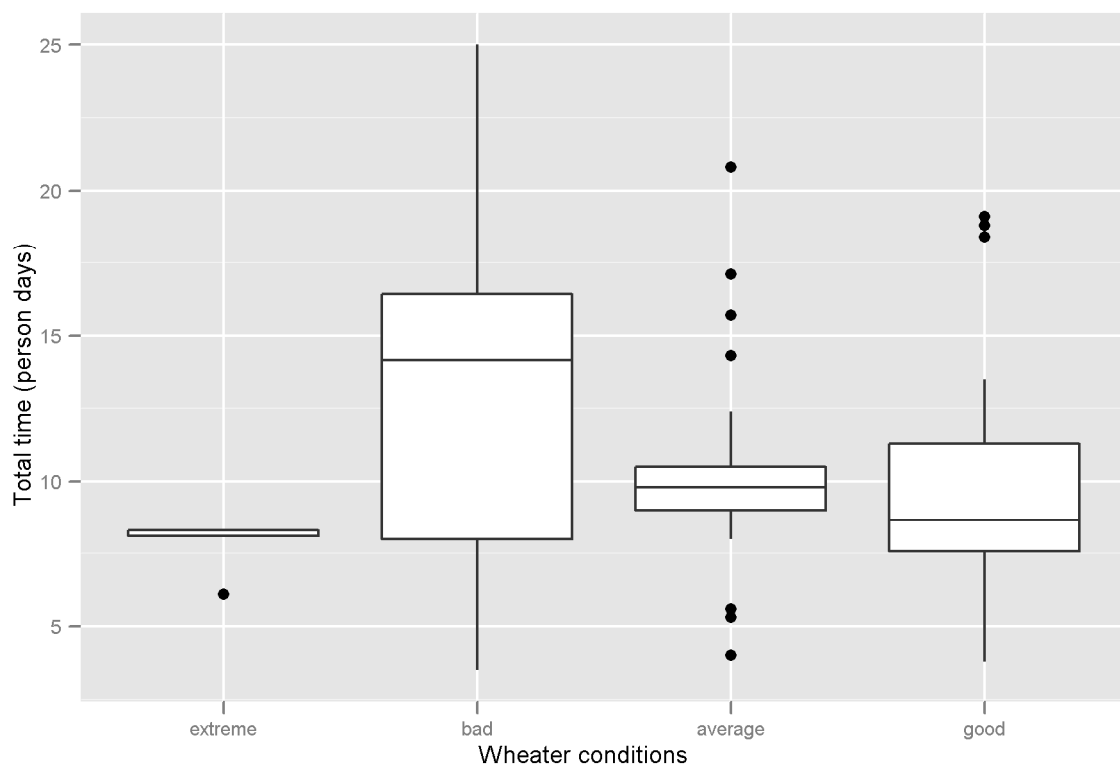
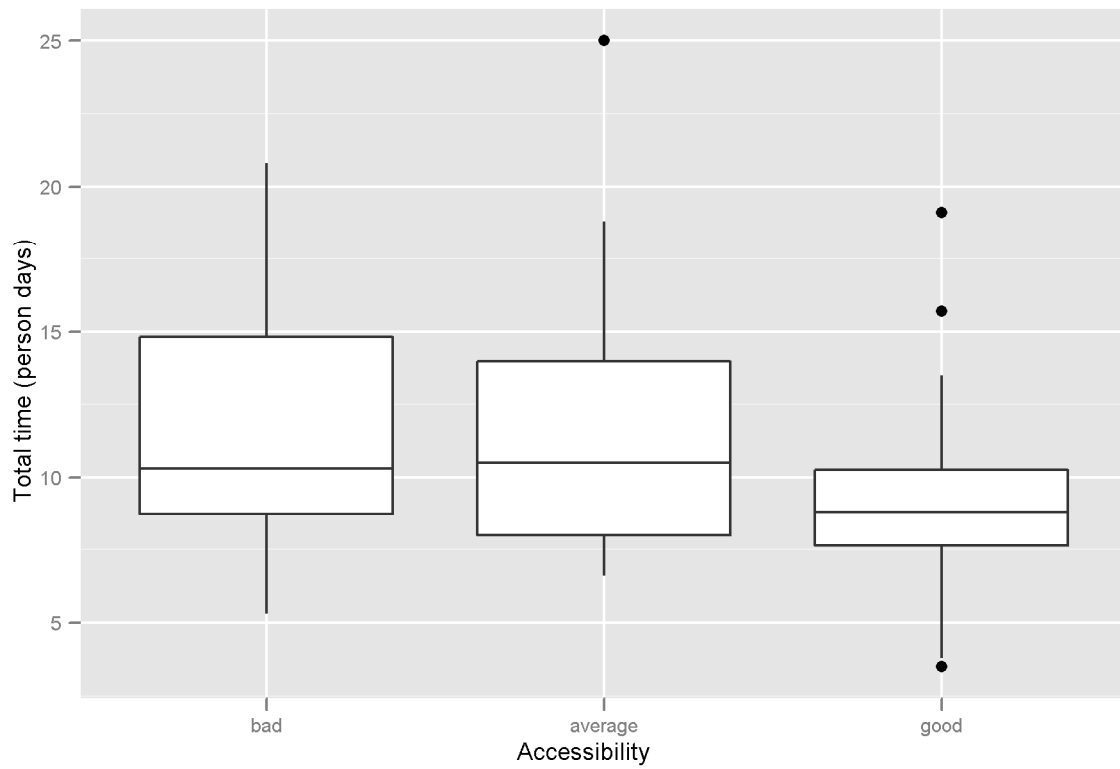


Figure 6.3: Variation in time requirements (in person days) for EBONE habitat mapping and recording with different weather conditions (bottom) and differences in accessibility of the field sites (top).

6.2. Data management requirements related to effectiveness of a monitoring programme

Data management was not taken into account in the previous analysis of time requirements. It is however a crucial factor in long-term monitoring programs, however its importance is often underestimated or neglected. Many monitoring programs have failed to employ adequate data management procedures, such as appropriate standards for data documentation, and few have provided adequate financial support (Caughlan and Oakley 2001). E.g. a general 'rule of thumb' for data management costs in the US Monitoring Long-Term Ecological Research (LTER) program is about 15% of total project costs (Lindenmayer & Likens 2010). Other recommendations for data management in long-term research and monitoring programs refer even to 20% of total project costs (Stohlgren *et al.* 1995; Jensen and Bourgeron 2001; Lindenmayer & Franklin 2002, Lindenmayer & Lickens 2010). So, adequate management of long-term environmental data sets is very challenging and requires substantial technical expertise and significant financial support (Michener and Brunt 2000). Promoters, managers and scientists have often failed to recognize the technical challenges, costs, and critical role of data management, preferring to utilize limited financial resources for other priorities in the monitoring project, such as extensive fieldwork campaigns. The reality is that the data handling and storage in reliable databases, is often an after-thought in the vast majority of complex, multi-sources and long-term monitoring programmes. As a consequence, monitoring efforts may become un-efficient, as a considerable amount of collected data remains unused. Other records can even be effectively lost due to inadequate documentation and quality control. So, ample time and resources should be allocated from the very first beginning of the development of a monitoring programme, to the elaboration and testing of an adequate and robust data management system. If not, the chance to end up with a disappointing and unbalanced monitoring strategy is considerable and partners may tend to abort the project before it is completed. The experiences of the EU BioSoil demonstration project are illustrative in this respect. Proper data validation, robust data storage and even preliminary data elaboration and evaluation were not extensively covered by the original project. It was estimated that these processes need at least another 3 years by experienced staff (Bruno De Vos, personal comm. 2012; see §3.1, Box 3). One of the lessons learned in BioSoil focus on the response of participating countries in the monitoring project: "Many countries encountered problems by using the BioSoil online data submission module. Difficulties when entering data and/or extensive 'error reports' are not particularly stimulating for Member States when entering large data series. Submission modules should be robust and user friendly and thoroughly tested and instructed prior to initialization. It appeared that large and valuable data were missing in the final database because a country was unable to finish the whole input procedure. This might lead to frustration and should be avoided" (see §3.1, Box 3).

Position of EBONE

In EBONE, these pitfalls were recognized and anticipated by organizing dedicated workshops and participation in training and demonstration before the field campaign. Extensive technical specifications for data handling are provided in a specific report (Peterseil *et al.*, 2010), and in the data management system, a data entry form with masks and predefined lists is incorporated, while integrity and consistency check routines are implemented in the data entry form database for the data provider after entering the data.

Nevertheless, also in EBONE, achieving a consistent data flow remained a burden, especially at the end of the project when analyzing the complete data set. Data integration for instance was hindered by interpretation errors in the mapping method and transcription errors from recording sheet to database. Problems were encountered with shape file structure and labeling, shape files and related databases could be inconsistent, etc.

Position of EBONE

Starting from its own experiences, EBONE suggests the following improvements to avoid the problems related to consistent data flow and handling which are regularly encountered in monitoring projects that involve many partners:

- Establish a single point of access, a portal, for data providers and data users
- Pay considerable attention to Improvements of check routines that are related to content and structure, on the level of the data generation and file upload with quality check done by data provider
- Organize targeted and intensive trainings both for the mapping methods applied and the data entry procedures before the start of data collection through field campaigns; repeat this on a regular base
- Ensure and facilitate a tight communication between data provider and data manager.

7. Cost-effective sampling design for biodiversity monitoring in Europe

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7.1. Precision and power calculations for the EBONE sampling design

7.1.1. EBONE monitoring objectives

In order to do power calculations or to calculate the precision of parameters we need to be explicit about the most important aspect to monitor (the monitoring objective). For the EBONE project, a major goal is to be able to make stock and change estimates for the general habitat categories, using the BIOHAB field protocols (in situ sampling units; Bunce et al. 2010) and the EBONE sampling design (Brus et al. 2011).

In addition, we want to explore if and how earth observation can be used in conjunction with in situ samples and specifically what the implications are regarding precision, bias and power for stock estimates (see 7.1.5).

There are two spatial scales at which estimates for stock and change are particularly relevant. First, at the European scale. Second, at the scale of environmental zones within Europe. For the EBONE sampling design, the sample size is allocated proportionately to the area of environmental zones.

7.1.2. EBONE sampling design

The proposed EBONE sampling scheme is inspired by the UK Countryside Survey. The sampling design is discussed in detail in Brus et al. (2011). In short, the sample is random, but spatially balanced. Each sampling unit is a 1 km x 1 km square. Field sampling is done according to BIOHAB field protocol, hence, the result is a 1 km x 1 km map of the spatial distribution of General Habitat Categories within each square. Sampling units are permanent and will be revisited once in every monitoring cycle. For the EBONE sampling design, it was decided that a sample size of 10000 1 km x 1 km squares was the maximum sample size achievable. It represents approximately 0.25 % of the part of the European surface area that belongs to the sample frame (for the exact definition of the sampling frame: see Brus et al. 2011). This population fraction is roughly comparable to that of the UK Countryside Survey.

More specifically, the EBONE sampling design has the following properties:

- **Sampling unit:** 1 km x 1 km square
- **Sampling methodology:** mapping of General Habitat Categories within 1 km x 1 km squares
- **Sample size:** $n = 10000$
- **Area of the sample frame:** 4027782 km²
- **Sample selection:** spatially balanced random sample (i.e. assuring that sampling units are distributed evenly across the sample frame but in a random fashion, not in systematic fashion). The sample is first stratified by Environmental strata (EnS; about 80 strata) and sample size within each EnS (n_{EnS}) proportional to the area of the EnS. Spatial balance is ensured by subdividing each EnS in so-called geostrata by means of a clustering algorithm that results in geostrata that have approximately the same area and have an edge-to-area ratio as high as possible. Within each geostratum five 1 km x 1 km squares are selected at

random (see space-time design) (the number of geostrata is $n_{EnS}/5$. EnS can be aggregated to Environmental zones (EnZ). The number of samples within an EnZ (n_{EnZ}) is proportional to the area of the EnZ (see Table). EnZs are an important spatial scale for reporting on stock and change estimates. This is because EnZs largely correspond with the biogeographical zones defined in the Habitat Directive (92/43/EEC) (Jongman et al. 2006).

- **Space-time design:** The space-time design is serially alternating with a cycle of five years and yearly sampling. Sampling units are permanent and will be revisited once in each cycle. Because each year one out of five random samples within each geostratum is chosen randomly without replacement, each year-panel (the samples collected in a given year) has the desirable property of being a spatially balanced sample.

Table shows the number of sampling units in each environmental zone as well as the area of each environmental zone. The average environmental zone has 859 1 km x 1 km sample squares.

Environmental Zone	Sample Size (n_{EnZ})	Area (km ²)
Alpine North (ALN)	580	231351
Boreal (BOR)	1075	435626
Nemoral (NEM)	400	161850
Atlantic North (ATN)	395	159188
Alpine South (ALS)	610	245390
Continental (CON)	2125	854814
Atlantic Central (ATC)	1045	420500
Pannonian (PAN)	730	294792
Lusitanian (LUS)	495	198660
Mediterranean Mountains (MDM)	910	244159
Mediterranean North (MDN)	1045	423632
Mediterranean South (MDS)	895	357820
First quartile	516	206833
Median	813	270091
Mean	859	335649
Third quartile	1045	422849
Standard deviation	471	191676
Interquartile range	529	216016

Table 7.1: Sample size and area per environmental zone, along with some summary statistics.

Since the sampling scheme compares well with the UK Countryside Survey, it is worthwhile to look at the data they have collected in the past to learn about the empirical estimates for total area of broad habitat types and their associated precision. This is what we will cover in the next section (section 7.1.3). For two reasons this is an important first step before the actual precision and power calculations for the EBONE sampling design will be done. First, it will allow us to get an insight in plausible ranges of values (area within a km-square, proportion of sampling units where the habitat is present, ...). Second it will provide a benchmark against which our procedures to calculate precision and power can be checked.

7.1.3. Empirical precision of stock and change estimates based on UK Countryside Survey

We used data from Howard et al. (2003). Howard et al. (2003) give estimates for stock and change based on data gathered in the 1990 and 1998 UK countryside survey. We focus first on the stock estimates for 1998. These estimates are based on a stratified random sample (strata are ITE land use classes) from 576 1 km x 1 km squares. This is about 0.26% of the total land surface. The total surface area of the sample frame (UK) equals 224280 km².

We first look at the actual field data and precision achieved on area estimates (stock estimates). Next, we simulate the precision for the 1998 CS and check if our results match the empirical estimates.

The empirical precision of stock estimates for the UK Countryside Survey is depicted in Figure . The upper panel depicts the relative precision, the lower panel the absolute precision.

Figure suggests that the relative margin of error is smaller for broad habitat types that have a larger total surface area and/or have a more even distribution (hence, a higher percentage of sampling units in which the habitat was found). Indeed, it is to be expected that a less common habitat type (say one that covers less than 10% of the UK) with a uniform spatial distribution can be estimated more precisely than one with a clustered spatial distribution (hence, absent from most sampling units).

According to these estimates, the broad habitat types cover between 0.2% (Montane habitats) and 24% (Improved grasslands) of the UK land surface. The percentage of 1 km x 1 km squares where the habitat was present ranges from 3% (Montane habitats) to 79% (Broadleaved, mixed and yew woodland).

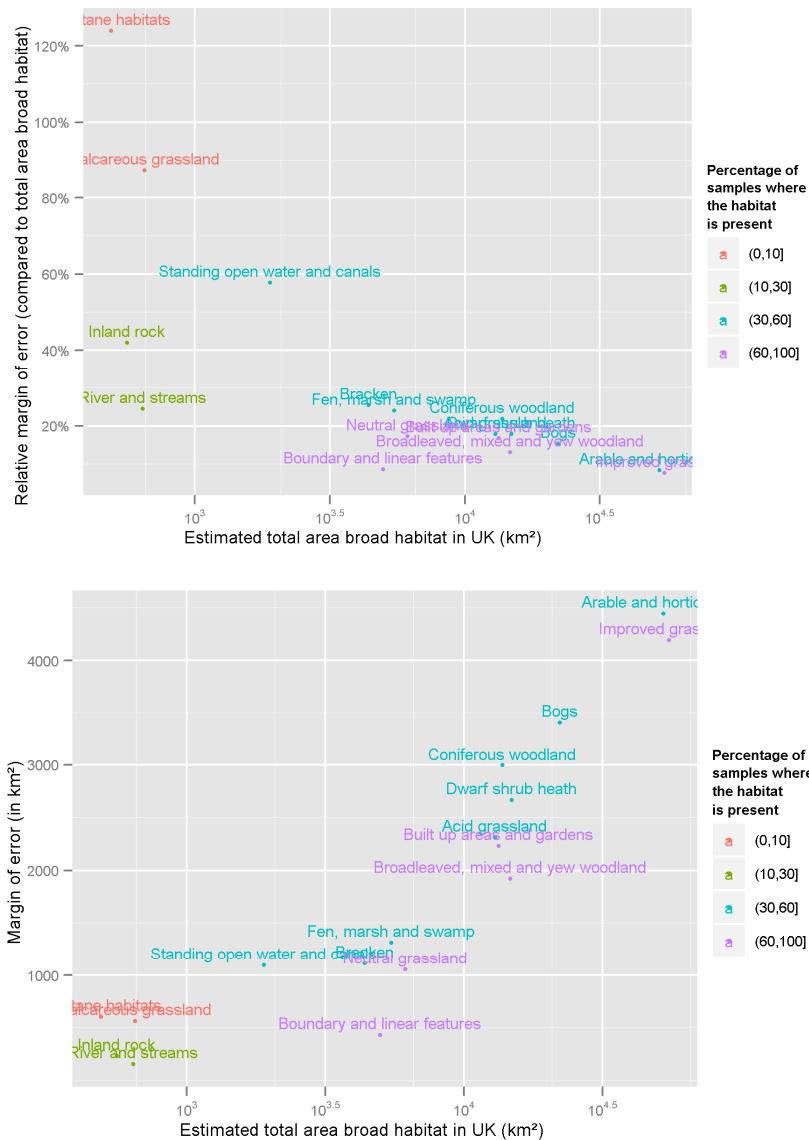


Figure 7.1: Empirical precision as calculated from data obtained in the 1998 Countryside Survey for the broad habitat types. Each point represents one broad habitat type. Precision is expressed as the half width of the 95% confidence interval (= margin of error). Upper: relative values for precision; Lower: absolute values for precision.

Given these observations based on the Countryside Survey sampling scheme, we developed a procedure to realistically estimate the stock precision.

