Landscape scale biodiversity assessment: the problem of scaling

Report of an electronic conference, March 2005

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Preface

Research on biodiversity is essential to help the European Union and EU Member States to implement the Convention on Biological Diversity as well as reach the target of halting the loss of biodiversity in Europe by 2010.

The need for co-ordination between researchers, the policy-makers that need research results and the organisations that fund research is reflected in the aims of the BioPlatform network. BioPlatform is a network of scientists and policy makers that work in different fields of biodiversity and aims at improving the effectiveness and relevance of European biodiversity research, fulfilling functions that provide significant components of a European Research Area. BioPlatform supports the existing “European Platform for Biodiversity Research Strategy” (EPBRS), a forum of scientists and policy makers representing the EU countries, whose aims are to promote discussion of EU biodiversity research strategies and priorities, exchange of information on national biodiversity activities and the dissemination of current best practices and information regarding the scientific understanding of biodiversity conservation.

This is a report of the BioPlatform E-conference entitled “Landscape scale biodiversity assessment: the problem of scaling” preceding the EPBRS meeting to be held in Budapest, Hungary from the 31st March to the 4th April 2005.

Summary

Both organising institutions, the Institute of Ecology and Botany and the Ministry for Environment and Water, have experience of habitat level biodiversity assessment and mapping through the development and operation of the Hungarian Biodiversity Monitoring System. During the organisation of the EPBRS meeting in Hungary, scaling emerged as an important issue to address among both natural and social scientists. Scaling, as a scientific problem has not been touched upon so far at EPBRS meetings, so the organisers hope to awaken the interest of the scientific community by selecting this topic as the basis of the EPBRS meeting in Hungary.

The aim of this EPBRS meeting is to focus on the ecological research problems raised by the different scales of biodiversity (with an emphasis on landscape scale), and the interaction between policy and social aspects with biodiversity conservation at different scales.

In order to reach those aims, the e-conference had four sessions running in parallel, on the following themes:

- Session I- Scaling problems in biodiversity assessment, chaired by Ariel Bergamini, Christoph Scheidegger and Lisandro Benedetti-Cecchi
- Session II- Biological scales and conservation, chaired by András Báldi and Keith Hiscock
- Session III- Political and economic scales in relation to biodiversity, chaired by Sybille van den Hove, Thomas Koetz and Ekko van Ierland
- Session IV- Integrating ecological and social scales, chaired by György Pataki, András Lányi and Ekko van Ierland.

The overall conclusions from those sessions are as follows:

Session I- Scaling problems in biodiversity assessment

In total, this session had 55 contributions (16 keynotes and 39 other contributions). The main topics of this session were scaling up and indicators.

Scaling-up: This is a topic of particular interest and several keynotes and contributions addressed it. Because we can never accomplish a complete assessment of biodiversity of a large area such as a landscape or a region, we will always have to scale-up from samples to the entire area. Approaches include linking remotely sensed data with field investigations, models using the detection probabilities of species in small plots to extrapolate to a larger region, and the development of scaling functions. However, there are some specific problems. For example, there is a lack of rigorous testing of the accuracy of the first approach. For the second approach a major problem seems to be that we can estimate species richness in different habitat types, but we have problems combining these estimates for a landscape estimate because of undetected (unidentified) species. Furthermore, contributors identified the potential of viewing ecosystems as self-organizing, i.e. as emergent systems, which change when crossing an emergence boundary. However, a lot of ecologists are not very familiar with this topic and, thus, for them the potential is not yet very obvious.

Indicators: Indicators are still discussed controversially and it seems clear that all indicators have some shortcomings such as scale-dependency of their strength as indicators. Regarding taxonomic scales or hierarchies, the morphospecies concept in particular seems to be controversial as was indicated by several critical contributions. The use of species lists was also questioned, but again, opinions were not uniform. One of the main problems identified with species lists is that all species are regarded as equal, which, of course, they are not. Moreover, they may not be very useful for short time scales; abundance measures were suggested as more suitable for short time scales. Another problem identified is that species lists are very time-consuming and not very cost-effective to produce. The non-congruence of richness patterns of rare and common species, which was emphasized in one keynote, seems to be of particular importance in biodiversity assessment. For example, if we identify drivers of species richness based on all species, these drivers are mainly relevant for the common species. Furthermore, we don't know anything about how congruence of richness patterns of rare and common species changes with spatial scale. Regarding genetic diversity and indicators we hardly know anything. While there has been a lot of work on relationships between various indicators and species richness, there is hardly any work done on indicators on genetic diversity. Unfortunately the EASAC guide to biodiversity indicators (http://www.easac.com) was published towards the end of the e-conference. This could have been a very good basis for our discussions.