For this procedure, we ignore the stratification and calculate the standard error as if the data come from a simple random sample. The estimator for the mean (total) for a simple random sample is less precise/accurate given the sample size than that for a stratified random sample (e.g. Cochran 1977). Thus, our precision estimates are an underestimate compared to the ones that would result if stratification was accounted for. Metzger et al. (2012) analysed if the stratification into Environmental strata (EnS) was beneficial by comparing the sampling variance of the estimated total for Corine Land Cover classes with and without stratification. Their results indicate that the stratification resulted on average in 14% more precise estimates at EU level. At the level of Environment zones (EnZ) the gain in precision was marginal (on average less than 1%). To account for the underestimation we could have included in our procedure a design effect based on the prior estimates obtained by Metzger et al. (2012), but this was not included in the procedure. The procedure was as follows:

- Determine plausible range of values for total area of a broad habitat type. The largest broad habitat type in the UK was improved grassland, which covered about 25% of the UK. Montane habitats were least extensive with a cover of only 0.2% of the UK land area (A_{UK}).
- Determine plausible range of values for the percentage of km squares (sampling units) where a habitat is present. Broadleaved, mixed and yew woodland was present in 79% of the km squares. Only 3% of km squares had Montane habitat.
- Determine plausible combinations of total habitat area, A_{hab} , and proportion of km squares with that habitat, f_{hab} . The following rule was used, which ensured that the average area of a habitat within the 1 km x 1 km squares where the habitat is present, μ_{hab} , cannot be greater than 0.5 km² (or 50% of the square):
 - $\mu_{hab} = \frac{A_{hab}/A_{UK}}{f_{hab}} \leq 0.5$
 - for the UK, the habitat type with the largest average area in km squares where it was present, was equal to 0.41 km² for arable and horticultural land
- Every one of these plausible combinations of habitat area and proportion of km squares where the habitat is present, were used as parameters in a zero-inflated beta distribution. The complement of the proportion of km squares where the habitat is present ($1 - f_{hab}$) gives the amount of zero-inflation (i.e. the probability of not encountering the focal habitat in a km-square). The average area of habitat in occupied km-squares (μ_{hab}) determined the mean of the beta part of the zero-inflated beta distribution (cf. Ospina & Ferrari 2010). A third parameter, a , allows to set the amount of variance given a mean. We set the parameter a to 1 (see Bolker 2008 pp 133-135; see Figure 7.2.; Box I for the effect of tuning the variance given a mean: parameter $a = 1$ and mean = 0.5 corresponds with a uniform distribution). This zero-inflated beta distribution can be interpreted as follows:
 - First, it tells us the chance of finding a focal habitat in a sample of km squares. This chance is equal to the proportion of km squares where the habitat type is present.
 - If the habitat is present, the zero-inflated beta distribution tells us that the area of focal habitat in the 1 km x 1 km square can have any value larger than 0 and smaller than 1 km², but that on average the habitat covers μ_{hab} .
- Next, we used formulas from Brus et al. (2011) to calculate precision of stock estimates. The formula to calculate the margin of error only needs the following data:
 - The significance level (α set to 0.05)
 - The sample size
 - The population size, which is the total number of 1 km x 1 km squares in the sampling frame (i.e. A_{UK} in km²)
 - The estimated spatial variance $S^2(y)$ (= population variance) for the zero-inflated beta distribution. The latter can be derived analytically (Ospina & Ferrari 2010):
 - $S^2(y) = f_{hab} \frac{\mu_{hab}(1-\mu_{hab})}{\varphi+1} + f_{hab}(1-f_{hab})\mu_{hab}^2$
- The formula to estimate the margin of error (ME) for a stock estimate – if we approximate the sampling design as a simple random sample – than equals (Brus et al. 2011):
 - $ME = \frac{N\sqrt{S^2(y)u_{1-\alpha/2}}}{\sqrt{n}}$, where $u_{1-\alpha/2}$ is the $(1 - \alpha/2)$ quantile of the standard normal distribution

Box 7: The beta distribution and zero-inflated beta distribution

The beta distribution is a continuous analogue for the binomial distribution. It is especially useful when proportions need to be drawn at random from a continuous distribution, because it is bounded between 0 and 1. The beta distribution depends on two shape parameters a and b , that determine the mean, variance and shape of the distribution. Parameter $a - 1$ represents the number of successes and $b - 1$ the number of failures. This interpretation only holds for a and b larger than 1. The mean equals $a/(a + b)$. Figure 7.2: gives the effect of parameter a on the variance of the beta distribution for a given mean. The mean value should be understood as the average amount of a habitat type in the 1 km x 1 km squares where that habitat occurs, but we ignore the zero-inflation part for the moment). Based on data in Howard et al. (2003) this average is unlikely to exceed 0.5 km² (the largest average was for arable and horticultural land and equalled 0.4 km²).

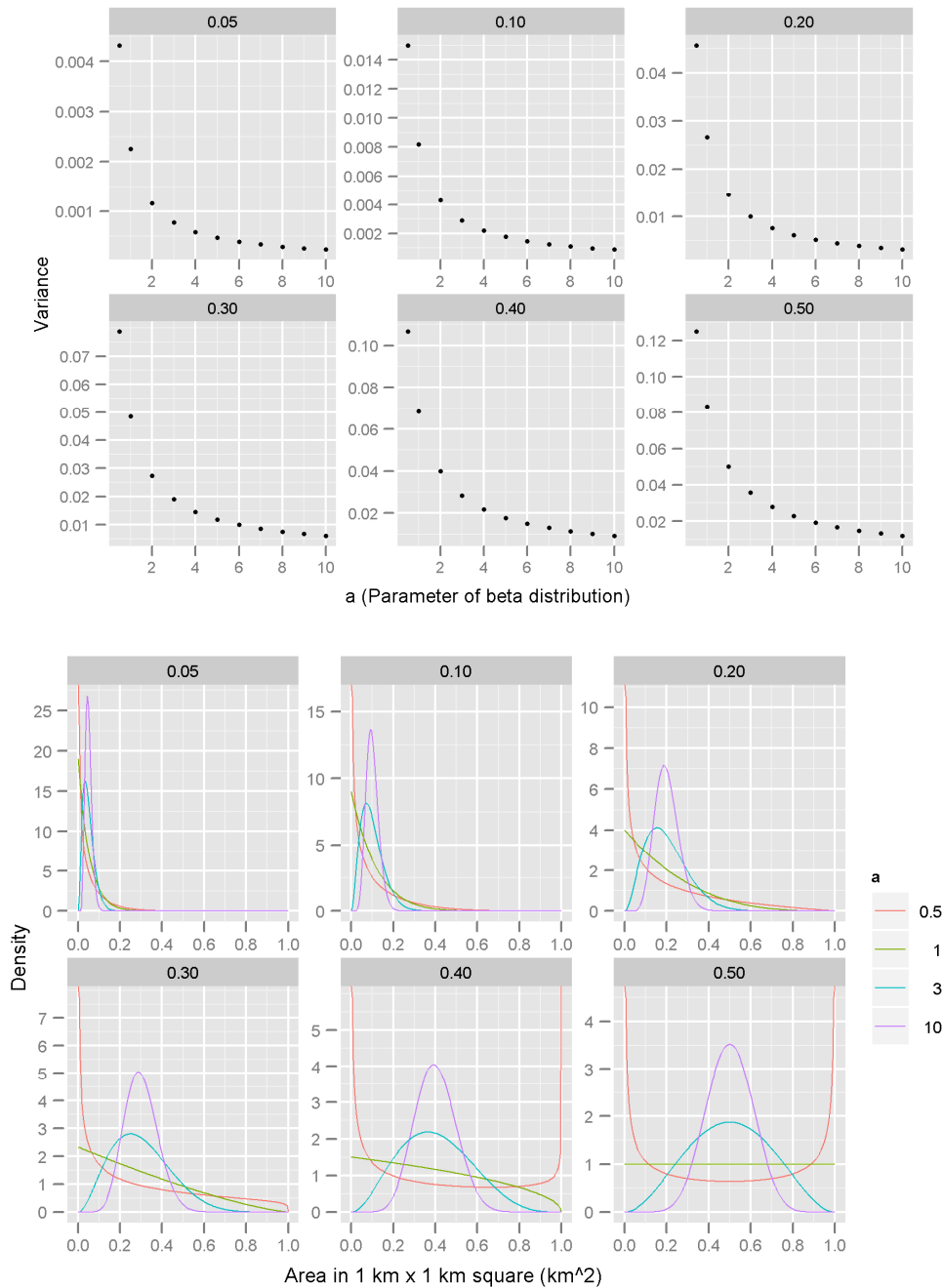


Figure 7.2: Influence of the shape parameter a on the variance of the beta distribution for a given mean (ranging from 0.05 to 0.50). Top: variance as a function of the mean and parameter a ; these points appear on a straight line when the axes are scaled on a logarithmic base. Bottom: corresponding density distribution plots for $a = 0.5, 1, 3$ or 10 .

Ospina & Ferrari (2010) discuss the properties of inflated beta distributions. They use a different parameterization for the beta part of this mixed distribution, namely based on $\mu (= a/(a + b))$ and $\Phi (= a + b)$. The mean for the zero-inflated beta distribution equals $f\mu$, with f the proportion of zero elements (= the amount of zero-inflation = the proportion of km-squares where the habitat is absent). Figure 7.3 shows that the effect of zero-inflation on the variance depends on μ . Figure 7.4: gives some examples of density plots for zero-inflated beta distributions.

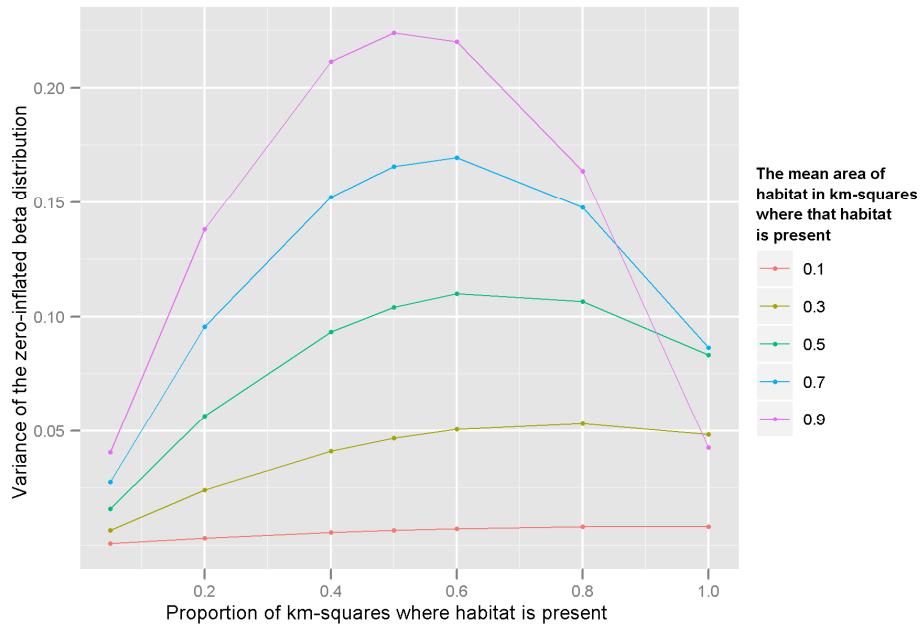


Figure 7.3: The influence of zero-inflation (i.e. the complement of the x-axis) on the variance of the zero-inflated beta distribution depends on the mean for the beta part of the distribution.

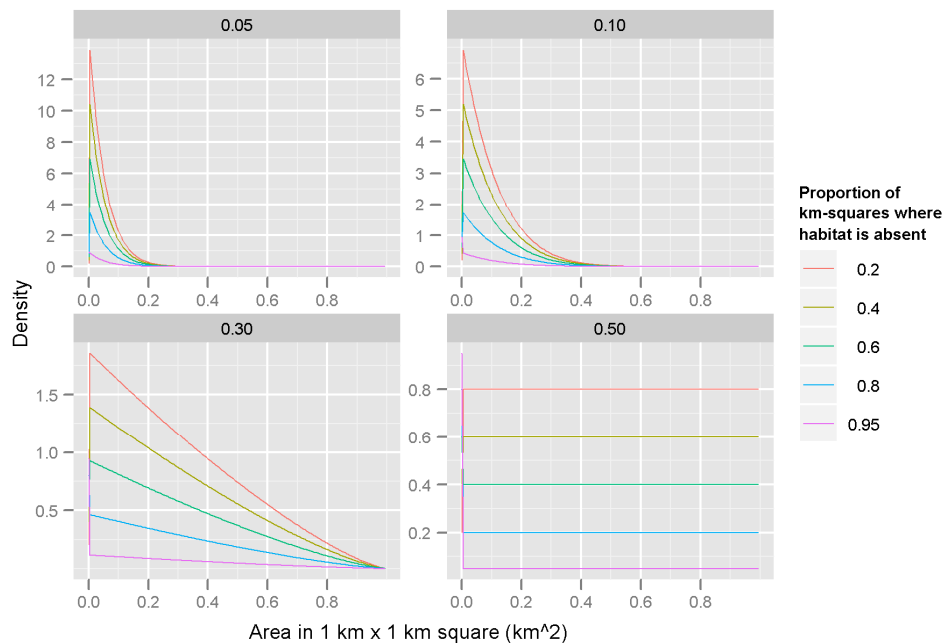


Figure 7.4: Density plots for zero-inflated beta distributions. Colours indicate the amount of zero-inflation: the proportion of km-squares where the focal habitat is absent. The title heading on each panel gives the mean value for the beta-part of the distribution (thus the mean excluding zeros). The scale parameter a was set to 1.

The results of the procedure are given in Figure 7.5.: Comparison of Figure and Figure 7.5.; we learn that the precision estimates based on the procedure are within the range of the empirical values. It is not a perfect match, and we could have fine-tuned the procedure more by adjusting the parameter of the beta distribution that determines the amount of variability between 1 km x 1 km sample squares where the habitat is present (see Box I Figure 7.2.:). However, as this would only complicate things, since the variability will be habitat specific, we decided not to do this. However, we should be aware that the results could give a more optimistic picture than the reality will turn out to be, because the variability between km squares may be higher. Only European wide data collected in an implementation or pilot phase of the monitoring schedule could provide certainty about the robustness of the results.

In the next section (7.1.4), we will use this procedure to assess statistical performance of the EBONE sampling design.

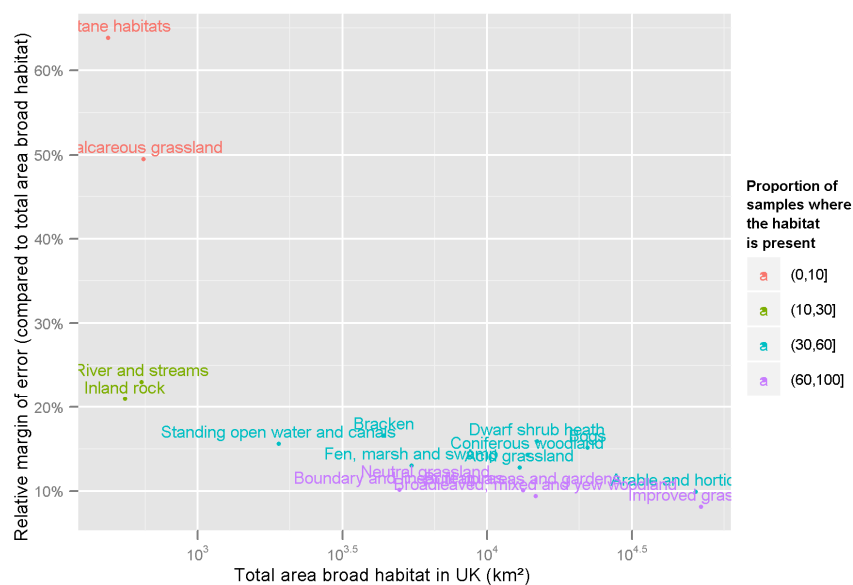


Figure 7.5: The relative margin of error as estimated with design-based formula and spatial variance derived from zero-inflated beta distributions. Compare with Figure .

7.1.4. Precision of stock and change for the EBONE sampling design

7.1.4.1. Precision of stock estimates

We based our simulations on sample sizes of 625, 2500 and 10000 km squares. Based on the data from the UK CS, we also let the proportion of km squares where habitat was present vary between 0.05 and 0.8. We did not explore different scenarios for the effect of differences in variability (given a mean area) in area between km squares where habitat was present (i.e. the a parameter for the beta distribution was fixed at 1).

We consulted Evans (2006) to have a rough indication of total area estimates in GHC classes or super-GHC classes. Evans (2006) gives total area estimates for the group and sub-group levels of Annex I habitat types in the Natura 2000 network. At the sub-group level (for instance code 91 for forests of temperate Europe), these estimates range from 500 km² to 37 000 km² (resp. 0.06% to 2% of Natura2000 network, and 0.01% to 0.36% of Europe). However, the area outside the Natura2000 network is unknown.

The latest data of reported areas of Annex I habitat types were also consulted (data provided by Doug Evans). Figure gives a histogram of reported areas of Annex I habitat types. The distribution of the data is highly skewed to the left. The median habitat type covers 647 km² while the mean area equals 3636 km². It should be noted however that the reported areas are probably an underestimation of the actual areas present (pers. comm. Doug Evans). Figure 7.1 shows the same data in boxplot form for each habitat group. Again, within each habitat group, there are few habitat types with relatively large reported total area. Furthermore, the habitat types that cover a large total area tend to be the ones which occur more widespread across Europe. This is indicated in Figure 7.8.: the total area distribution shifts upward with increasing number of biogeographical regions in which habitat types are known to occur. This is especially evident for habitat groups 2xxx, 3xxx, 6xxx and 9xxx. Aggregating the Annex I habitat types into subgroups (Figure 7.9.) and groups (Table 7.2.) gives us an indication of the range in total area at these levels. At habitat subgroup level (for instance 91xx), the overall median and mean total area is 15204 km² and 23253 km², respectively. At habitat group level, median and mean total area is 75176 km² and 85236 km², respectively.

Given these observations, we took as lower limit 1000 km². This is only 0.025% of the European area frame. However, we were also interested in precision estimates relevant at the scale of a biogeographical zone. Therefore, this low limit is relevant, but we have to remember that biogeographical zones have a smaller sample size (the average biogeographical zone will have roughly 860 km squares, see Table). Furthermore, as we also want to explore whether the EBONE design can yield precise estimates for Annex I habitat types, 1000 km² is a relevant lower limit. As upper limit, we simply take the maximum observed relative total area of broad habitat types in the UK (24%), which, for the European case, represents about 1 000 000 km² (\approx 25% cover of a habitat type in Europe). This is still about four times higher than the maximum reported total area aggregated at habitat group level (habitat group 9, forests, 272487 km²).

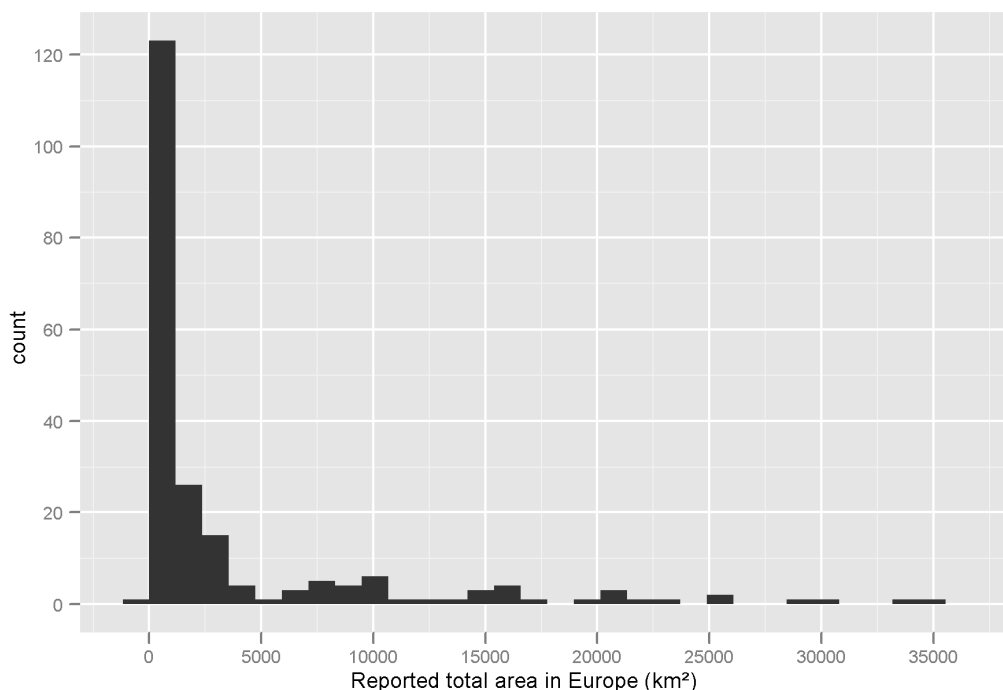


Figure 7.6: A histogram of reported total area of Annex I habitat types in Europe.

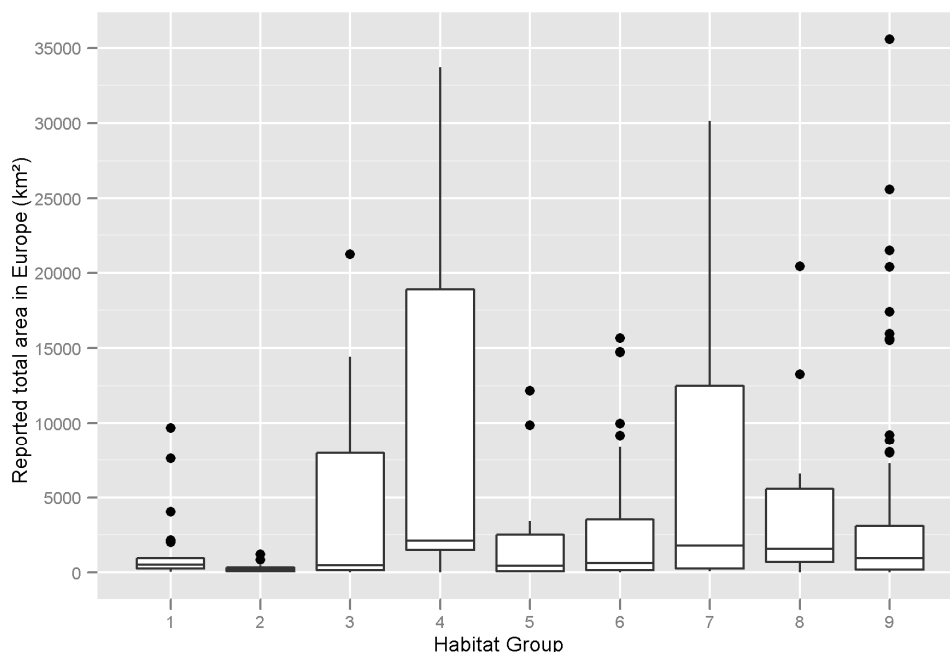


Figure 7.1: Boxplots showing the distribution of reported total area of Annex I habitattypes (for instance habitattype 9120) in Europe per habitat group. Habitat groups: 1 coastal and halophytic habitats, 2 coastal sand dunes and inland dunes, 3 freshwater habitats, 4 temperate heath and scrub, 5 sclerophyllous scrub (matorral), 6 natural and semi-natural grassland formations, 7 raised bogs and mires and fens, 8 rocky habitats and caves, and 9 forests.

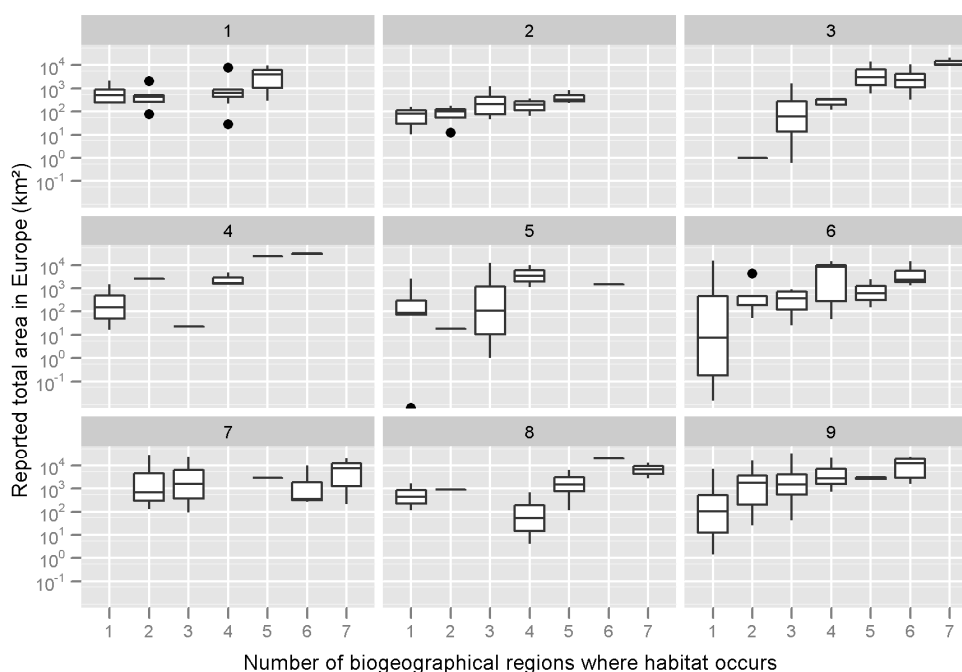


Figure 7.8: Boxplots showing the distribution of reported total area of Annex I habitattypes (for instance habitattype 9120) in Europe per habitat group (panels) and as a function of the number of biogeographical regions where they are known to occur. The Y-axis is scaled logarithmically. Habitat groups: 1 coastal and halophytic habitats, 2 coastal sand dunes and inland dunes, 3 freshwater habitats, 4 temperate heath and scrub, 5 sclerophyllous scrub (matorral), 6 natural and semi-natural grassland formations, 7 raised bogs and mires and fens, 8 rocky habitats and caves, and 9 forests.

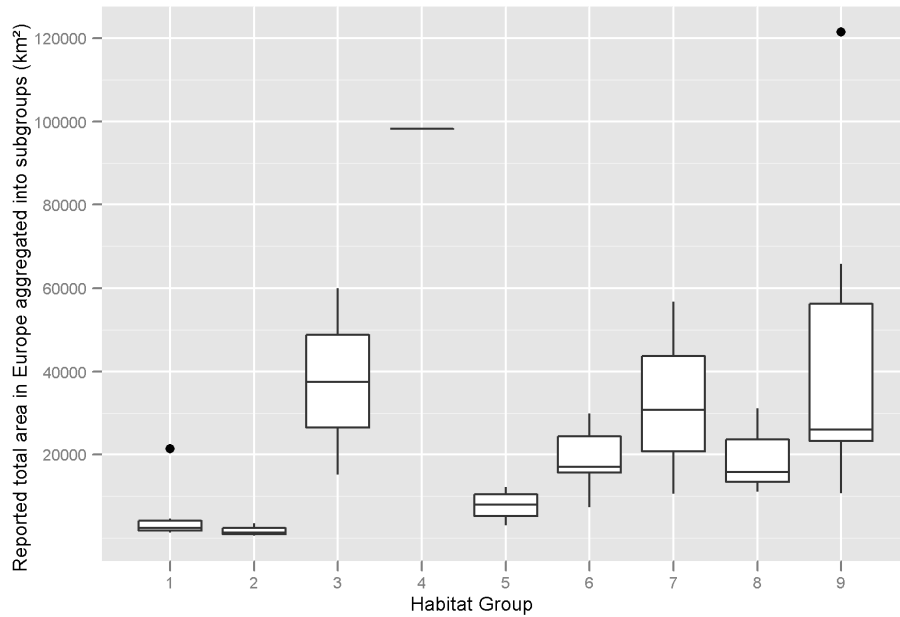


Figure 7.9: Boxplots showing the distribution of reported total area of Annex I habitat subgroups (for instance habitat subgroup 91) in Europe per habitat group. Habitat groups: 1 coastal and halophytic habitats, 2 coastal sand dunes and inland dunes, 3 freshwater habitats, 4 temperate heath and scrub, 5 sclerophyllous scrub (matorral), 6 natural and semi-natural grassland formations, 7 raised bogs and mires and fens, 8 rocky habitats and caves, and 9 forests.

Group	Total area (km ²)
1	33778
2	5395
3	75176
4	98265
5	31260
6	94543
7	98235
8	58226
9	272487

Table 7.2: Reported total area of Annex I habitat groups in Europe. Habitat groups: 1 coastal and halophytic habitats, 2 coastal sand dunes and inland dunes, 3 freshwater habitats, 4 temperate heath and scrub, 5 sclerophyllous scrub (matorral), 6 natural and semi-natural grassland formations, 7 raised bogs and mires and fens, 8 rocky habitats and caves, and 9 forests.

We estimated the precision of stock estimates using the same procedure as in 7.1.3. We used the ranges for proportion of habitat present and total area mentioned before. The results are given in Figure 7.1 and Figure 7.2. The (relative) margin of error is halved with every fourfold increase of the sample size. For the EBONE sample design, with 10 000 km squares spread across the European sample frame, the relative margin of error is between 2% and 12% for stock estimates. The margin of error is very sensitive to the proportion of km-squares where the habitat is present in the sense that low proportions strongly inflate the margin of error.

The question boils down to what level of numerical precision is agreed to as acceptable. This can be formulated in quality objectives. If we accept that the numerical quality of our stock estimates for broad habitat types is within $\pm 10\%$ relative margin of error, than, in most cases, the EBONE sampling design will do. This means that we can say with 95% confidence that the true area, A , of a focal broad habitat

is within the interval $[A - 0.1A, A + 0.1A]$. Thus, a habitat with an observed area of 1000 km² has a 95% confidence interval of 900 km² - 1100 km². If, however, the proportion of samples where the habitat is present becomes too low (say ≤ 0.05), we will have to accept a less stringent quality objective.

Furthermore, restricting the geographical scope (for instance singling out one or more biogeographical regions) reduces the sample size and therefore reduces precision (the relative margin of error increases) for stock estimates. To achieve a quality objective of $\pm 10\%$ relative margin of error would require the proportion of km squares where habitat is present to be larger than 0.4 in the case of 2500 km squares and larger than 0.8 in the case of 625 km squares.

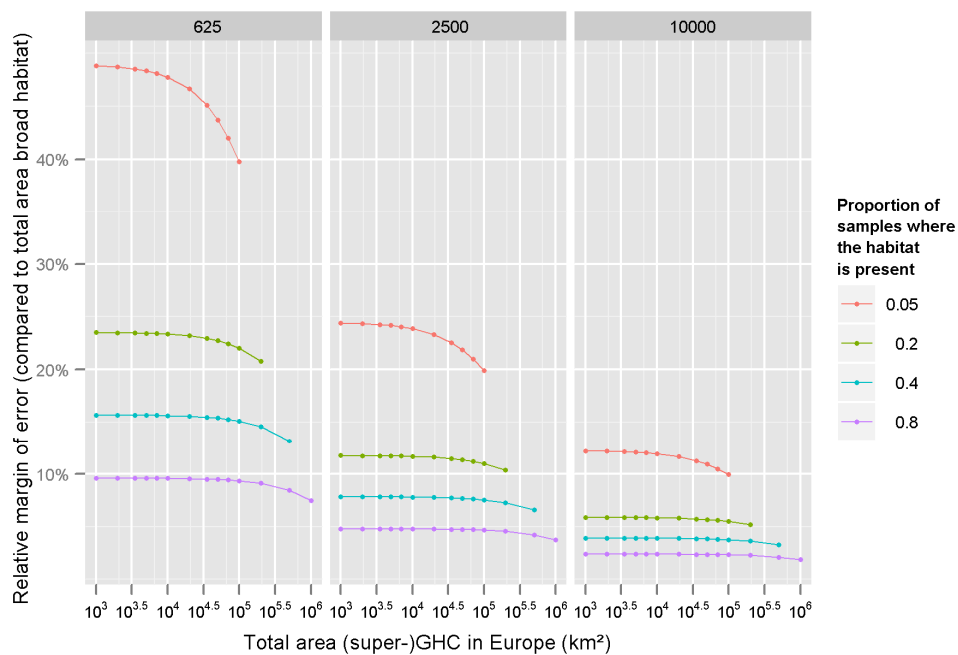


Figure 7.10: Formula-based results for the relative margin of error given the (expected) total area and the proportion of km squares where the habitat is present. The population (or spatial) variance is from the corresponding zero-altered beta distribution.

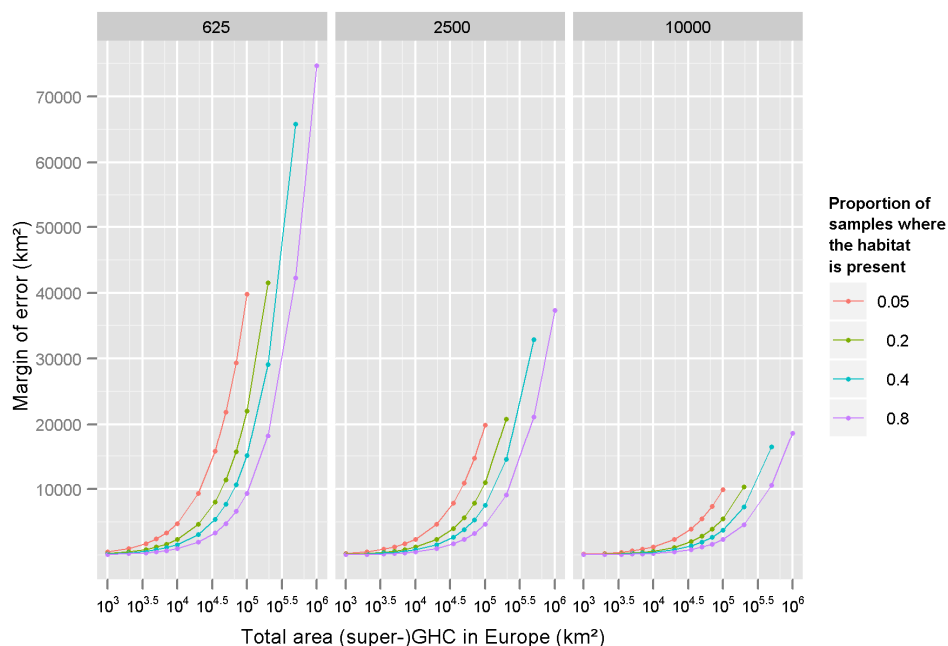


Figure 7.2: The same data as in Figure 7.1, but axes are expressed as absolute values in km².

7.1.4.2. What if a habitat can only occur in a specific biogeographic region?

Many Annex I habitat types are restricted to few biogeographical regions (Figure 7.3) and cover a small total area (often less than 1000 km², see Figure , Figure 7.1 and Figure 7.8:). This raises the question whether we should restrict the scope to only those regions when determining stock estimates, and, if so, what kind of precision can we expect for these rare habitat types.

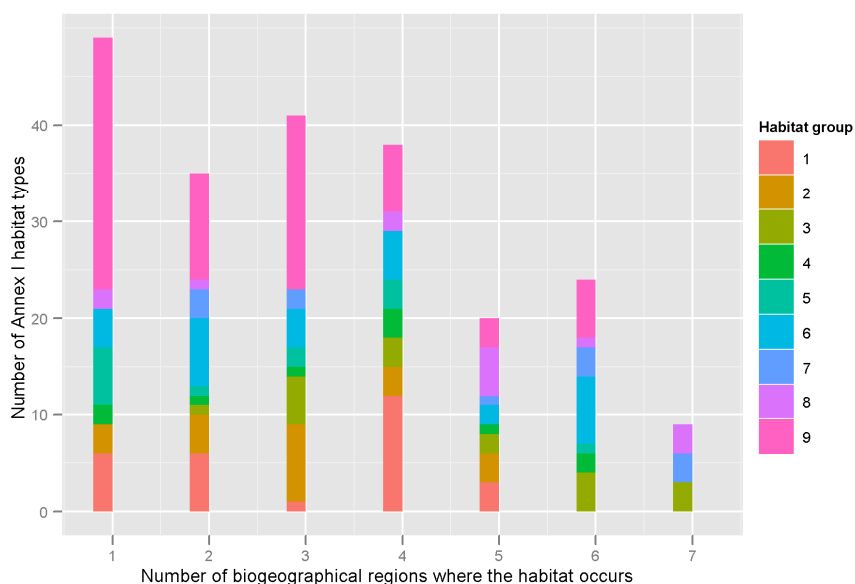


Figure 7.3: The number of Annex I habitat types as a function of the number of biogeographical regions to which they are restricted. Colours indicate habitat groups: 1 coastal and halophytic habitats, 2 coastal sand dunes and inland dunes, 3 freshwater habitats, 4 temperate heath and scrub, 5 sclerophyllous scrub (matorral), 6 natural and semi-natural grassland formations, 7 raised bogs and mires and fens, 8 rocky habitats and caves, and 9 forests

Suppose we take a parameter combination that represents a rare habitat that is present in a restricted area of Europe. We use one of the parameter combinations used in Figure 7.1. Say we want to know the relative margin of error for an Annex I habitat type that can occur in a region that is only 6.25% of the European sample frame. In isolation, that region will count about 625 km-squares that belong to the EBONE design. Suppose further that the total area for that habitat is only 1000 km² and that the proportion of km squares where it is present is only 5% (31 km squares out of 625).

For the total EBONE design, this means that the proportion of samples where the habitat is present is only 0.0031. What would be the precision expressed as the relative margin of error for the stock estimate? It can be shown that this precision is almost equal to the precision obtained when we restrict the geographical scope. With 10000 km squares we have a relative margin of error of 47% (Figure 7.4). We already know from Figure 7.1 that the relative margin of error equals 48% with 625 km squares. Figure 7.4 also shows this for a range of sample sizes and different assumed total areas of habitat. The relationship holds for larger total areas as well, although there is a slight increase in precision with larger sample size (i.e. extending the geographical scope beyond the area in which the habitat occurs naturally). The reason is that the expected decrease in precision due to addition of a large number of zeros, which increase variability given a fixed mean for the zero-inflated beta distribution, is counterbalanced by the expected increase in precision due to increase of the sample size.

This answers the question that stock estimates for many of the Annex I habitat types with a restricted geographical natural distribution will be imprecisely estimated. Brus et al. (2011) already stated in their introduction that the EBONE sampling design is designed for common, widespread habitat and that more tailored monitoring designs are required for rare habitats. Our calculations show that “rare” is in the first place coupled with the number of km-squares where the habitat is present and not so much with the total area of habitat in Europe. Just as we concluded before, presence in 5% of 10000 km squares (500 km squares) already is a very reasonable lower limit, with a corresponding relative margin of error of 12% (Figure 7.1).

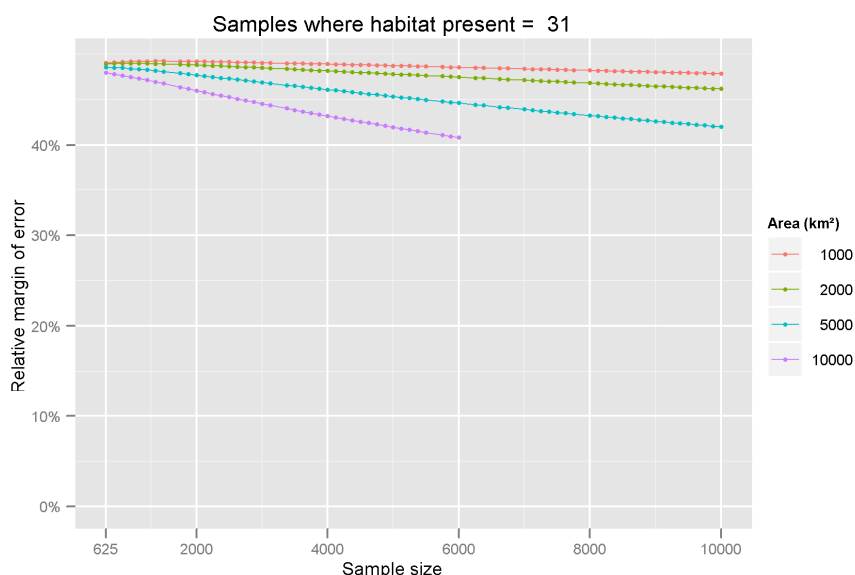


Figure 7.4: The effect of increasing the geographical scope on precision. The hypothetical example starts from a situation where a habitat type is present in 5% of 625 sampling units and a given total area. The precision was calculated starting from the initial condition and recalculating the precision with increase in sample size outside the natural range of the habitat (thus adding excess zeros).