There were other topics, which were not rigorously discussed, but are nevertheless important in this discussion:

1. The selection of conservation-relevant areas (in terms of biodiversity) is scale (or grain) dependent.
2. The trade-off between geographical precision and taxonomical precision.
3. Rarefaction was discussed as the method for quantification of biodiversity patterns at the landscape scale since most biodiversity indices are strongly sample-size dependent. Furthermore, a profound distinction was identified between species density and species richness. These two metrics may yield completely different answers to the same question.
4. Conclusions or biodiversity pattern detected critically depend on the design of the study, i.e. how sampling units are spaced (i.e. coverage and distance apart) and placed (i.e. simple random sampling v. stratified random sampling).
5. A lack of long-term monitoring data exist even in Europe. It is thus not easy to differentiate between population fluctuations and real trends.
6. The usefulness of methods or indicators depends on the time scales considered. While grid data may be useful to monitor species richness over centuries, abundance based measures should be promoted when considering shorter time-scales.
7. Monitoring schemes should not be set up unless we know how to relate observed changes to ecological processes and their drivers.
8. Additive partitioning of gamma diversity may help to identify sets of habitat areas that comprise the largest beta diversity. These areas deserve special attention in formulating land-use practices or in prioritizing areas for protection. Additive partitions of diversity may also inform us about sampling designs or monitoring strategies by identifying the sampling scales that contribute most to beta diversity. Temporal partition of diversity is also possible and may be important in monitoring biodiversity.
9. It was emphasized that we should start using the data we have already collected. This issue is not related to scaling problems in particular but still very important (as can also be seen in the EASAC guide to biodiversity indicators).
10. There is a lack of basic taxonomic agreement in various groups. It is of utmost importance to find a consensus and to compile full synonymic checklists. Even for

vascular plants, which belong to the best known groups in Europe there is no actual checklist and the Flora Europaea is somewhat out of date. If we want to combine lists of species from different regions, agreement on species names or least full synonymic checklists are essential.

Session II- Biological scales and conservation

Session II received a total of 42 contributions (12 keynotes and 30 other contributions). From a marine perspective, many issues of identifying marine protected areas of different sizes and incorporating entities from species to landscapes are well addressed and have not changed much, in terms of criteria used, for many years (although repackaging occurs). What has changed is the ability to use structured marine biological information. We have a directory of marine species (the European Register of Marine Species) and we have a biotopes classification in the European Nature Information System produced by marine biologists. We have criteria (in the UK at least) from which to identify rare and scarce species. OSPAR have identified workable 'threat' criteria for marine species and habitats.

From a terrestrial perspective, there seems to be more information, and at least some of it is easily available (e.g. bird and plant atlases). However, the research questions on scale issues are far from being well understood. Contributions identified key topics and many important questions.

Following through some of the discussion, it is clear that some issues of both marine and terrestrial conservation are scale independent. They include the importance of good stewardship wherever rare, scarce, fragile, aesthetically, culturally or recreationally important species, landscapes or habitats are present.

A few research questions that have emerged from discussions are:

1. By protecting a full range of marine habitats in strict MPAs, would we protect the full range (or what proportion would we protect) of marine species?
2. Do MPAs do the job or should we be working much harder on a ´good stewardship´ approach - perhaps exemplified in the Water Framework Directive?
3. What habitats and species are most at risk from human activities and will need strict protection? (We have new scientifically based ways of assessing sensitivity - see http://www.marlin.ac.uk).
4. Are the consequences for loss of biodiversity different for different biotopes in terms of functionality and long-term survival.
5. Can we use coarse levels of taxonomic discrimination in a meaningful way to identify biodiversity changes, biodiversity hotspots etc in the sea - and then manage human activities to maintain that biodiversity.

Additional research topics from the terrestrial perspective are:

1. Identify threshold values, address why local population catastrophes have drastic effects on larger spatial scales ("transfer of catastrophe across spatial scales"), and find the cut-off values in reserve designs (subdivision vs. single large).
2. What is the time scale of the time delayed extinction due to habitat loss (scale dependence of the extinction debt)?
3. How do spatial responses of metapopulations to disturbances change in relation to spatial scale?
4. Which are the appropriate scales for the conservation of networks of ecological interactions?