7.1.4.3. Power to detect a trend

According to the Habitats Directive detection of an average 1% per year decline for a habitat type must be possible. The EBONE space-time design is serially-alternating with yearly sampling and a five-year cycle. We therefore translate the requirement from the Habitats Directive into the detection of a 5% decline after 5 years (an average trend of 1%), stronger trends will be detected sooner.

We follow the method to calculate the power of a linear trend according to Brus and de Gruijter (2010) and Brus et al. (2010). Just as before we focus on the calculation of power or precision for a given sample size, rather than the determination of sample size in order to achieve a certain power or precision. This is because we want to evaluate the proposed EBONE sampling design for which a total sample size of 10000 1 km x 1 km sample squares has been put forward. We also approximated the EBONE design as a simple random sample. We did not focus on yearly (annualized) power estimates for the linear trend, but on the power after completion of one or more full cycles (i.e. after multiples of 5 years).

Brus and de Gruijter (2011) derive analytical formula to calculate the variance of a linear trend estimators for a range of different space-time designs. They follow Breidt & Fuller (1999) for the definition of a linear trend:

$$b = \frac{\sum_{j=1}^r (t_j - \bar{t})(\bar{y}_j - \bar{\bar{y}})}{\sum_{j=1}^r (t_j - \bar{t})^2} = \frac{\sum_{j=1}^r (t_j - \bar{t})\bar{y}_j}{\sum_{j=1}^r (t_j - \bar{t})^2} = \sum_{j=1}^r w_j \bar{y}_j$$

With

$$w_j = \frac{t_j - \bar{t}}{\sum_{j=1}^r (t_j - \bar{t})^2}$$

In our case, the t represent the different cycles, r the number of cycles and the y represent the areas (km²) of a habitat type in a km-square.

The formula for the variance of the linear trend than is (Breidt & Fuller 1999, Brus & de Gruijter 2011):

$$Var(\hat{b}) = w'(X'CX)^{-1}w, \text{ with } w = (w_1, w_2, \dots, w_r)$$

The design matrix, X , for our case is simply the identity matrix of size $r \times r$. The variance-covariance matrix C , is assumed to be structured following a first-order autoregressive model (AR1). This means that paired observations which are further apart in time are less correlated. For instance, with an autocorrelation factor, ρ , we have for four cycles:

$$\begin{bmatrix} 1 & \rho & \rho^2 & \rho^3 \\ \rho & 1 & \rho & \rho^2 \\ \rho^2 & \rho & 1 & \rho \\ \rho^3 & \rho^2 & \rho & 1 \end{bmatrix} \frac{\sigma^2}{n}$$

The spatial variance (or population variance), σ^2 , is assumed to be constant. As before, we calculated the spatial variance based on formula for the variance of a zero-inflated beta distribution.

It should be noted that in this way, we simplified the original serially alternating design to a static synchronous design. The reason is that we concentrate our power calculation on change or trend detection after cycles have been completed. We did not look at the intermediate situation where you may be interested in the change after two, three, four, ... nine years. Our results for power are however completely equivalent if we had used the year-by-year design matrix for a serially alternating

design. The following variance-covariance matrices illustrate the difference for a change between two cycles:

$$\begin{bmatrix} 1 & \rho \\ \rho & 1 \end{bmatrix} \frac{\sigma^2}{n}, \text{ for two cycles}$$

$$\begin{bmatrix} I_5 & I_5 \rho \\ I_5 \rho & I_5 \end{bmatrix} \frac{\sigma^2}{n}, \text{ for ten years (two times five years), with } I_5 \text{ the } 5 \times 5 \text{ identity matrix.}$$

Before we present the results, however, we want to explore what range of autocorrelation values could be seen as realistic for the type of data. Breidt and Fuller (1999) estimated the autocorrelation coefficient for the National Resources Inventory, which is a stratified, two-stage area sample of the non-federal lands in the United States. The autocorrelation for the mutually exclusive land classification classes ranged between 0.74 and 0.99. We also consulted Howard et al. (2003) from which we took the estimates of change in area and associated standard errors (Figure 7.5). The autocorrelation could also be estimated from these data. They ranged between 0.66 (neutral grassland) and 1.00 (Montane habitats, Standing open waters and canals).

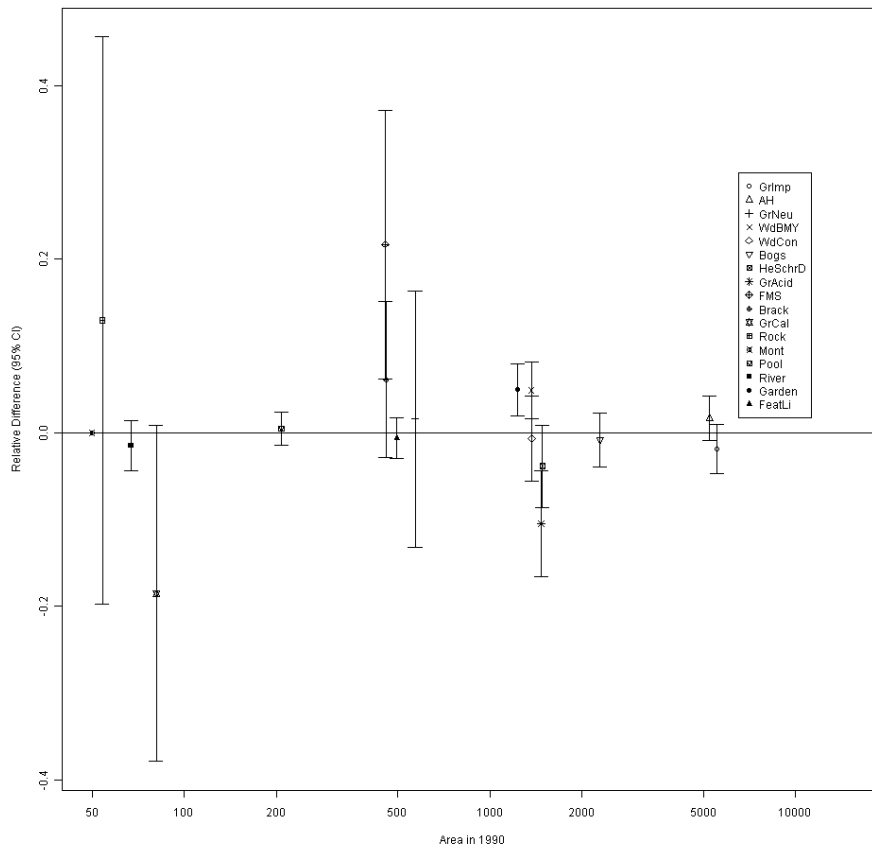


Figure 7.5: Relative difference and 95% confidence interval of the change in area of broad habitat types between 1998 and 1990. Data from Howard et al. (2003).

For our calculations, we let the autocorrelation vary between 0.7 and 0.99, but, we also included the case of no autocorrelation for reference. We used a significance level of 5%. Sample size was set at 625, 2500 and 10000, as before. The total area of habitat ranged between 10^3 km² and 10^6 km² and the proportion of km squares where habitat is present between 0.05 and 1.

The results are presented in Figure 7.6 for the achieved power after two cycles and in Figure 7.7 for the power after five cycles. A first observation is that a high autocorrelation yields a higher power. Similarly, the proportion of km squares where habitat is present also had a positive influence on the power to detect the linear trend. It should be noted however that we assumed in our calculations that when a habitat was absent from km-squares during the first cycle, it remained absent during the subsequent cycles. A third observation is, of course, that power increases with increase in sample size: a fourfold increase in sample size doubles the power. Fourth, with time, the power will increase as can be seen from the upward shift in power for two versus five cycles.

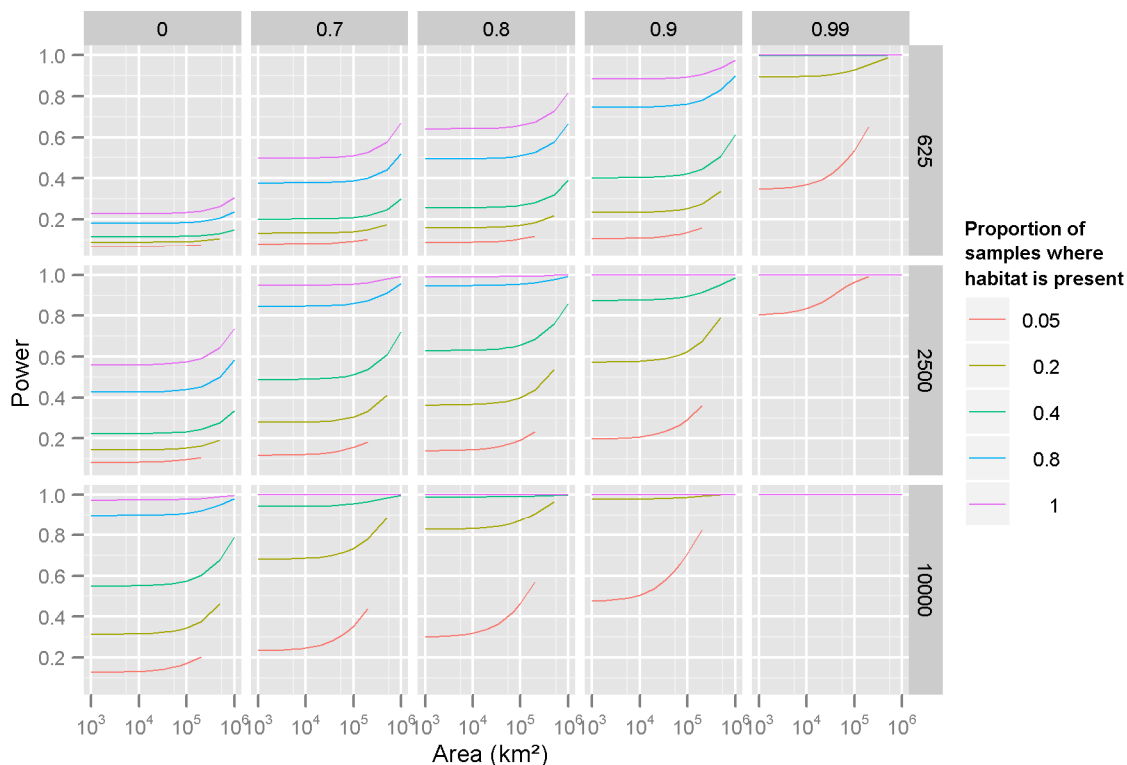


Figure 7.6: Achieved power for detection of a linear trend (5% change after 5 years) after two cycles. The column headings give the autocorrelation value. The row heading give the sample size. Colours indicate the proportion of km squares where habitat is present.

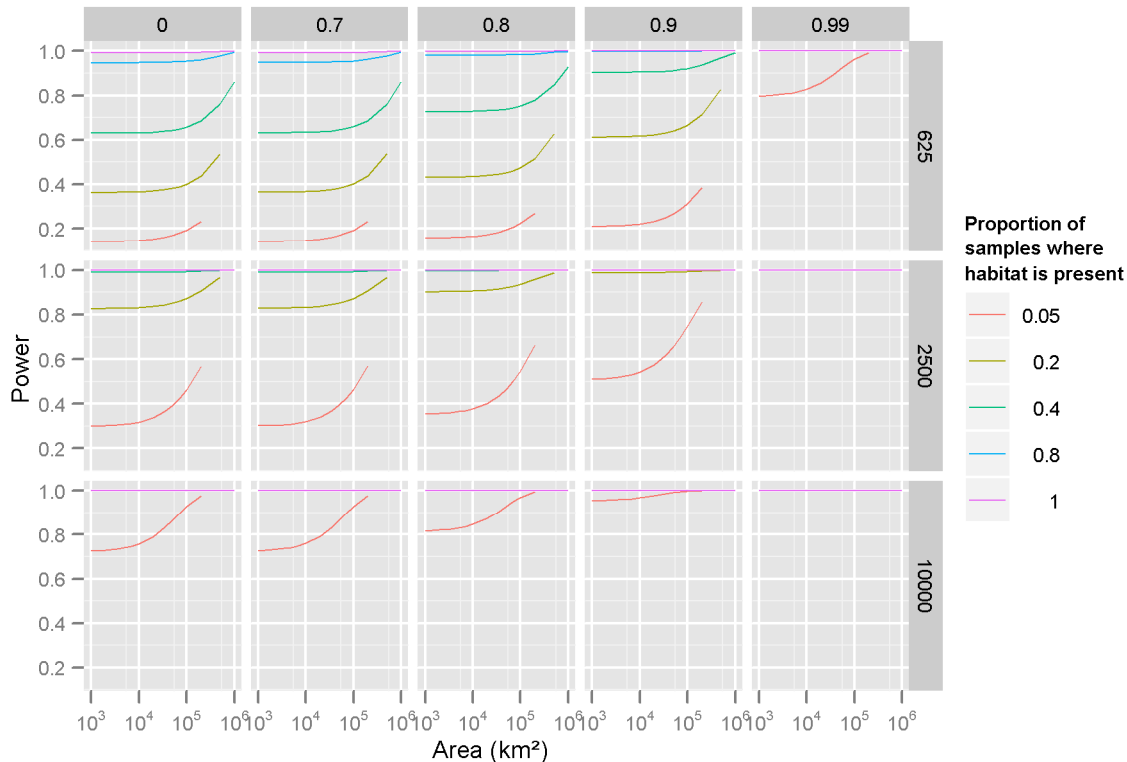


Figure 7.7: Achieved power for detection of a linear trend (5% change after 5 years) after five cycles. The column headings give the autocorrelation value. The row heading give the sample size. Colours indicate the proportion of km squares where habitat is present.

7.1.5. Precision and accuracy of stock estimates based on earth observation sampling units

We can now ask if we could increase the quality of the estimates of habitat extent by sampling additional 1 km² sampling units by earth observation (EO). But, how do we incorporate the reduced accuracy of EO sampling units compared to in-situ sampling units?

In order to have an indication of the accuracy of EO sampling units compared to in-situ field measurements (GHC methodology), we can rely on accuracy matrices. An accuracy matrix is an N x N matrix of "observed" and "classified" cells corresponding to N land cover classes (derived from GHC classification). The matrix depicts the land cover classification category derived from earth observation versus the field-observed land cover type. The diagonal cells indicate correct observations, i.e. classified correctly according to the field observations. Any observation off the diagonal indicates a misclassification. Especially, it would be very helpful to have realistic values for both producer and user accuracy for several of the broad habitat categories. Producer accuracy tells us, from the point of view of someone who produces a map based on earth observation data, how well the producer classified the earth observation data (i.e. how many times does the producer mistakenly classifies a pixel to habitat A; false negative = omission errors). User accuracy takes the point of view of users of the map and tells them how well the EO-derived map performs in the field (i.e. how many times does the map tell us we are standing in habitat A, while in reality we are not; false positives = commission errors).

We used data from Morton et al. (2007) to see the range of values that can be expected for user and producer accuracy. The authors discuss the first UK land cover map (LCM2007) with land parcels

(the spatial framework) derived from national cartography by a generalisation (simplification) process. LCM2007 is the first land cover map to provide continuous vector coverage of UK Broad Habitats derived from satellite data. The data from Morton et al. (2007) allow the comparison between the field data gathered during the 2007 UK Countryside Survey (591 1 km x 1 km squares in Great-Britain) and aerial photos with high spatial resolution (25 m). Note that the UK CS squares were not used as ground reference points/data to produce the LCM2007 map. The comparison was made *ex post* as a sort of quality check to gain confidence in the LCM2007 map. A separate set of ground reference points, which is not of interest for our purpose here, was used to calibrate and validate the LCM2007 classification.

Table 7.3: shows the accuracy results when the EO map (LCM2007) is compared with the CS 1km survey squares (data from Morton et al. 2007).

LCM2007	Countryside survey in 2007										Sum of columns (ha)	User accuracy
	Broadleaved woodland	Coniferous woodland	Arable and horticulture	Improved grassland	Semi-natural grassland	Mountain, heath, bog	Saltwater	Freshwater	Coastal	Built up areas and gardens		
Broadleaved woodland	1212	205	42	131	245	24	2	4	2	92	1959	0.62
Coniferous woodland	134	2503	9	23	84	85	1	3	1	26	2869	0.87
Arable and horticulture	103	41	8643	1944	676	31	3	3	3	424	11871	0.73
Improved grassland	194	38	654	7533	2769	50	4	3	3	371	11619	0.65
Semi-natural grassland	135	80	170	1066	2660	1377	5	10	22	153	5678	0.47
Mountain, heath, bog	84	155	13	43	1068	3479	5	9	5	49	4910	0.71
Saltwater	0	0	0	0	0	0	172	0	2	0	174	0.99
Freshwater	1	2	2	1	3	2	54	346	5	3	419	0.83
Coastal	3	0	42	20	63	3	335	0	87	8	561	0.16
Built up areas and gardens	27	1	48	24	36	1	0	2	2	1109	1250	0.89
Sum of columns (ha)	1893	3025	9623	10785	7604	5052	581	380	132	2235	41310	
Producer accuracy	0.64	0.83	0.90	0.70	0.35	0.69	0.30	0.91	0.66	0.50		0.67

Table 7.3: Accuracy table for the 2007 countryside survey compared to the UK land cover map. Data from Morton et al. (2007).

Figure 7.17 retakes some of the data in Table 7.3:.. It shows the range of observed data for user and producer accuracy. Most broad habitat types do not deviate far from the line of no difference. Coastal and Saltwater deviate much from the 1:1 line but in opposite directions, which probably just indicates that they represent two sides of the same coin: the omission errors made in one class are offset by commission errors in the other class. Furthermore, we observe that (when Coastal and Saltwater are excluded) the user accuracy range is from 0.5 to 0.9, while producer accuracy ranges from 0.35 to 0.9.

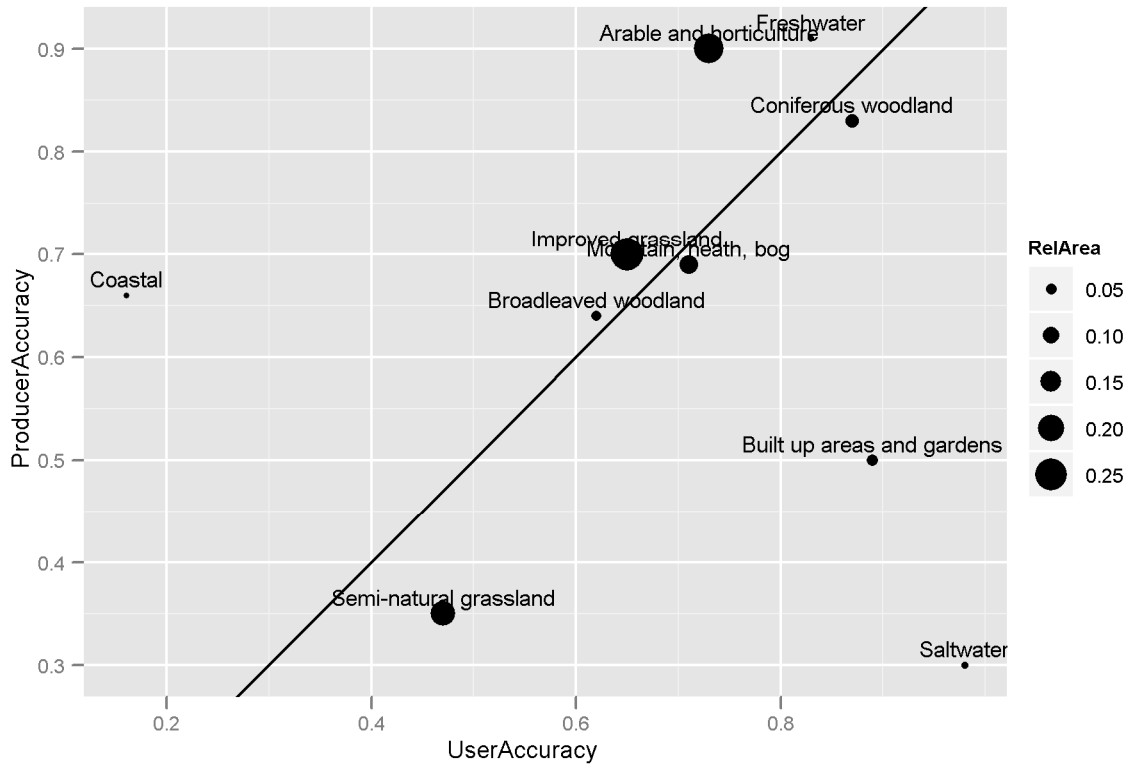


Figure 7.17: Scatterplot of producer versus user accuracy. The size of each circle (broad habitat type) is proportional to the estimated total area for that habitat type.

We can single out each broad habitat type in Table 7.3: and construct a new table that looks like the one presented in Table 7.4.:

Earth observation	In situ samples (GHC methodology)		Sum	User accuracy
	Habitat A	Not habitat A		
Habitat A	TP	FP	TP + FP	TP / (TP + FP)
Not habitat A	FN	TN	FN + TN	
Sum	TP + FN	FP + TN		
Producer accuracy	TP / (TP + FN)			

Table 7.4: A simplified accuracy matrix that compares a focal broad habitat type against all non-focal habitat combined. TP = true positives, FN = false negatives, FP = false positives and TN = true negatives.

From the perspective of the earth observation map producer, we have:

$$\text{Producer accuracy} = P(EO = A | IS = A) = \text{Sensitivity} = \text{True positive fraction}$$

Thus, the producer accuracy equals the conditional probability of observing the focal habitat with earth observation given the in situ observed focal habitat. When the latter is observed without error, the producer accuracy is also referred to as the sensitivity or the true positive fraction (Quataert 2011).

From the perspective of the users of this map, we have:

$$\text{User accuracy} = P(IS = A | EO = A) = \text{Positive predictive value}$$

Thus, the user accuracy equals the conditional probability of observing the focal habitat in the field given the distribution of the focal habitat on the earth observation map. This fraction is also called the positive predictive value (Quataert 2011).

We will now use both fractions to explore the precision and bias of EO sampling units. But before we do this, we make the following assumptions:

- When a habitat is present in a km-square sample, both earth observation and in situ sampling will detect it
- We assume that the in situ sampling units, represent ground truth. They represent the true state without measurement error.

We can use Bayes rule to relate the producer accuracy to the user accuracy:

$$P(A|B) = \frac{P(B|A)P(A)}{P(B)}$$

$$P(EO = A|IS = A) = \frac{P(IS = A|EO = A)P(EO = A)}{P(IS = A)}$$

If we rearrange this, we can estimate the unconditional probability of observing habitat A with earth observation as a function of both user and producer accuracy and the ground truth probability for habitat A:

$$P(EO = A) = \frac{\text{Producer accuracy}}{\text{User accuracy}} P(IS = A)$$

Equation 1

The following conclusions can be drawn from Equation 1 regarding potential bias of the EO-sampling units:

1. When $PA = UA$, the earth observation estimates will be unbiased
2. When $PA \neq UA$, the earth observation estimates will be biased
 - a. $PA > UA$, results in a positive bias (overestimation)
 - b. $PA < UA$, results in a negative bias (underestimation)
3. When PA and UA can be estimated, we have a means to correct for bias. The bias-corrected estimates can be calculated as follows: $P(EO = A) \frac{\text{User accuracy}}{\text{Producer accuracy}}$
4. Given the previous conclusion, we need to make sure that PA and UA estimates can be obtained from the combined IS and EO sample design
5. Even when PA and UA are much smaller than 1, unbiased estimates can be obtained (for the average or total area in the sample); however, the produced EO maps will have very large spatial errors regarding the spatial location of the habitat and will therefore be useless.

Equation 1 can also be used in simulation of the effect that earth observation has on precision and accuracy (bias). We simulated data from a zero-inflated beta distribution, but simply replaced the simulated values (habitat area in a 1 km x 1 km square) with the value that would result if the sample were observed with earth observation (Equation 1). We used values for user and producer accuracy from 0.6 to 0.8 in steps of 0.1. To be able to compare with the result from in situ sampling units we also included user and producer accuracy equal to 1 (i.e. the situation where EO and in situ produce exactly the same result: 100% accuracy).

The conclusions regarding bias – discussed above – are depicted in Figure 7.8.

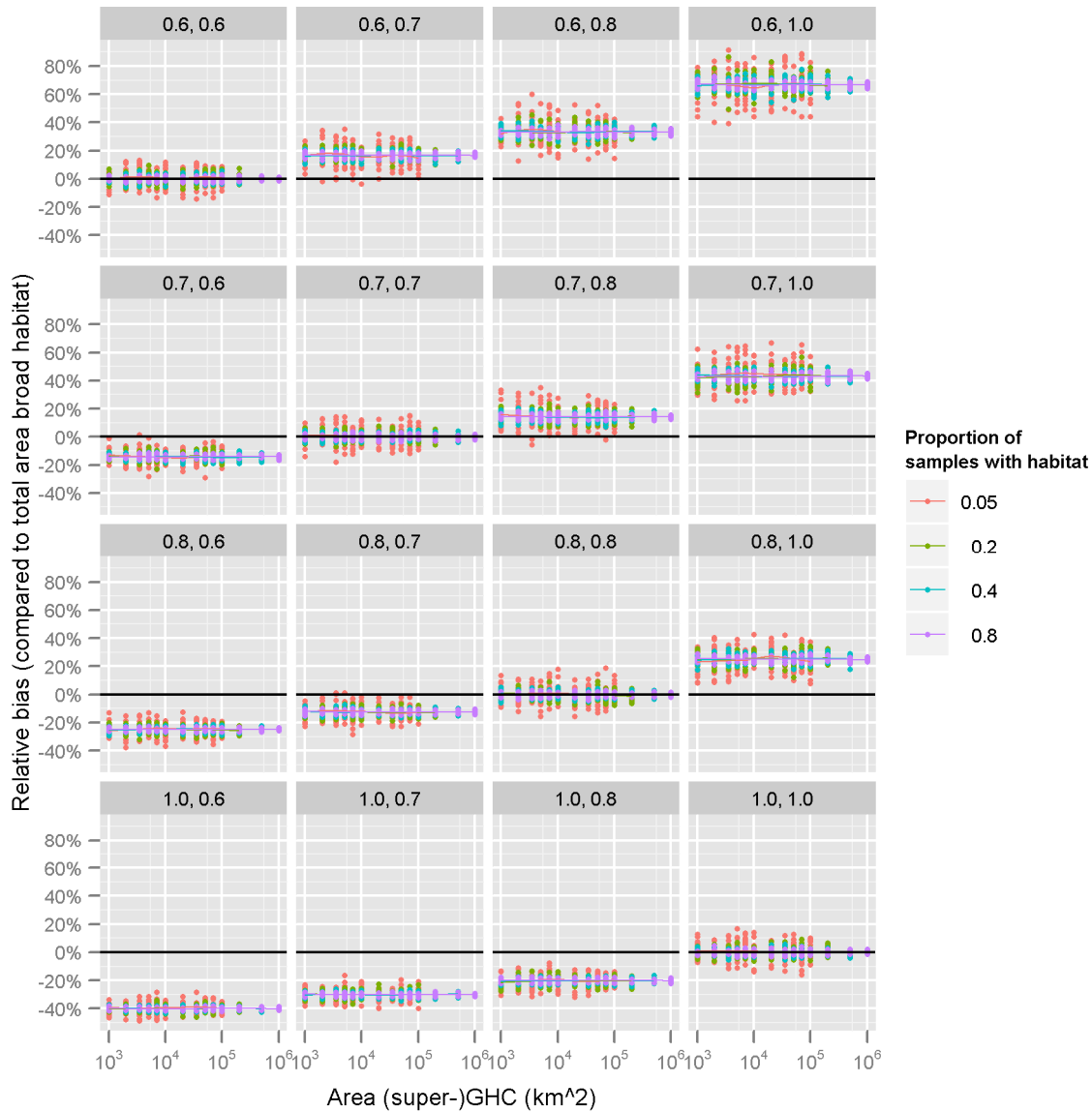


Figure 7.8: Relative bias as a function of user and producer accuracy. The heading above each panel gives user accuracy and producer accuracy respectively. The lower right panel corresponds with the situation where in situ sampling units and earth observation sampling units give identical results (i.e. 100% accuracy; for comparison purposes only). The sample size is equal to 10000.

Starting from equation 1, we can also derive what effect the user and producer accuracy have on precision if we calculate the variance:

$$\text{var}[P(EO = A)] = \text{var}\left[\frac{PA}{UA} P(IS = A)\right] = \frac{PA^2}{UA^2} \text{var}[P(IS = A)]$$

The latter equation and inspection of Figure 7.9 allow us to draw the following conclusions regarding precision (which is inversely related to the relative margin of error; the margin of error equals the half-width of a 95% confidence interval):

1. When $PA = UA$, the precision is the same as that obtained from an IS sample (sampling errors only; no measurement errors assumed)
2. When $PA \neq UA$, the precision is different from that obtained from an EO sample:
 - a. $PA > UA$: a less precise estimate will be obtained
 - b. $PA < UA$: a more precise estimate will be obtained

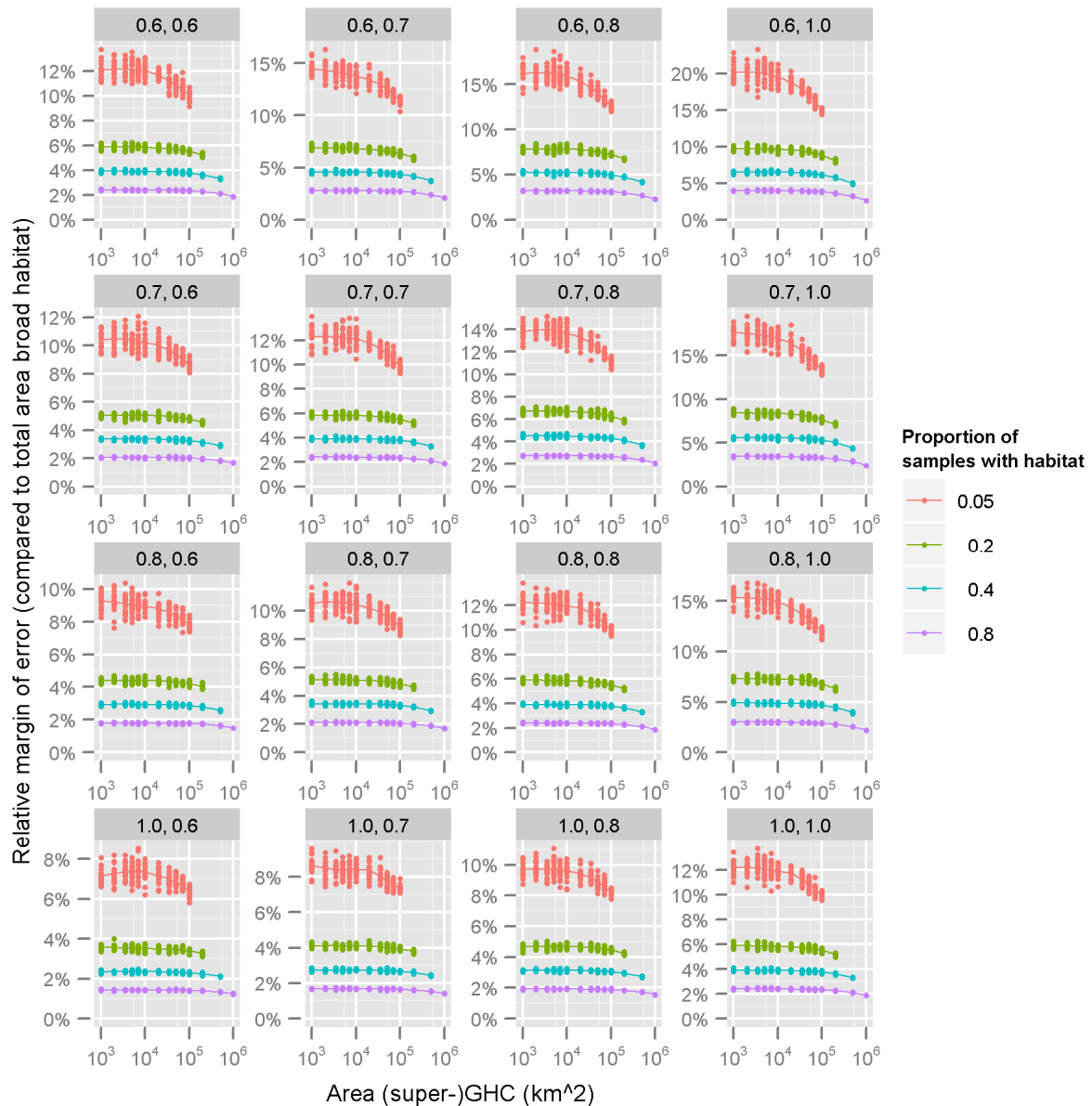


Figure 7.9: Relative margin of error as a function of user and producer accuracy. The heading above each panel gives user accuracy and producer accuracy respectively. The lower right panel corresponds with the situation where in situ sampling units and earth observation sampling units give identical results (i.e. 100% accuracy; for comparison purposes only). Sample size used was 10000.

It is clear from the above discussion that the effect that earth observation sampling units has on precision or bias will depend crucially on differences in user and producer accuracy. The most important take home message is probably that we can correct for possible systematic bias if and only if the earth observation sample and the in situ sample partly overlap. This overlap, however, should be sufficiently large to ensure that user and producer accuracy themselves can be estimated precisely and without bias. In this respect, it is also crucial that the overlapping part of both sampling units is a spatially balanced, random sample to avoid bias.

7.1.6. Cost-effectiveness of the sampling design

Starting from the sampling design scenario's, we can explore the implications for a cost-effective monitoring design. The problem we want to solve is how to achieve a good balance between output quality of the design and available monetary budget or alternatively, the constraint could be formulated in terms of time. The effectiveness can often be related to statistical concepts, such as the margin of error or the sampling variance. Which measure for effectiveness will be most useful will depend on the question at hand. For estimation of a mean or a total, higher effectiveness is related to a narrower confidence interval, as we have shown. For trend detection, the effectiveness will depend on the power to detect a trend, and thus this will depend on the magnitude of the trend that needs to be detected.

For a given sample size, we can thus assess effectiveness. The pilot data gathered during the EBONE project also allow us to get insights into time requirements for field work (see 7.2 for more details). Confronting the time requirements with the effectiveness yields a first rough approximation of cost/time-effectiveness.

Table 7.1 gives a summary of effectiveness for stock and change detection and time requirements for the EBONE design.

At the European level, precise stock estimates can be obtained for habitat that is present in 5% or more of the sampling units. Change estimates ($\pm 5\%$ after 5 years) will also have sufficient power ($> 80\%$) provided that the habitat occurs in at least 5% of the sampling units and that autocorrelation is very high (which is often the case). However, rare habitat types, among which many Annex I habitat types will not have precise stock estimates nor sufficient power after two cycles for change detection.

At the level of an average biogeographical zone, the reduced sample size evidently lowers power and precision. Still, for common and widespread habitat types precise stock estimates can be expected (cf. the UK CS). Change detection depends strongly on the autocorrelation that can be expected for the habitat type. Insufficient power is certain for dynamic habitats, whereas stable habitat types may have fairly high power to detect the change.

			European level (n = 10000)	Level biogeographical zone (n = 850)
	Proportion km squares where habitat is present	Autocorrelation	Quality objectives met?	Quality objectives met?
Stock	0.05		(Yes)	No
	0.5		Yes	(Yes)
Change (two cycles)	0.05	0.7	No	No
		0.99	Yes	No
	0.5	0.7	Yes	No
		0.99	Yes	Yes
Time per cycle (person days)			80000 - 110000	6800 - 9350
Time per year (person days)			16000 - 22000	1360 - 1870
Full time equivalents (FTE) per year			80 - 110	6.8 - 9.35

Table 7.1: Evaluation of effectiveness of the EBONE sampling design and a tentative indication of the amount of time required to gather and input the data.

7.1.7. Conclusions

The EBONE sampling design is effective and efficient for relatively common/widespread habitat. However, for many Annex I habitat types tailor made monitoring designs remain necessary. We see this exercise as a first approximation to performance that can be expected from the EBONE sampling design. As a general rule, precision and power calculations are not sacrilegious. The details of the monitoring design are just as important, as well as the institutional, data management and other factors that will be crucial for implementing the design. We therefore also recommend that, an implementation phase will be foreseen, and that data gathered during that phase will be analysed again in the light of statistical performance, field procedures, data storage, etc.

8. Institutional aspects of European wide biodiversity monitoring

8.1. Towards a federation of National Biodiversity Observation Networks

A long-term, pan-European biodiversity monitoring project, involving various countries and institutions, serving a diversity of users, and depending on voluntary commitment and active collaboration of the stakeholders, needs a proper and efficient governance structure. However, proposing yet another and totally new initiative for coordination and co-operation, makes no sense. For that purpose, already a number of structures are established that facilitate (with more or less success) mutual exchange of information, common data storage, standardization and inter-calibration of norms, harmonization of methodologies and protocols.

In a special report of EBONE, D1.3 '*Recommended Institutional Framework*' (Jongman et al., 2011), an assessment and analysis of the current situation regarding management of the different pan-European biodiversity monitoring initiatives, the data infrastructure organisation and the reporting obligations, in Europe and beyond, are given. In that report it is concluded that for the objectives pursued by EBONE and for the monitoring procedure developed, the best way forward for EBONE or a similar programme is to link up with GEO BON.

GEO, the 'Group on Earth Observations' is coordinating efforts to build GEOSS, a 'Global Earth Observation System of Systems'. GEO was launched in response to calls for action by the 2002 World Summit on Sustainable Development by the G8, recognizing the need for international collaboration for exploiting the growing potential of Earth observations to support decision making. GEO is a partnership of governments and international organizations. It includes 76 governments and the European Commission as well as 51 intergovernmental, international, and regional organizations with a mandate in Earth observation or related issues, the Participating Organizations (situation 2010).