Session III- Political and economic scales in relation to biodiversity

The contributions and comments made throughout this session of the e-conference can be grouped in three different areas of concern in relation to scales and biodiversity: (i) economic and value issues; (ii) political and structural issues of multi-level governance; and (iii) more general theoretical issues related to epistemology and how to address integration of knowledge in the context of complex environmental matters. The session received 31 contributions, 10 of which were from keynote contributors.

Before presenting the main conclusions we would like to make a preliminary remark: It seems to us that there has been a lot more focus on the economic rather than the political issues during the e-conference discussion. This raises some interesting questions: Are economic matters so much more important than political issues? Is it because politics might be primarily driven by economic thinking, making it an issue of economics most of the time? Or do we already know enough about political issues and policy processes making them less interesting to discuss? Or in contrast is it maybe that we know so little about them that we just don't really know what to ask? Or is it a result of the group this forum is addressing, lacking participation of people from political sciences and from administrations? Is it a problem of framing the problem to be dealt with by this science-policy interface (EPBRS) - having politics and policy-makers on the one side, ecological and socio-economic sciences on the other, but leaving political sciences aside?

Research questions/needs or problems that were addressed focusing on economic and value issues were:

1. The need for studies clarifying whether 'monetary values of nature are convincing authorities or the public to preserve nature' or if 'using monetary values may lead to crowding out of moral arguments for nature preservation' (Marzetti, Rauschmayer).
2. Scientists can (and should) offer a lot more than one (the monetary) perspective on the value of biodiversity and the ways in which to approach its management. There is the need to identify characteristics of institutional and social systems that take a more multidimensional stance on values (Spash).
3. There is a strong gap in understanding socio-economic and socio-cultural aspects of biodiversity conservation for sustainable development in specific ecosystems, such as mountains (Chettri).
4. In order to avoid misunderstandings and problems when integrating knowledge from the economic and ecological disciplines, research approaches should not discuss the problem of space in an abstract manner but rather start from a particular conservation problem, whose structure will determine the spatial scale for both economic and ecological research (Wätzold).
5. Research is needed to analyse public-private partnerships for biodiversity conservation and management in order to bridge the gap between biodiversity interests and economic interests at the local scale (case studies and research on underlying juridical, political and social issues) (Jansen).
6. Arguing that biodiversity validation and not biodiversity valuation will halt the loss of biodiversity, Jurgen Tack calls for (1) urgent action to increase innovative research in the field of environmental problems, particularly biodiversity related research, to balance technological progress between ecology and economy; and (2) innovative research on a much larger scale.
7. Valuation of biodiversity in a broader sense requires a better understanding of the processes behind the loss of biodiversity and a whole new ecological and economic language which is not mathematical (not in the way we know mathematics today) (Tack).
8. Research needs to address alternative methods for expressing the values people hold with respect to biodiversity and reasons for its preservation (Spash). Such methods and their results depend on context (Rauschmayer), in particular on which stakeholders are involved, and on space and time scales (Sharman).
9. The diversity and role of some organisms and ecological processes which provide important services (such as e.g. the role of decomposer organisms in selfpurification of water) must be studied to estimate their contribution to environmental goods and services (Rossi).
10. Methods are needed that allow for the valuation of whole ecosystems or landscapes, taking account of all socio-ecological elements (Dick).

Research questions or problems that were addressed focusing on political and structural issues of multi-level governance were:

1. Research in political science is needed to understand the dynamics of EU biodiversity policy – unifying the research insights gained from the study of the EU as an international

institution and actor with the knowledge gleaned from the study of the EU's internal system of policy governance (Baker).

2. Stressing the mismatch in several important areas between the international obligations of a country as a Party to the CBD and the different levels of government, Horst Korn concludes that research is needed on possible mismatches between international obligations of a country and its internal structures for implementation, in order to suggest improvements of the system, taking into account the different political structures of a country.
3. Jouni Paavola highlights the need to pay more attention to issues of social justice that arise in multi-level governance, in particular as means to influence the effectiveness of environmental governance solutions which rests on voluntary compliance and legitimacy. He emphasises systematic studies from a social justice viewpoint to draw applicable lessons.
4. Further explore the effect of government interventions in order to reduce adverse impacts on biodiversity – including studies of the potential of decentralization and self-organization (van den Heide).
5. More research is needed into the equity aspects of biodiversity conservation, restoration and management as the heaviest burden of biodiversity conservation tends to be borne by people in rural areas, in the vicinity of protected areas (van den Heide).
6. Open questions have been raised such as what polity-level in the EU multi-level governance is responsible for the definition of a reliable method for monitoring of biodiversity, its realisation, and the policy analysis of conservation efforts (such as the Natura 2000 network) (Jansen).
7. Seeing the black boxing of the social and the political in modelling and mapping decisions as a self-made socio-political trap and a recipe for (mostly bad) surprises, Chimere Diaw argues in favour of the reposition of the people at the heart of a broad range of conservation strategies. Accordingly he stresses the construction of socially-oriented multiple use landscapes at local and regional levels as the key challenge for research and action in development and biodiversity governance.