GEO BON stands for the 'Group on Earth Observations Biodiversity Observation Network'. By facilitating and linking efforts of countries, international organizations, and individuals, GEO BON wants to contribute to the collection, management, sharing, and analysis of data on the status and trends of the world's biodiversity. The network is established on a voluntary and legally non-binding basis, with voluntary contributions supporting its activities. No one has an authority to direct participants. The purpose is merely to provide guidance and recommendations to participants and contributors. Nevertheless, also in GEO BON the need for coordination and co-operation remains. It is argued by Jongman et al. (2011) that this should now be organized on the European level. Therefore clear recommendations are given in the report:

Require cooperation within countries or regions could be achieved in National Biodiversity Observation Networks (NBON) or Regional Biodiversity Observation Networks (RBON). Besides, it will also be dependent on the establishment of effective and efficient networks within these NBONs between the executing monitoring agencies, NGOs, science groups and the clients. The overarching European umbrella organisation can be a federation of National BONs, with supporting exchange mechanisms for the clients, the data providers and the European and global mechanisms regarding reporting on the state of biodiversity.

National BONs can have different compositions in different countries. There are countries and regions where agencies do not carry out monitoring work, but let this be done by NGOs, scientific institutions or consultancies. There are also countries where NGOs are poorly developed and activities are being carried out by consortia

of agencies and universities. So, a uniform concept of the constitution of a BON does not exist.

Fig.1. illustrates the potential framework of a National Biodiversity Network (from EBONE deliverable D.1.3 - Jongman et al., 2011).

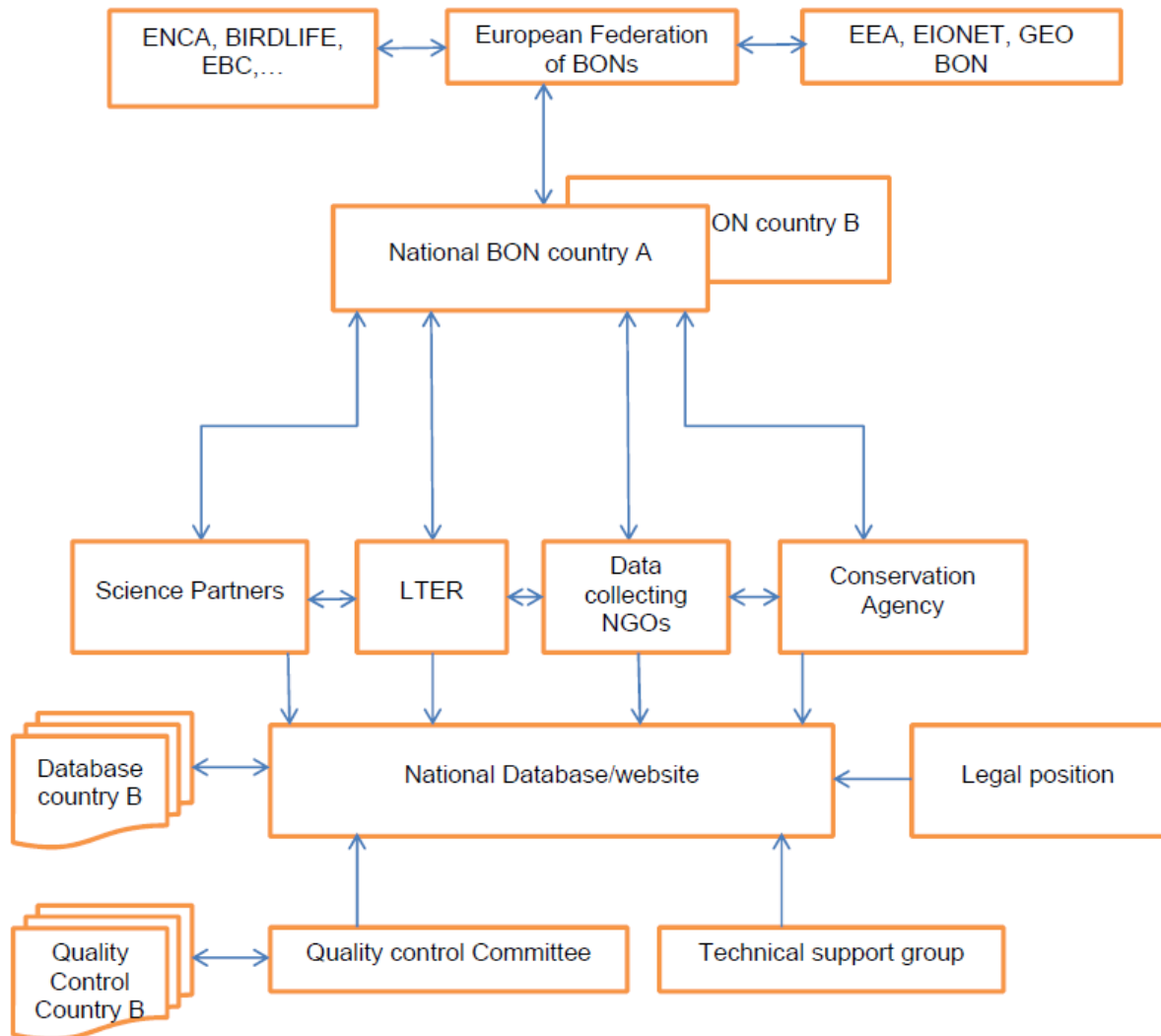


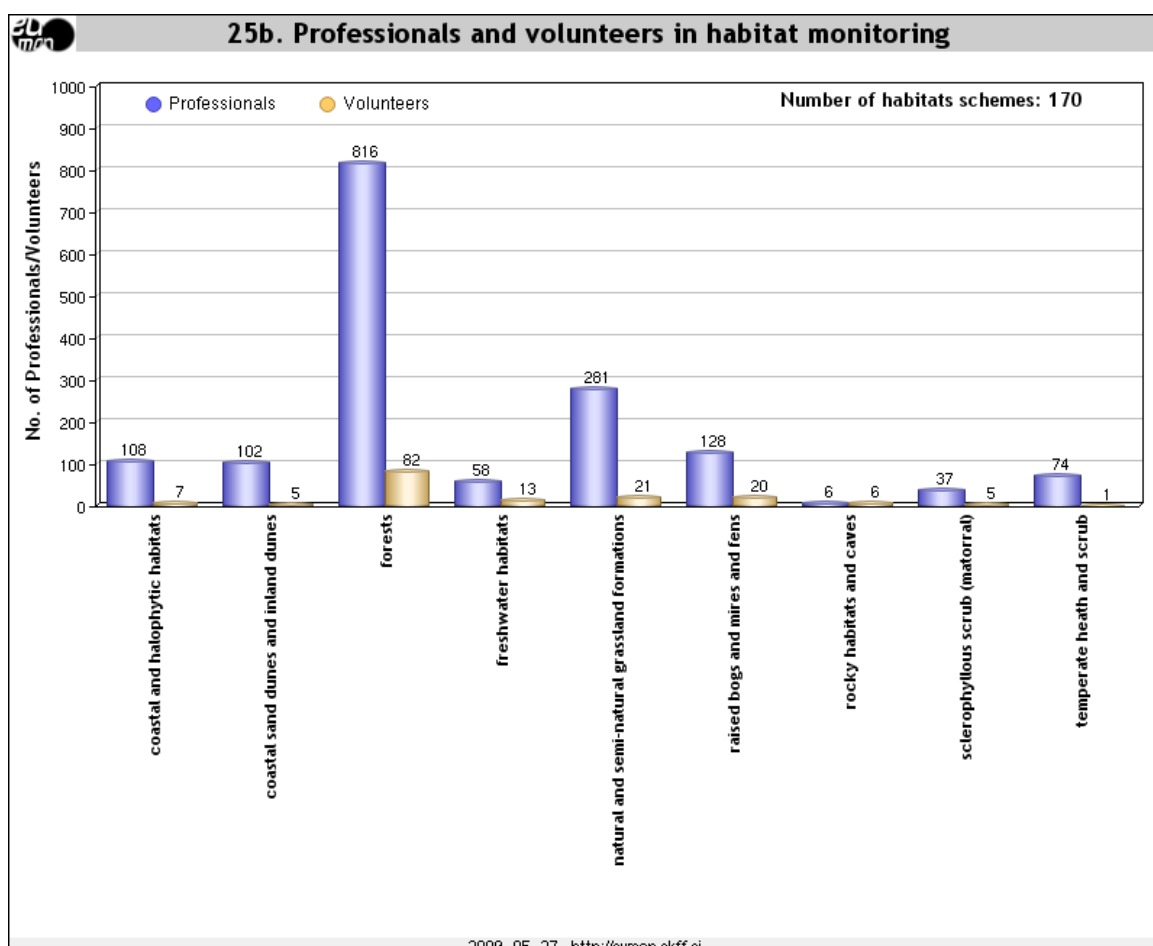
Figure 1. Structure of a national Biodiversity Network in the context of its territory and in relation with international organisations.

8.2. Contribution of volunteers in field data collection

Biodiversity monitoring faces at least 2 practical difficulties: (1) the need to maintain a sustained effort of monitoring across years to ensure the collection of relevant monitoring time series and (2) the need to obtain precise monitoring data that allow the detection of significant changes across space and time in biodiversity. These needs come into conflict with the usually limited amount of available financial and human resources (Schmeller et al. 2008). As precision of monitoring data is a function of the number of monitored sites, one possible way to cost-effectively maximize sampling effort is the involvement of volunteers in field data collection.

Findings in the Eumon-project (see also §3.2) do not support the common belief that volunteer-based schemes are too noisy to be informative and are prone to higher biases than professional monitoring schemes (Engel & Voshell 2002, Genet & Sargent 2003). Several examples show the opposite and prove that volunteer-based schemes can provide relatively reliable data with state-of-the-art survey designs or data-analysis methods, and consequently can yield unbiased results (Schmeller et al. 2008). In Europe many monitoring organizations already rely on such volunteer programs (e.g. van Swaay et al. 2008, Thomas 2005, Gregory et al. 2005, Voříšek et al. 2011). The involvement of the public seems to be imperative to reduce the cost of biodiversity monitoring and has the added value of enhancing citizen participation in science practices and thus environmental awareness (e.g. Bell et al. 2008). Quality of data collected by volunteers is more likely determined by survey design, analytical methodology and communication skills within the scheme rather than by volunteer involvement per se (Schmeller et al. 2008).

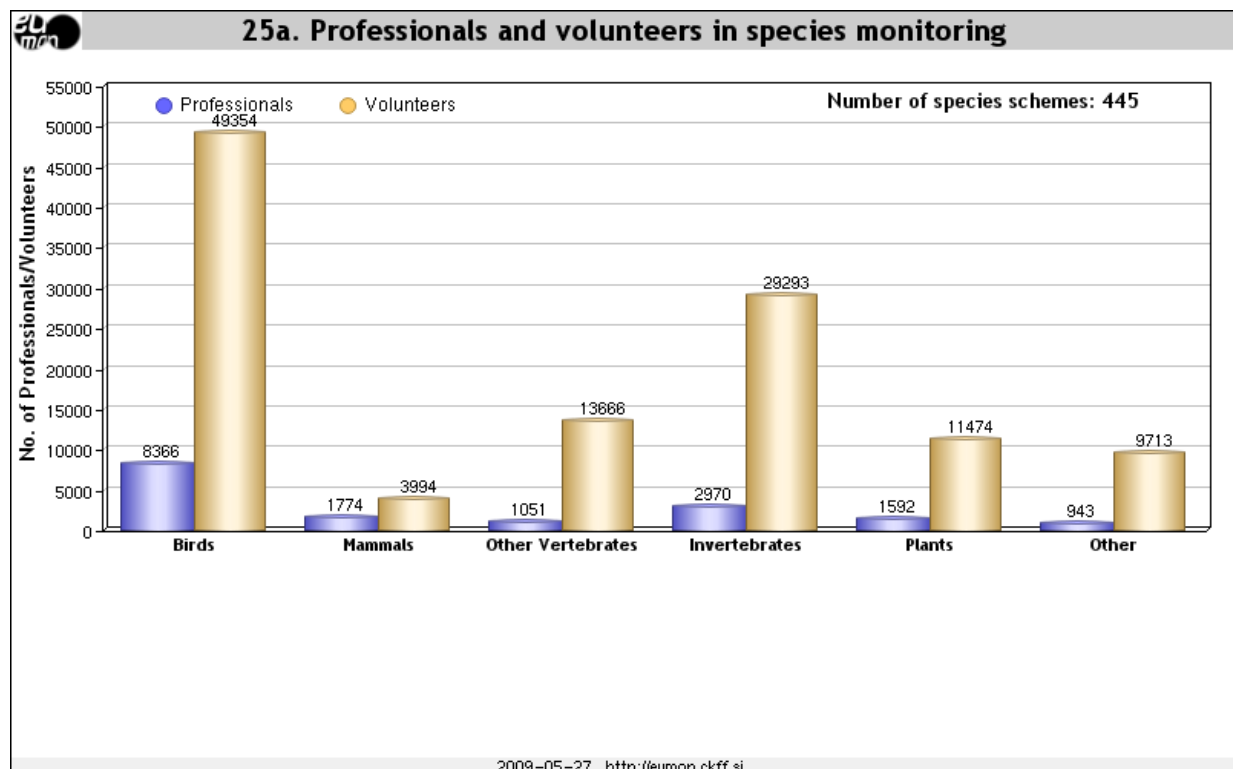
A good and recent overview of potential and benefits, as well as constraints and shortcomings of citizen participation to biodiversity knowledge and monitoring is given by Zisenis et al. (2011). One restriction to volunteer involvement seems to be the main focus on species monitoring. Habitats are underrepresented in current monitoring schemes with substantial citizen involvement (BioMAT 2011) and this type of monitoring will probably remain less attractive for volunteers. A reason for this can be



Excerpt from the Eumon-database : *Professionals and volunteers in habitat monitoring*

that habitats are less unequivocal and often show a complex description. As a result, more common training is needed and more rules to apply to achieve reliable recording. Besides, because of the complexity and the relationship with the landscape scale, sustained efforts in the field are required, which may make habitat monitoring less satisfactory in relation to the time spend.

Additionally species monitoring is mainly focused on a few popular species groups. For birds, vascular plants, butterflies, mammals, and some other groups of organisms, the amount of volunteers is substantial, but for many other groups, there are only few people with sufficient species knowledge. This is the reason why there is a huge amount of data on birds and vascular plants, but nearly no data available, for example, on centipedes and fresh water algae (Zisenis et al. 2011). However, species are unequivocal. So, the specialist's skills are part of the profile of volunteers and they are recognized for that. Efforts and time spend while monitoring species are more in balance with the degree of satisfaction.



Excerpt from the Eumon-database : *Professionals and volunteers in species monitoring*

To increase volunteer collaboration for habitat or EBONE recording, some action can be undertaken. Closer co-operation with professionals can be pursued by means of joined survey training or by forming mixed survey teams. The latter however may provoke problems, one member of the team being paid and the other not. One may also consider the adaptation of the often very detailed habitat classification and key. Simplification could be achieved when intermediate monitoring only focuses on recording changes since the last monitoring session. With this as an objective, the professionals have to select the appropriate habitat characteristics for the recording.

The professionals should The main constraint remains however: here is yet no network of volunteers for habitat recording or monitoring. Volunteer recruitment also has a geographic component to consider. For example, in Eastern Europe in general it is more difficult to attract volunteers to participate in biodiversity monitoring. The lack of volunteer participation may be linked to political and economic conditions in some countries (Schmeller et al. 2008).

It is also crucial to understand what citizens can perform and cannot perform. Volunteers collect data because of fun and of usefulness. They perform the work on their free time and free time is a limited resource. With a small amount of resources, it is possible to improve the quality and substantial harmonization. However without full payment, it is not possible to encourage volunteers doing anything (Zisenis et al. 2011).

Nevertheless, the involvement of volunteers in a structured way has an enormous potential for increasing the knowledge and for monitoring the trends of biodiversity. Next to already existing projects and schemes, Zisenis et al. (2011) suggest that there is a large potential to even increase the number of participants in Citizen Science observations and monitoring. Some preconditions for engaging people in the monitoring project are e.g. information available on the Internet, a good online-system in place showing that people can participate in a larger context, and the guarantee that the information collected will be used. Zisenis et al. (2011) give a general framework and key factors for successful involvement of volunteers in biodiversity observation and monitoring (see also figure 9.1). The objective and methods must be clear and very easy to explain and understand. A structured scheme linking participants together and a well-designed and standardized protocol are needed. Scientists must ensure that the protocol matches what people can and want to collect. The protocol must also allow rigorous data analysis and interpretation. Also feedback is very important. People involved must be aware of what is done with their data and why. Results must be rapidly available and regularly updated. A good communication strategy is also crucial to recruit new participants and to gain credence of people involved. The continuity must be guaranteed. A permanent professional team is needed for coordination and to ensure that the general framework is working and that data are actually leading to analysis and publications (Cooper et al. 2007, Devictor et al. 2010, Zisenis et al. 2011). Validation of the data reported is also crucial, if they are to be used for biodiversity assessments in support to policy decision (Zisenis et al. 2011).

Schmeller et al. (2008) suggest that in countries with high salary costs biodiversity-monitoring obligations can only be implemented if volunteer involvement is maintained or increased through targeted recruitment campaigns and if funds are provided for the coordination of volunteer-based monitoring programs. Such a biodiversity monitoring can be organized by for example national agencies or universities. Non-governmental organizations with successful volunteer involvement could serve as partners for such campaigns.

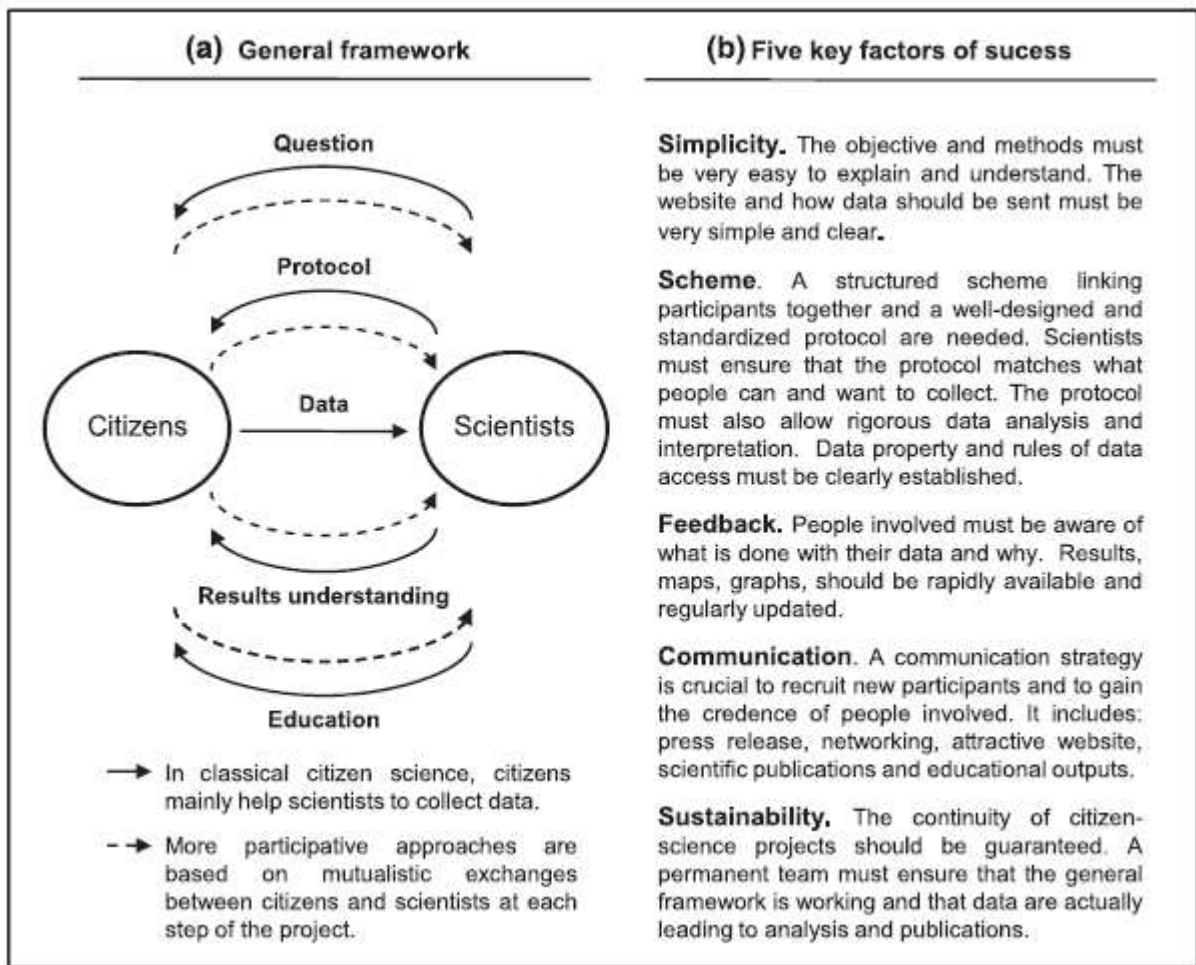


Figure 9.1: Conceptual framework and key factors of success of a volunteer-based biodiversity monitoring programme. (a) A general framework generates a reciprocal connection between scientists and citizens from the question being asked to the educational benefit. This framework can range from top-down projects (black arrows) to more bottom-up and participatory approaches (dashed arrows) depending on whether and how citizens are involved (adapted from Cooper et al. 2007). (b) To ensure that the framework is actually working and maintained requires several key factors that encourage success (Devictor et al. 2010). (Taken from Zisenis et al. 2011).