Research questions or problems that were addressed focusing on general theoretical issues related to epistemology and how to address integration of knowledge in complex environmental matters were:

1. Mario Giampietro calling for research on participatory integrated assessments (1) required for developing a new epistemology, which acknowledges that the observer/narrator is a part of the self-modifying system, (2) that focus on the quality of the process of evaluation (who decides whose perspectives count and how) avoiding collapsing the descriptive with the normative when dealing with sustainability issues – facilitating the necessary abandonment of reductionism.
2. Kate Farrell stresses the need to develop a fourth distinct interdisciplinary nomenclature, ontological and epistemological structure with regard to biodiversity that will articulate into non-mathematical integrative methodologies. In order to develop such methodologies she emphasises (1) the role of time as a complex and scale dependant factor, (2) the importance of knowledge on human cognition, philosophy of the mind, organisational management and group behaviour, and in particular (3) the role of political philosophy. Furthermore, she stresses the need for research into the prevalence of mathematical analytical approaches to overcome scale differences and the ontological and epistemological consequences of this practice.
3. Further research is needed to explore structural issues related to the application of economic and political theory on biodiversity issues on the one hand (what/how, explanations of structure and operation going on at lower levels), and functional issues of the embedding economic and political systems and of potential alternatives (why/how, explanations of finalized functions and purposes, going on at or in relation to the higher level) (Koetz).

Session IV- Integrating ecological and social scales

This session received 30 contributions (14 from keynotes and 16 from other contributors) and has taught us new substantive as well as methodological aspects of the social science of biodiversity and has gained support for the relevance and significance of studying biodiversity as a social, political and economic problem. From a methodological point of view, we have learnt that sociological network analysis has much to offer to ecological analysis. Based on network analysis, the individuals-in-community perspective may be developed further in a quantitative way (Jordán, Balázs). Moreover, the mathematical tools of social choice theory, developed within economics, might also be applied to the ranking of conservation policy options, highlighting their different value judgements (Weikard).

The productive exchange of ideas between natural and social sciences was clearly demonstrated by adopting the concept of metapopulation from population ecology to describing and understanding farmers seed exchange systems, a complex socio-economic phenomena (Van Dusen). Similarly, the term of cultural keystone species represent an invention in terms that has the great advantage to highlight the fundamental co-evolutionary interrelatedness of ecosystems and human cultures (Garibaldi and Turner). Such terms should help us to overcome our tendency for thinking and analysing in dichotomous terms that constrain the advance in interdisciplinary work.

At the interface between natural and social sciences, the concept of ecosystem services has established a productive research field, mostly occupied by researchers identifying themselves with ecological economics. What are the biological or ecological processes and conditions that are related to ecosystem services? What is the relationship between biodiversity and ecosystem services? Gonczlik and Goslee highlight two important questions that need to be addressed from a natural science base: What kind of values do people attach to different ecosystem services? What level should institutional mechanisms for the management of ecosystem services be designed and operated at? Again, these are only a few of the most important and controversial issues that were raised (Hein and van Ierland).

Lots of social conflicts are experienced around nature conservation and biodiversity preservation. Models and insights from social psychology can helps us to understand the nature and intensity of these conflicts and design or re-design policies in order to avoid or at least mitigate the conflicts. On the one hand, there seems to be an untapped opportunity for involving citizens in nature conservation efforts – given the growing public knowledge on relevant issues and a sense of readiness to act (Székely). However, the process of decision-making needs to be designed in a strongly democratic way by involving all stakeholders and paying particular attention to those with the least power to influence and most to loose. Participatory and deliberative decision support and conflict resolution techniques were advocated, along with designing more adaptive institutional mechanisms that by giving voice to local communities and tapping the wealth of their traditional ecological knowledge make biodiversity policies not only more effective but socially just (Stoll-Kleemann, Brown, Roth, Rauschmayer, Berge, Muessner and Chettri). There is an intimate relationship between the spatial organisation of different types of environmental knowledge and their associated organisations of power relations (as the keynote contribution by Roth pointed out with regard to the difference between traditional and scientific environmental knowledge). Spatial flexibility seems to be a desirable characteristic, therefore, for the science of biodiversity as well, with the goal of adaptive science for biodiversity (Rauschmayer).

The issue of social justice at the global level was evidently clear in the discussion of global commons, intellectual property rights (IPRs) and other mechanisms of biodiversity politics at the international political level (Boda, Oksanen and Weikard). The commodification of biodiversity is a strong political force prevailing in our market societies and dominating international politics. Biodiversity issues have become a new arena of political conflict – as some commentator previously put it. There is a need for overcoming ethnocentric myopism, primarily on the part of our culture, and honestly discussing and researching biodiversity issues as deeply political and ethical in nature.

Taking the social, political, and ethical dimensions and complexities of biodiversity issues seriously, we believe, is a must for research and management efforts all over the world.

The philosophical and political significance of place, therefore, should not be underestimated (Lányi, Bela and Kohlheb). Without essentialising locality and placeness, the morality of place and the implication for a more democratic science and politics of biodiversity should be emphasised.

List of contributions

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Scaling problems in biodiversity assessment

Introduction to Session I: Scaling problems in biodiversity assessment

Ariel Bergamini and **Christoph Scheidegger**, Swiss Federal Research Institute WSL, and **Lisandro Benedetti-Cecchi**, Dipartimento di Scienze dell'Uomo e dell'Ambiente, University of Pisa, Session I Chairs.

In many regions of the world, biodiversity loss is seen as a major ecological, economical, and cultural problem. For example, the European Union aims to stop the loss of biodiversity by 2010. Assessment and monitoring of biodiversity over large geographic regions is, therefore, an urgent task for assessing the success of nature conservation policies or agri-environmental schemes intended to stop this decline. Given the urgency of the task, it seems not surprising that the number of scientific papers which include in their abstract both 'biodiversity' and 'assessment' increased from 21 papers in 1995 to 141 in 2004 (slope = 15, $p << 0.001$, data based on literature found by searching the ISI Web of Science on the 1st of March 2005).

The purpose of this e-conference session is to evaluate scaling problems in biodiversity assessment and to identify both gaps of knowledge and further research needs. Obviously, in identifying gaps, it should be clearly stated what we want to know or what a set of stakeholders wants to know, respectively. But, different stakeholders such as politicians, conservationists or scientists might want to know quite different things. However, as a general definition, the following might be useful for this session: A biodiversity assessment describes the state of biodiversity in a certain area (ranging from hectares to millions of square kilometres) such as number of species, describes patterns of biodiversity within the considered area, and, if repeated, should deliver accurate and precise estimates of trends in biodiversity. Thus, a biodiversity assessment may provide the basic data for, e.g., designing networks of nature conservation areas, controlling the effectiveness of agri-environmental schemes, and developing hypotheses on processes underlying diversity patterns.

Obviously, there are several problems related to scale when assessing biodiversity. For example, methods to describe the state of biodiversity (for example number of species of a certain taxon) will depend, among others, on the extent and the complexity of the study area. To describe spatial patterns within the study area, grain size may be critical. Furthermore, the perception of spatial and temporal scales of variability may change depending on the level of taxonomic resolution.

For the purpose of this session, we identified three main issues under the following headings:
1) Biodiversity indicators and scaling problems
2) Patterns of biodiversity and scaling problems
3) Measuring and predicting biodiversity in large areas

However, all these issues are interrelated and sometimes it was not obvious under which heading a certain topic should be discussed.

Biodiversity Indicators: Gathering biodiversity data in most regions of the world is expensive and time-consuming. Thus, cost-effective methods that enable the prediction of species richness have to be developed. A popular approach is to identify some sort of indicator which correlates with the species richness of a particular group or even with overall species richness. Several methods have been suggested which are based on assumed relationships (1) between environmental variables and species richness, or (2) between the richness of different, often unrelated taxa such as birds and plants, or (3) between numbers of species and numbers of supraspecific taxa such as genera or families. This last approach may cause some specific problems such as repeated taxonomy changes or low sensitivity to changes. Another approach is to search for species richness indicators within taxonomic groups (e.g. macrolichen richness as indicator of crustose lichen richness). However, all these relationships may critically depend on the spatial and temporal scales at which a particular study is conducted which, in turn, depend on the specific hypotheses being tested. If identifying appropriate indicators of biodiversity is an important scientific task, linking measurements with clearly stated hypotheses is a must to guarantee scientific progress. In the marine area, the resurgence of interest in indexes of diversity is focusing attention on what to measure without raising analogous concern about how measurement s should be made – i.e. about issues of sampling design in relation to specific hypotheses. For example, the use of bio-indicators to assess human impacts on diversity are often advocated without reference to appropriate controls. This approach rests on the assumption that the chosen metric has no inherent variability and that there are true reference values against which impacts should be assessed. This assumption is unrealistic for any natural system.