9. A decade of experience in national-scale landscape biodiversity monitoring – National Inventory of Landscapes in Sweden (NILS)

Author: Johan Svensson (SLU)

9.1. Introduction

Unknown land use and land management premises are to be expected due to new and challenging premises caused by, e.g., new land use strategies and priorities, climate change, and globalizing natural resource markets. This implies a critical need for timely, accurate, relevant and applicable biophysical landscape data input into governance systems and decision-making processes on national and pan-national levels. Continuous supply of relevant information is imperative for the decision-making at all levels, from global policy conventions to land-use management decisions on specific estates and sites (Haines-Young et al. 2003, Löfvenhaft 2002, Ahlqvist 2008). Governance systems encompasses both landscape planning modules that provide the operational and tactical applications, and policies, frameworks and conventions that provide the strategic overview. Moreover, as policy, legislation and other fundamental strategic groundwork more and more evidently becomes a pan-national issue (Nassauer and Opdam 2008, Bunce et al. 2008), there is a need to develop landscape-based monitoring systems that are pan-national by nature, but that also have an inherent capacity to comply to national and local premises and needs. This requires monitoring systems that are based on an understanding of ecosystem processes on appropriate scales, that encompass features that are possible to monitor with adequate accuracy given the available techniques, and that relate to policy and decision-making in a societal context (Noss et al. 1992, Lovett et al. 2007).

In a wider context it is also needed to integrate biophysical landscape data with socio-economic data to provide holistic, socio-ecological background information for sustainable land use and management on strategic and operational levels (Rametsteiner 2009). Although environmental monitoring exists in many countries, there is a growing need, thus, for monitoring infrastructures that allow for broader applications covering economic, ecological and socio-cultural dimensions on landscape scale. Moreover, monitoring need to have an inherent capacity to adjust to societal needs and, if demanded, also inclusion of new or supplementary variables, albeit with long-term robustness in core data and methods in monitoring to allow consistency in analyses and reporting schemes (Svensson 2009). Several biodiversity-oriented environmental monitoring programs are ongoing in different countries, although most of them have been established fairly recently (Ståhl et al. 2011). At present, however, there is a lack of consistency between different programs, which impede sharing of knowledge, experiences and information. Approaches towards standardized framework of surveillance and monitoring on European level are being developed (Bunce et al. 2008, McRoberts et al. 2010), however, as in the EBONE project.

The Swedish NILS program (www.slu.se/nils; Ståhl et al. 2011) – National Inventory of Landscapes in Sweden – is a unique creation in Europe and internationally. It is developed to monitor and analyze conditions and trends in landscape biodiversity and land use on all terrestrial habitats across the Swedish land base. In its original set up, NILS data and analyses are applicable on national or sub-national levels. County-level approaches are currently being developed to assist in county-level environmental frameworks. In addition, new innovative local approaches are being explored with the objective to deliver monitoring data and analyses into landscape planning modules. Such an approach requires methodological processing and development as well as conceptual and operational input

about the specific needs and premises among land use actors and decision makers. NILS was launched in 2003 and despite its short history of existence NILS has developed an infrastructure that is applicable for many different purposes. NILS has the base funding from the *Swedish Environmental Protection Agency*.

9.2. Background to and main direction of the NILS program

The concept of biodiversity has evolved as an issue on the global agenda of environmental concern with the 2010 International Year of Biodiversity as a major event (Larigauderie & Mooney 2010). From a focus on systematic and taxonomy in the 1970s and 1980s the concept presently is linked to ecosystem functioning, ecosystem services and human well-being, but also to economic and environmental consequences of decision making and other practical implementation processes (Dirzo & Loreau 2005). As reflected by the Convention of Biological Diversity (CBD), several EU agreements (United Nations 1992, UNEP 1993, Council of Europe 2000; European Commission 2008) as well as national frameworks (e.g., Ministry of the Environment Sweden 2004), maintained biological diversity is widely acknowledged as a mainstream objective. Since the Rio Summit (Council of Europe 2000), massive work has been conducted to define the concept of biodiversity, to develop appropriate indicators, and to develop suitable monitoring techniques (e.g., Geoghegan et al. 1997, Yli-Viikari et al. 2002, Ståhl et al 2011).

The NILS program is directed towards *biodiversity monitoring* (Ståhl et al 2011). To monitor biodiversity, however, there is a need for methods and indicators that address compositional, structural and functional attributes at different spatial and temporal scales (Noss 1990). Owing to the large number of species in nature-landscapes, and the fact that many species occur sparsely in nature, most species are difficult to monitor and assess with adequate accuracy. Assessment of habitats and substrates rather than of individual species is often a more practical approach. Although the NILS program has species monitoring, it does not include complete species inventories, but rather document species that are indicators of habitats and habitat change, and that are fairly easy to recognize. The NILS species list currently (2012) consists of slightly over 200 different plant species. Thus, NILS biodiversity orientation is on *habitats / plant community / biotope level* rather than on species level.

In land use and management it is evident that the complex *nature and function of landscapes* (for defining landscapes see, e.g., the European Landscape Convention; Council of Europe 2006) has to be regarded and that spatial as well as temporal perspectives on landscape use have to be approached. This is in particular true under changing premises, caused by climate change, new land use policies or changing demands on natural resources. From a biophysical point of view the landscape is a continuum of land cover types, e.g. forest, agriculture land and water bodies, and the transitions between them. In a temporal context the current land cover type or land use on any point in a landscape, is a temporary phase that will change over time in response to natural changes or anthropogenic influence. Monitoring large geographical areas and various types of nature is demanding, however. There is a need to combine different monitoring techniques and methods to obtain enough precision in critical variables. The NILS approach is to combine field inventory in circular sample plots and along line transects for details and precision, with remote sensing for the complementary landscape context and configuration. Thus, the intention is to be able to put each single point, e.g., the cover of a species or a species group recorded in a small circular sample plot or a forest edge recorded along a line transects, in the context of its landscape. The *multi-scale monitoring approach covering all terrestrial habitats* is fundamental in the NILS program.

In 1999 the Swedish Government adopted 16 broad *Environmental Quality Objectives* as a framework for efforts to achieve sustainable land use and land management on national level (Ministry of the Environment Sweden 2001). Those objectives are hierarchically constructed, with sub-targets and threshold values for continuous evaluation and assessment. A central purpose with the NILS program is to document conditions and changes in terrestrial habitats and provide data into measuring the realization of these environmental quality objectives. As well it is understood that experiences in the NILS program will be used to refine and improve the objectives and their sub-targets with respect to arrangement, definition and threshold values. NILS currently provides data and information for the evaluation of existing sub-targets as well as for the formulation of new targets within ten out of the 16 objectives, to a major or minor degree.

- Thriving wetlands
- Sustainable forests
- A varied agricultural landscape
- A magnificent mountain landscape
- A good built environment
- Reduced climate impact
- Zero eutrophication
- Flourishing lakes and streams
- A balanced marine environment, flourishing coastal areas and archipelagos
- A rich diversity of plant and animal life

The broad application of NILS data, monitoring methods and analyses is a necessity to secure relevance and impact. As well, close cooperation with stakeholders (academia, decision makers, public authorities, land managers) are of fundamental importance. In summary the following main NILS directions can be defined (Svensson 2009):

- Documentation, assessment and development of the Swedish Environmental Quality Objectives
- Input to national policy development
- Background data for international reporting to various conventions and frameworks
- General input into strategic planning and large-scale landscape planning
- Cooperation with other national and international monitoring programs
- Platform for applied research on data, analyses, technology and methodology

9.3. Other fundamentals in the NILS monitoring program

The specific design and methodological approaches within the NILS program have been presented in Ståhl et al. (2011), and are also available at <http://slu.se/nils>. For the purpose of this contribution, however, the following fundamentals of the NILS program are outlined (Fig. 1):

1. The basic design is a random systematic grid of 631 permanent 5x5 km squares. All terrestrial habitats are represented; forests, agriculture land, wetlands and peatlands, alpine environments, shorelines and urban areas. It takes five year to complete one inventory rotation. Hence, a random subset of 1/5 of the squares is monitored each year.
2. Two parallel and integrated inventories are conducted; field inventory in plots and along inventory lines, and interpretation of color infrared (CIR) aerial photos. This allows combination of different

types and orders of data, to specifically address current problems and challenges in landscape analyses.

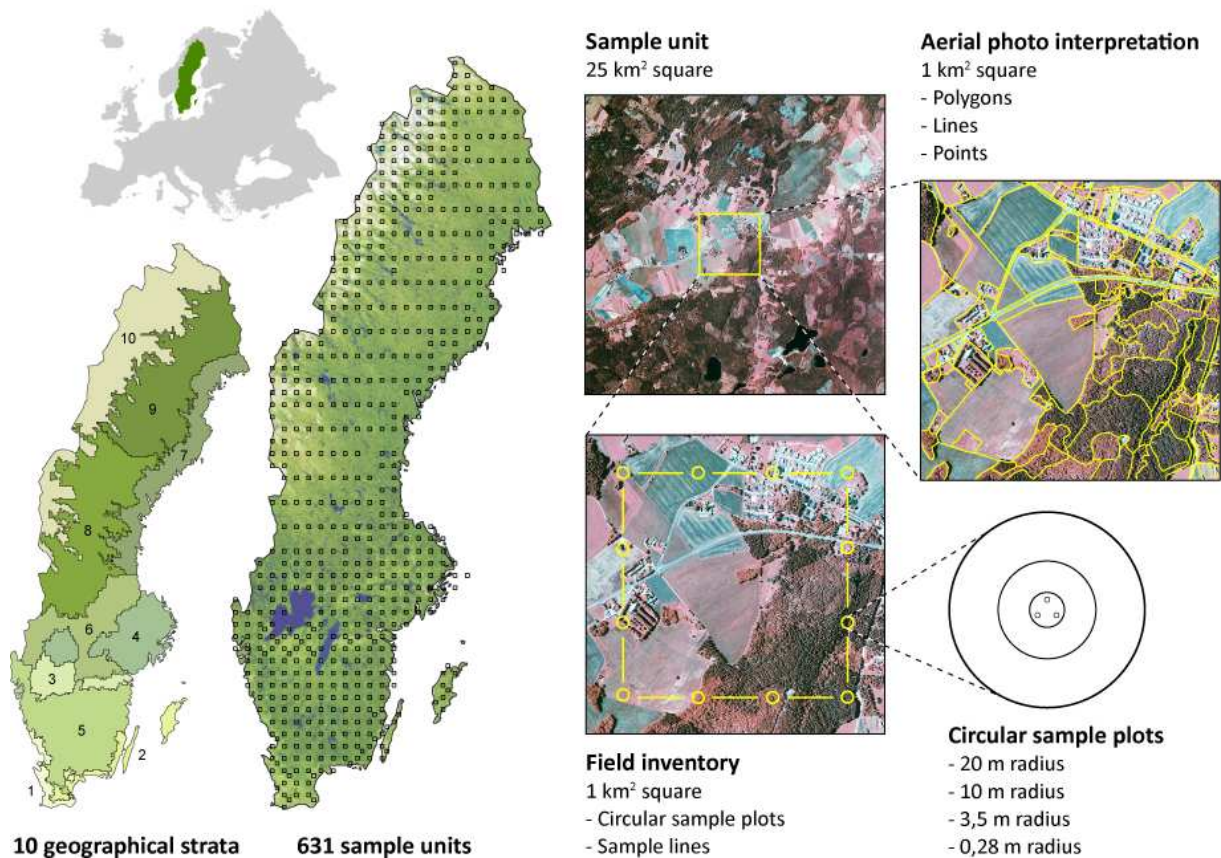


Figure 1. NILS sample design (Ståhl et al. 2011)

3. So far there has been a focus on the central 1x1 km square within the 5x5 km square. There, field inventory is conducted on a set of 12 circular sample plots and 12 sample lines to capture areal (e.g., cover of trees or bottom layer mosses) and linear features (e.g., forest edges, streams or roads).
4. The plots are composed of a sequence of circular plots with different radius; 20 m, 10 m, 3.5 m and 0.28 m, where different sets of variables are documented, representing different resolution in precision and accuracy.
5. The aerial photo interpretation generates polygon (areas down to a minimum of 0.1 ha), linear and point information. As for the field inventory there is so far a focus on the central 1x1 km square.
6. Taken together the field inventory and aerial photo interpretation allows for analyses on various geographic scales, from large-scale landscape analyses (25 km²) on the spatial configuration of habitats, land use and other important landscape information, to occurrence of specific species on point scale (0.25 m²).
7. NILS employs in total 356 variables in the current monitoring protocol, whereof 269 in the field inventory and 87 in the aerial photo interpretation. Some variables are in common for the field inventory and the aerial photo interpretation to allow compilation of the two sets of data into one common data set and, hence, co-analyses of geographically determined data. An example of variables in the aerial photo interpretation is presented in figure 2. The box outlines the set of variables that are of importance on wetland and peatland habitats (Jeglum et al. 2011).

8. Variables are documented without pre-classification (*a posteriori*). This allows for problem-oriented analyses of data, since variables can be combined and re-combined perpetually according to various classifications to specifically address those questions that are in focus. The *a posteriori* approach also allows analyses across land cover types, if, e.g., land use scenarios are being created based on forest and agriculture habitats in combination.

Variables in NILS relevant to mires		
Hydrotopographic Mire Type Fen <ul style="list-style-type: none"> • Flat or slightly sloping fen • Limnogenous fen • Sloping fen • Hill fen • String flank fen • Fen, strongly influenced Bog <ul style="list-style-type: none"> • Plateau bog • Dome bog • Sloping bog • Flat or weakly raised bog • Net bog • Blanket bog • Bog strongly disturbed Mixed mire <ul style="list-style-type: none"> • String mixed mire • Mosaic mixed mire • Palsa mire • Mire, weakly raised bog or fen 	Microtopographic Series <ul style="list-style-type: none"> • Hummock (Dwarf shrub dominated) • Lawn • Carpet • Mud-bottom • Flank pools • Bog pools • Marsh fen, Shore fen Field / Bottom Layer <ul style="list-style-type: none"> • Field cover/bottom layer missing • Graminoid- or herb-dominated • Graminoid-dwarf shrub type • Dwarf shrub dominated • Lichen-dwarf shrub type • Lichen type • Belt formation of e.g. <i>Phragmites</i> spp. • Tall sedge • Low sedge • Sphagnum dominated • Other moss dominated • Logging residues • Field layer/Ground layer in shadow 	Trees and Shrubs <ul style="list-style-type: none"> • Tree cover in 4 types • Tree height • The pattern of tree distribution in 8 subclasses • Tree species mixture in 6 subclasses • Height dispersal in 4 subclasses • Occurrence of broadly crowned trees, in percentage • Occurrence of shrubs in 4 subclasses • Shrub cover, deciduous and needle-leaved. • The pattern of shrub distribution in 8 subclasses

Figure 2. NILS aerial photo interpretation variables relevant to wetlands and peatlands (Jeglum et al. 2011).

9.4. The NILS process so far

Preparation phase

The NILS program became operational in 2003. The launch was forgone by intensive preparation work since the end of the 1990th. An important preparation step was an *information analysis* (Esseen et al. 2004) which became the basis for the variables included in the monitoring protocol. About 90 persons at public authorities, organizations, sector companies, research institutes and universities were interviewed. Questions addressed included type of variables, purposes of the monitoring, type of analyses, habitats, reporting procedures, temporal resolution and statistical properties.

The first inventory rotation was completed in 2007. The focus was to a large degree on *training and building up experiences* on what works and what delivers good data. Methods were refined and routines were set. Since we have a maximum of about 4 months for the *field inventory*, and less than half of that in the mountains, there had to be a focus on the field work. The aerial photo interpretation methodology was still under development.

First NILS rotation phase, 2003 – 2007

An important step during the first rotation was the inclusion of a second national-scale inventory into the NILS monitoring system. In cooperation with the *Swedish Board of Agriculture* a *monitoring of semi-natural grasslands and pastures* was launched in 2006. This inventory is partly conducted by the NILS field inventory teams, on original NILS plots and on additional plots in such habitat types within the NILS 5x5 km squares. Those plots are the same type of circular plots as in the original NILS methodology, but with another six of the smallest circular plots (Fig. 3) to obtain enough data on field- and bottom layer vegetation. As well, further plant species are included on top of the standard NILS species list. Apart from that, the same methodology and procedures are used. Another part of this commission is a special inventory of butterflies, bumble bees and large solitary trees, i.e. key indicators of a maintained open and managed agricultural landscape. This inventory is conducted separately from the original NILS field inventory, with one-person field teams specifically trained for this purpose, and partly with a separate field support system. The “NILS butterfly monitoring” is also a national-scale system. Furthermore, a designated *reporting on scattered biodiversity habitats in the agricultural landscape* was included, based on data from NILS aerial photo interpretation, as part of the cooperation with the Swedish Board of Agriculture.

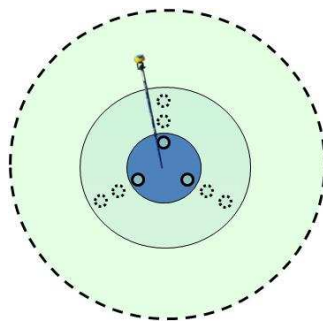


Figure 3. Six additional small circular sample plots are monitored in the monitoring of semi-natural grasslands and pastures.