Patterns of Biodiversity: Biodiversity patterns are dependent on both grain size and extent of the study area. For example, the famous latitudinal gradient of species richness is only visible if the study area includes several biogeographical zones. However, even if the extent of the study area is very large it is not certain that the gradient would emerge if we used plots of only 1 m2. Furthermore, grain size is critical for the detection of trends in biodiversity and may differ depending on whether we want to measure trends in rare or common species. Furthermore, sensitivity to detect changes of biodiversity in Europe would be quite poor if we considered the whole of Europe as one large plot. Thus, the question is: Are there any general rules to apply to grain size if the objectivity is to measure trends in biodiversity, for example to assess whether we reached the 2010 target? How does rarity (or commonness) affect choice of grain size to detect trends in biodiversity patterns? Comparisons of biodiversity of differently sized regions (extent) is a further, scale-related problem which is often poorly accounted for. Furthermore, the partitioning of diversity in its alpha- and beta-components over multiple spatial scales may provide testable hypotheses on the processes causing the patterns. Understanding patterns of biodiversity also requires proper quantification of variability in chosen response variables at a hierarchy of scales. Because different ecological processes operate over distinct scales, detecting the scales over which biodiversity changes most is important to focus attention on likely causal processes that can be investigated experimentally in subsequent studies.

Measuring and predicting biodiversity: Assessing species richness of large (and complex) landscapes is still a challenging task. If an area is too large for a direct assessment of species richness, as it is in most cases, sampling is necessary. However, species numbers in a given area strongly depend on sampling effort. Therefore, we have two possibilities: to standardize sampling effort or to develop methods that are independent of sampling effort. Thus, the following questions may arise: What methods do we currently know to assess large-scale biodiversity? What are their limitations and pitfalls? Are the methods working equally well for different organism groups such as bryopyhtes and mammals? How can the estimates of diversity obtained with some of these methods (e.g., rarefaction curves) be compared in structured sampling designs including a range of spatial or temporal scales?

Finally, results (or opinions) may depend on the organisms concerned and on the landscapes or ecosystems where the studies are conducted. For this reason, we hope for contributions from scientists with widely varying backgrounds.

Scale issues in agri-biodiversity linkages

Kevin Parris, Policies and Environment Division, Agriculture Directorate, OECD, Paris.

A major challenge in most countries is how to reconcile this trade-off between expanding agricultural production - especially given the projected need to increase global food production by over 20% by 2020 - while securing our planet's biodiversity. Part of the task is to quantify the linkages between human activities and biodiversity. This is not an easy task. There are few countries which have in place systematic monitoring systems that track trends in biodiversity. In addition, there are formidable scientific difficulties in linking changes in biodiversity associated with agriculture to specific policy measures, compared with the influence of other factors on biodiversity, such as natural predators and climate change, although much valuable evidence is being collected, in some cases at the farm or local level.

In response to overcoming some of these data and analytical deficiencies the OECD with its Member countries is developing a set of agri-biodiversity indicators that can help improve the understanding of changes in environmental conditions in agriculture and so contribute to better policies. The OECD work on agri-biodiversity indicators has been developed from an expert meeting held in Switzerland in November 2001 (see details of the related publication and website below).

One job has been to establish a common agri-biodiversity framework (see figure), that helps to simplify the complexity of agri-biodiversity linkages and identify suitable indicators to track agri-biodiversity trends. The framework depicts agriculture in terms of a hierarchical structure. The first and basic scale or layer consists of farm land and its interaction with adjoining ecosystems, such as forests; crop and livestock production species; and production support species, such as earth worms.

The second scale or layer consists of identifying those elements which constitute the quality of the farming system which affects its ability to support a varied biodiversity through different habitat types (e.g. field crops, orchards, meadows); varying structures (e.g. hedges, small/large fields, trees); and farming practices (e.g. organic, extensive or high intensity farming). The final scale or layer in the framework relates the quality of the farming system to its use by wild species (e.g. breeding, feeding), implying that the higher the quality the greater the ability of farming to support a rich and varied biodiversity.

This framework helps us to structure our analysis of agri-biodiversity linkages and respond to a number of important questions that remain to be addressed. What are the impacts of alternative farming systems, such as organic farming, on biodiversity and on sustainable food production capacity? What are the impacts on biodiversity of maintaining current farm policies into the future, relative to those of reducing subsidies to farming? In what ways are international interests in biodiversity and trade liberalisation complementary, or in conflict? Compiling data and developing relevant indicators to monitor agri-biodiversity trends are important steps towards answering these and other questions.

The framework could also be adapted to a more generalised view of all terrestrial biodiversity, and not just confined to agriculture.

See the OECD report (2003) Agriculture and Biodiversity: Developing Indicators for Policy Analysis. A summary of the report is also available in French, German and Spanish, see the related OECD website at: http://www.oecd.org/agr/env/indicators.htm and look under OECD Expert Meeting on Agri-biodiversity Indicators.

Scaling problems in biodiversity assessment

Nakul Chettri, International Centre for Integrated Mountain Development, Kathmandu, Nepal

During the course of biodiversity assessment and scaling, indicator species such as butterflies and birds have played a pivotal role in the development of general theories on conservation arena. Typical host specific requirement of butterflies and well-designated guild structure of birds, provide best indication of habitat quality. Thus, they became the ideal organism to investigate the impact of habitat disturbances and many species may thus serve as bio-indicators. Similarly, plants have been the most widely used predictors of physical conditions and specific site factors but their application has been primarily confined to plant ecology. In the course of biodiversity assessment, many taxa of organism such as ants, tiger beetles, mammals etc. were extensively used as indicator for assessment of biological diversity. Even Odum, noted in his Fundamentals of Ecology (1971 p 138) "…the ecologist constantly employs organisms as indicators in exploring new situations or evaluating large areas" which were used for this purpose in assessment of habitat quality. Conservation prioritization of areas often relies on comparison of the relative or absolute number of species (species richness - e.g. Myers 1990). However, this information is not often readily available. Even for small well-studied groups as birds and butterflies, data are often sparse, especially for regions with high species richness. Most studies have attempted to use indicators to identify areas of overall high biodiversity, by seeking positive correlation between the species richness of the chosen groups and the richness of other groups. Of necessity, however, comparisons are usually made at coarse spatial scales, often across widely divergent habitats or ecosystem, and between groups of organisms, which do not necessarily share the same, or even similar, ecological requirements.

The ever-dwindling global biodiversity and conservation measures to address this have been a major concern for all levels of stakeholders globally. In the recent past, biodiversity conservation and scaling have focused more on holistic basis than species or indicators. Now approaches are evolving. Global ecosystem analysis, global ecoregion analysis, gap analysis, Biodiversity Hotspot analysis, and more recently coldspot analysis, Millinium Ecosystem Assessment and gap analysis for Critical Ecosystem Partnership Fund are some of the assessment made to priorities for conservation. Many of these approaches used rapid assessment techniques. However, none of these approaches has ever addressed the dynamics of biodiversity in relationship with the needs of common people in relation to their socio-econmic and socio-cultural aspects. Due to complexities in approaches and limited coverage of important ecosystems, even conservation measures through protected areas for global biodiversity conservation has been questioned. In past conservation initiatives, the conservationists have been stressing on what to conserve and where to conserve than addressing how to conserve biodiversity practically.

There is an intrinsic reality where the human population increase has been subject to biodiversity loss. However, this factual and dynamic scenario neither received any serious thought in the conventional conservation practices nor in the biodiversity assessment. Global conservation efforts are focusing mainly on protected areas and species. Securing the conservation of biodiversity while at the same time promoting sustainable economic development is one of the greatest challenges of our time. The predominant focus has gone to the creation of protected areas seen as islands of biodiversity, which need to be protected from human intervention. More recently, however, there is increasing recognition of the value that local communities can bring to the process of conserving biodiversity, and of the need for a range of conservation types from strict protection to multiple sustainable uses. Thus biodiversity assessment needs to be related to their surroundings, for which we are yet to device approaches and methodologies to address integrated but complex and dynamic processes at socio-economic, socio-cultural, socio-political and ecological levels.

Biodiversity assessment through Red Book data

Kajetan Perzanowski, Carpathian Wildlife Research Station, Museum and Institute of Zoology, Polish academy of Sciences, Poland

At first, when the deadline of 2010 for stopping the loss of biodiversity was set, this date seemed to be so distant, that even such a formidable task did not look so totally impossible. Now there are only 5 years left and we still are at the point of how to precisely define and measure biodiversity.

I am personally very pessimistic about the chances of preserving biodiversity at the scale of the whole biosphere. In fact - what kind of biodiversity do we want to protect in urban, industrial and large scale agricultural areas?