Second NILS rotation phase, 2008 - 2012

The second inventory rotation started in 2008. The field inventory continued with addition fine tuning in methodology and logistics. The *aerial photo interpretation* was further processed with test and new projects to explore complementary and alternative approaches. With a complete national-level data set sample it was also possible to approach and develop the *NILS data management system* with secured and improved routines and techniques for high quality data processing from input of values in field computers and aerial photo data templates, via automatic quality assurance and error check applications, to compilation and secure server storage of data that are ready for analyses. Furthermore, development of routines and applications for *analyses and reporting / communication* was initiated, including a concept for a web portal for NILS data. The intention is that data, aggregated data in figures and tables, statistical properties of the data, metadata and other information should be fully available for stakeholders. Hence, the communication routines and web portal functions are under development together with a group of data stakeholders who assist with advices and ideas.

Two more additional monitoring protocols were included during the second NILS inventory rotation. In Sweden, environmental reporting is done at both national and county levels. Hence, in 2009 a number of counties initiated a *regional environmental monitoring system* with more specific monitoring according to their reporting protocols (Fig. 4). As for 2012, NILS cooperates with nine counties under

the leadership of *Örebro County Administrative Board*, central Sweden, in monitoring of wetlands and peatlands, grasslands and pastures, and of scattered habitats in the agricultural landscape. Part of the inventory is done by the original NILS field teams as with the Swedish Board of Agriculture commission, and part (scattered habitats) of is done with a separate system. The normal procedure is that the inventory persons in the NILS butterfly monitoring completes their seasonal employment with this scattered habitat monitoring during late summer. The field inventory is preceded by directed aerial photo interpretation of grid points in focal habitat types within the NILS 5x5 km squares.

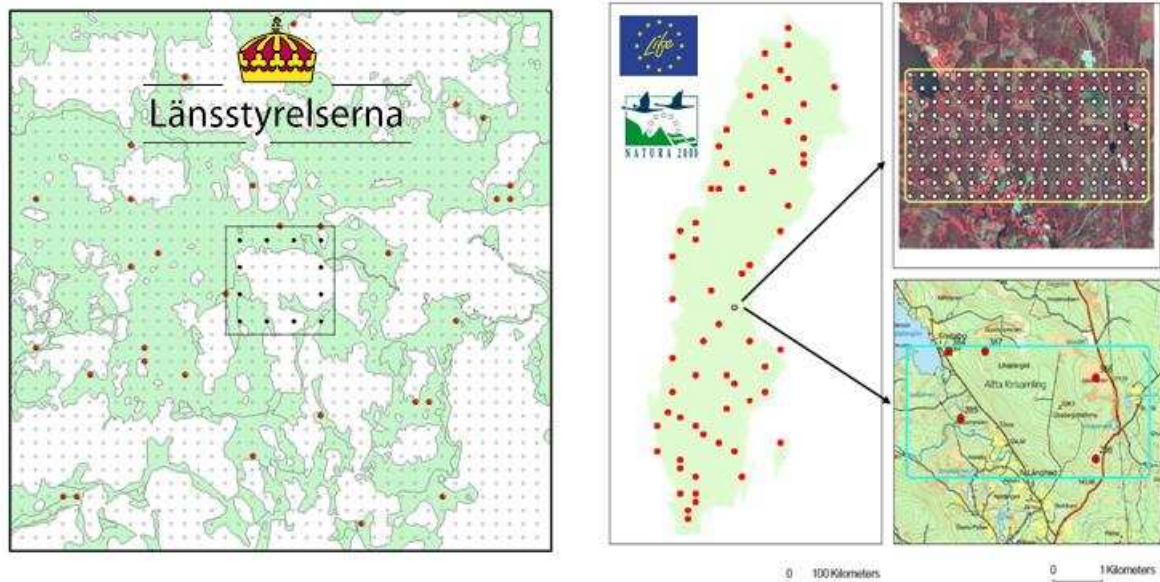


Figure 4. Sample for the regional environmental monitoring (left) in on 5x5 NILS square. Grid-based sample plots are selected for wetland and grassland habitats, specifically. Sample system for MOTH monitoring (right). A grid-based sample on a segment of a NILS 5x5 km square.

The second additional monitoring protocol added during the second NILS rotation phase was the *classification of habitats under the EU Species and Habitats Directive* (European Commission 2008). This is done as a complement to the NILS field inventory, and to the Swedish National Forest Inventory (Axelsson et al. 2010), on commission from the *Swedish Environmental Protection Agency*, and was initiated in 2008. Furthermore, a special project aiming for developing methods for monitoring of habitats was constructed – MOTH, *Demonstration of an Integrated North-European System for Monitoring Terrestrial Habitats* (LIFE08 NAT/S/000264), 2010 - 2014. The MOTH field inventory was formally launched in 2010, and as with the Regional Environmental Monitoring System the field inventory is preceded by directed aerial photo interpretation of grid points within the NILS 5x5 km squares (Fig. 4). The MOTH field monitoring under this project is at its maximum during field seasons 2012 and 213, where it is expected to have a size equal to the original NILS monitoring.

9.5. Inventory routes – building data input capacity

Field inventory

The field inventory is conducted in the 12 circular sample plots and inventory lines within the 1x1 km squares. All plots are visited in the field, except those that are situated in arable fields, in water, in built-up areas, or where it is not not legal or secure (e.g., steep areas) to work. Some basic variables are always recorded, however, i.e. land use and type of land cover, either from a distance or from maps and other additional databases. The line intersect sampling is specific for the NILS program.

Those data have several possible applications and can be combined with data from the aerial photo interpretation to estimate length of objects by type of land cover or habitat for assessments on the spatial characteristics of a landscape and how that changes over time and with land use. Another application is to document changes in the quality of linear elements in a landscape. For example, changes in vegetation cover and management on vegetation strips and stone walls provide important information for managing biodiversity associated with these objects. Our initial results from analyses on the line intersect sampling clearly show that linear landscape objects and fragmented habitats are important features of the Swedish landscape and contribute significantly to landscape diversity (Svensson 2009). Details on the field inventory methodology is presented in Ståhl et al. (2011).

The NILS field inventory system has developed into a well-working infrastructure (cf. Gallegos-Torrell & Glimskär 2009). Over the period of a year, the following main phases can be distinguished:

- January – March: Continued management of data; delivery of a one-year quality-checked field database to the central NILS database; Evaluation and improvement of all components in the field inventory on yearly basis.
- February – March: Preparation and planning of the field inventory season and field staff courses; preparation of equipment, including technical equipment, inventory manuals and other printed material; complementary purchase of equipment.
- February – April: Advertising field inventory positions; Recruiting field staff; Employment procedures. All field staff is employed on a season contract. Reserves, usually 2 persons, are employed by hours and are present at the courses.
- April: Final preparations.
- May: A two-week intensive course is given in south Sweden to prepare the field staff with necessary knowledge and skills; A one-week start-inventory period in south Sweden, with real inventory work, and with mentors and supervisors available.
- May – September/October: Fully equipped two-person teams work in their part of Sweden. Accommodation and travels are included on top of the salary, as well as per diems. Living in tents is needed in remote areas. Helicopter transportations are used when necessary.
- June/July: Two 3-day calibration courses are arranged – one in south Sweden and one in the mountain region – to further ensure that the data is recorded with as high and even quality as possible.
- September/October: The field inventory is completed.
- May to September/October: Support is available on distance; Field visits are conducted by office staff; Data is sent and received by a server on daily basis, if possible with respect to connections; Data is automatically checked for errors; Daily reports are sent back to the field teams.
- October to December: Equipment control and repair / replacement; Data management and data processing.

The field inventory is a central component in the NILS and added monitoring programs. It is currently also the most expensive separate part of the program. An average field person has a bachelor degree in biology. Hence, they have a good theoretical background, which is needed to secure good data. Table 1 shows the total number of field persons employed from 2003 to 2011; 176, whereof about 60% women. A total of 81 different persons have had season employment up to 2011. The variation in numbers of employees depends on adjustments of the methodology and on the fact that other inventories have been added to the NILS program. The drastic decrease between 2003 and 2004 depends on an initial reduction from 16 circular sample plots and 16 inventory lines, to 12, respectively. The increase in 2010, and 2011 in particular, depends on the expansion of the MOTH program. For 2012 it is estimated that the share of the original NILS program will be about 40% of the complete field inventory, whereas the MOTH share will be as high or even slightly higher, that the

share of the inventory of semi-natural grasslands and pastures will be about 15%, and finally that the share of the regional environmental monitoring system will be 5-10%.

Year	No field persons	whereof women	whereof recruits
2003	22	12	22
2004	15	9	3
2005	14	10	5
2006	19	12	7
2007	17	10	5
2008	18	9	6
2009	19	11	8
2010	22	12	6
2011	30	18	19
	176	104	81

Table 1. Number of field inventory persons 2003 to 2011, whereof women and new recruits.

Office staff is responsible for support, preparation and other work directly related to the field inventory, including data management. For 2012 it is estimated that the total amount of work input equals to close to five full-time positions.

Note that the text and figures above does not include NILS butterfly monitoring or the special inventory of scattered habitats in the agricultural landscape within the county-level regional environmental monitoring.

Inventory of aerial photos

Interpretation of aerial photos was chosen as the NILS remote sensing approach to obtain landscape context data with enough resolution, as a complement to field data (Ståhl et al. 2011, Jeglum et al. 2011, Ramezani et al. 2011). There are obvious advantages is using aerial photos in recording detailed data on the spatial structure of landscapes and the configuration of land cover types, compared with other remote sensing techniques (e.g., Allard 2003; Ihse 2007; Reese 2011). The interpretation is based on Color Infra Red (CIR) aerial photographs taken from 4800 m elevation, which provides 0.5 m resolution on ground level (the methods are described in detail in Allard, 2003). The technology is based on viewing the digital images in stereo in a computer-based photogrammetric system.

Field-based calibration of interpretations, inter-calibration of personnel, and continuous development of calibration tools and routines are employed to reduce the variation between interpreters. External databases are integrated when supplementary data are needed to maintain data quality standards, and a decision tree has been developed to secure that the polygon delineation is as interpreter-independent as possible. In total 67 variables are recorded for the polygons. When the objects are too small in size for being delineated as polygons (0.1 ha), important features are mapped as linear (e.g., stone walls or ditches) or point objects (e.g., ponds or boulder mounds, with 10 variables, respectively). The detailed polygon interpretation of the 1x1 km square is extended 50 m outside the borders of the square to avoid edge effects.

Other important aspects of the NILS aerial photo system include:

- Aerial photos were purchased through procurement, to ensure that photos of the NILS squares were taken \pm one year around the field inventory year, to maintain temporal integrity between field data and data recorded in the photo interpretation.
- The logistical processes and preparation of aerial photos into stereo-models were supervised, and partly done, by NILS staff.
- The interpretation and inventory methodology, included definitions, variable flows, techniques, database templates, etc., has been developed parallel to operational interpretation.
- Computer systems and other technological components needed for the interpretation has been tested, evaluated and developed.
- The interpretation was done by office staff, specialized on interpretation or interpretation in combination with other duties.
- Pilot analyses and methodological development has been done continuously to improve the approach.
- Alternative and complementary remote sensing techniques have been investigated to some extent.

In the original NILS design it was planned that pre-interpreted aerial photos should be used in the field inventory. Owing to the challenging approach, as indicated above, the aerial photo interpretation currently is lagged compared to the field inventory. An evident conclusion is that the resources needed have been under estimated and that the original expectations and premises have not fully been accomplished. During 2010 and 2011, however, when the interpretation of the fourth year (2006) aerial photos was completed, the outcome was satisfying and representative for a well-working NILS aerial photo interpretation according to the original approach. In short the following data can be presented for this period:

- A total of 108 NILS squares (1.1x1.1 km) with completed polygon, line and point sampling.
- Eight different persons have been involved, some with minor time for interpretation.
- On average it took 25 to 28 hours effective interpretation per square.

Facts on the number of polygons, line- and point objects are presented in figure 5. On average there were 81, 19 and 26 polygons, linear and point objects, but with a large variation among squares.

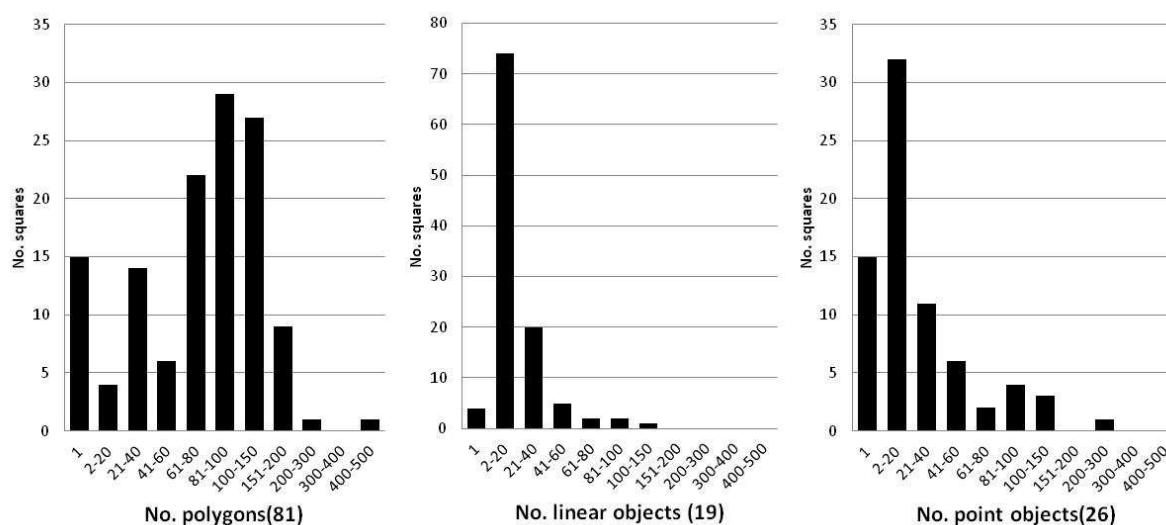


Figure 5. Number of polygons, linear and point objects in 108 NILS squares, field inventory year 2006.

Future prospects

Data management in NILS is currently under development, and partly already implemented. This includes automatic data check applications as early as possible the data processing system – e.g., already in the field computers and/or by immediate reports to the field crew after their data transfer to the central server. Furthermore, the relational database structure has been established, data are under correction and quality check, and analytical and statistical applications are being prepared and introduced as various analyses have been done or are under way. Hence, data management and data analysis systems are in progress and will gradually evolve following further complements of data that allows temporal assessments. NILS is now approaching the third rotation and will continue to develop routines for analyses, reporting procedures, communication and implementations of data and methodology.

The core of the monitoring system is set, but additional fine tuning continues perpetually to produce a monitoring system with variables and methods that are clearly defined, relevant, accountable, and that timely can provide statistically reliable data on landscape biodiversity on a national scale. In this process the attention is now being directed towards evaluating which features of the program that are successful, and which have to be further developed. Currently, a certain emphasis is placed on the aerial photo inventory and, generally, remote sensing as a monitoring technique. A variety of remote sensing approaches is or will become available in the near future, e.g., laser-scanned data for image matching to extract tree height and detailed ground topography. Thereby, a close connection between monitoring and research needs to be maintained. With the rapid technological development in hand-held filed computers NILS is currently also revising the variable content and flow in the field inventory system along side with testing new hardware with better software and visual capacity.

The inclusion of the MOTH and the Regional Environmental Monitoring programs adds promising components. Both programs are oriented towards more rare habitats and phenomena, whereas the original NILS is oriented towards general conditions and changes, by introducing methods for directed sampling. In the first program it concerns habitats in the EU Species and Habitats Directive, and in the second program certain habitats types of special interest. Future plans include also Woodland Key Habitats (Gustafsson et al. 1999) and the possibilities to combine directed “search and find” inventories with national monitoring to better obtain statistically relevant and representative information.

9.6. Conclusions and lessons learned

NILS provides a national level set of data for land cover and biodiversity analysis on different geographic scales. Experiences clearly show that the NILS monitoring infrastructure is attractive to other initiatives and provides a platform for various approaches on top of the original set up. Some 15 (large and small, short- and long-term) parallel and integrated development projects has been developed since 2003, and about as many side projects (large and small, short- and long-term) has been organized by other organizations in connection with NILS. Cooperation is established with several national and regional public authorities, and research projects are conducted based on NILS data, technology and methodology. The inherent flexibility in the NILS sample design and methodological setup is an obvious strength both in term of its applicability and usefulness for other environmental monitoring and research initiatives, and in terms of its capacity to add and make use of supplementary information, which is certainly of critical value (cf. Bunce et al. 2008). Hence, externally generated information, i.e. other monitoring programs or other databases, can be used to deepen and broadening the NILS scope, just as NILS can provide background data for other purposes.

The need to apply landscape data and a landscape perspective in biodiversity issues or other central environmental issues, e.g., on ecosystem resilience, land use sustainability, is undisputed (e.g., Ahlqvist 2008; Wiens 2008). As is adjustments in landscape monitoring to the need for reliable data to be entered into international and national frameworks and conventions. In this perspective the connection to research is critical to validate the monitoring approach and the data, and to investigate further the applicability of the monitoring program (*cf.* Lovett et al. 2007). Despite fundamental advances in landscape ecology, the routes to policy and decision making is still undeveloped, however (Bunce et al. 2008; Nassauer and Opdam 2008). In particular under a climate change scenario, cause and effect analysis on empirically derived landscape data is central to evaluate ecosystem response and processes (e.g., Metzger 2008; Shao and Wu 2008). Experiences in NILS show, however, that landscape-based monitoring can provide essential data into research, and information to decision makers and other stakeholder on current issues on land use and landscape management.

Also new data stakeholders appear and new types of data are demanded, which calls for a need to secure flexibility and capacity to add new variables and inventories on top of the core, long term monitoring protocol. In a societal context it is also obvious that the use of monitoring data becomes more diversified; e.g., as baseline information for policy and governance, for strategic scenario/impact analyses of land use, and for large-scale landscape planning modules. This also implies a need for effective and immediate cause-and-effect analyses and, hence, a close cooperation with the research community. Experiences indicate that the NILS infrastructure allows for inclusion of parallel and supplementary inventories and projects on national and sub-national scale. In a pan-national perspective it is also evident that there is a need to harmonize existing environmental monitoring programs and create common monitoring protocols and analyses and reporting procedures.

As the first NILS inventory rotation was completed in 2007, a national set of data for all terrestrial habitats in Sweden became available. With a second inventory rotation completed in 2012 the database will become increasingly better in quality and more available and useful to data stakeholders. Some essential lessons learned from a decade of experiences with the NILS programs are:

- NILS is still developing but offers and promising infrastructure for pan-national biodiversity monitoring approaches. It is evident that there is a need to harmonize existing programs and create common protocols to capture similar or comparable data using transparent methodology.
- There is a need for an intrinsic capacity to adjust the monitoring protocol to current issues and new demanded data, and, hence, to add variables or inventories on top of the core monitoring methodology which form the basis in the long-term monitoring program. New variables and inventories can be included over short- or long-term depending on the specific issue to address.
- A long-term core program with consistent methodology will be extremely important for many points of view and purposes.
- It takes time to develop a national-level biodiversity monitoring program, and it has to take time since there are many different stakeholders and potential implementations, which are not always possible to foresee. As well, techniques, methods, analyses application and implementation options perpetually change, which has to be acknowledged.
- A combination of detailed field inventory with remote sensing is needed to operate on multiple geographical scales. A large scale – landscape scale – spatio-temporal setting is fundamental to address environmental concerns. New remote sensing techniques has to be taken into account, tested and potentially and included.
- Research components should be linked to the monitoring, to secure appropriate cause-and-effect analyses and to secure data quality end methodological / statistical properties.

- Funding is critical; long-term funding commitments are necessary.
- Communication is critical; stakeholders that are expected to benefit from and use the monitoring, should be actively involved.

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