It seems that the most effective, and probably the only realistic approach is to concentrate our efforts on protected areas like national parks and nature reserves. Those areas contain the most valuable set of biological features for a given region, so if we will be able to stop the biodiversity loss within their boundaries, a reservoir of possibly complete range of presently existing biological variability could be maintained.

Protected areas can be then regarded as representative samples for their geographical zones or eco-regions so any changes recorded in their species richness, composition or trend of biological processes should indicate a direction of changes that can be expected at a regional scale. Another benefit from focusing biodiversity conservation on protected areas results from the fact that usually for such areas there already are quite detailed inventories of flora and fauna which makes any attempt to measure biodiversity much easier.

Such areas are also usually refuges for rare and threatened species which, being the most sensitive to any unfavourable changes in living conditions, may be the most useful indicators of biodiversity loss. Since most probably we will never be able to monitor all species in the biosphere, perhaps a concentration on “red book” species could be a realistic solution.

A lot of efforts and money has been spent on making biological inventories for protected areas, preparations of “red data” books, biological monitoring etc. It seems that finally comes the time to make a practical use of those materials.

Re: Biodiversity assessment through Red Book data

Alan Feest, Water and Environmental Management Research Centre, University of Bristol, UK

At last someone regards biodiversity as more than a list of species or “Species Richness” (see introduction by Ariel Bergamini). The simple application of species richness assumes all species are the same; the domestic cat is the same as a tiger! So the suggestion by Katejan that we should consider other criteria of biodiversity in any assessment is essential if we are to be able to link this with indicators and biodiversity across the landscape.

Comments on the use of butterflies and birds for biodiversity indicators are very helpful but note that the use of birds as biodiversity indicators in the UK might have been very misleading if the choice of indicators had been the obvious one of top predators such as Peregrine Falcon or Red Kite (both of which have expanded their populations dramatically in the last few years). These predators use their habitats on large scale (Golden Eagles need 40 square miles of territory per pair!) and might be expected to indicate the biodiversity of the wider landscape but it is the small birds such as Corn Bunting that have declined dramatically.

Butterflies are much more hopeful as indicators since they will be more sensitive to the precise habitat requirements of their whole life cycle and also respond to global warming (being poikilotherms). This seems to be the case in the UK and is even more obvious when looking at other mobile invertebrates such as Dragonflies, Grasshoppers and Bush Crickets all of which have expanded across the landscape of the UK dramatically. I suggest therefore that the problem of scaling is complex and needs to relate to biodiversity quality however that is measured.

Re: Re: Biodiversity assessment through Red Book data

Kajetan Perzanowski, Carpathian Wildlife Research Station, Museum and Institute of Zoology, Polish academy of Sciences, Poland

Alan Feest is perfectly right. The same process we observe in Poland, where for a number of years all birds of prey were under strict protection and there were even special actions undertaken (e.g. subsidised mowing of abandoned pastures) to improve habitat for them. As a result, there is a visible increase of lesser eagles, buzzards, hawks etc. but at the same time some small birds, especially those nesting on the ground (e.g. corncracker, quail) become less frequent.

Some “top” predators (e.g. wolf) are known to be quite flexible regarding the selection of prey, and may easily shift to other species if their main prey is disappearing.
Therefore the nice idea of using “umbrella species” as indicators of biodiversity cannot be used indiscriminately.

On the other hand, the presence of some animals like beavers and large herbivores which create and maintain new habitat niches for other species may perhaps be useful to indicate areas of higher biodiversity.

Biodiversity valuation

Nakul Chettri, International Centre for Integrated Mountain Development, Kathmandu, Nepal

SUMMARY: Human induced disturbances are changing the world biodiversity at unprecedented rates. Activities such as firewood extraction and grazing may bring just subtle manipulations than commercial logging, but both are equally important for the changing scenario. Identification of areas of high conservation interest requires substantial time, resources and effort for detailed inventories. Therefore, conservationists are interested in selecting a few efficient indicator taxa for measuring and monitoring biological diversity (Kremen 1992; Faith and Walker 1996).

In the mountains, birds have been considered as good predictor of the quality of habitat as they exhibit numerous relationships with changes in their associated habitats (Shankar Raman et al 1998; Shankar Raman 2001, Chettri et al 2001) because they respond to habitat structure (Chettri et al. 2005) and represent several trophic groups or guilds (Chettri et al. 2005). Many bird communities' distributions are affected by habitat fragmentation or other habitat parameters and reflect interspecific dynamics and population trends associated with the habitat (Chettri et al. 2001, 2005). A number of such studies are available that suggest that bird communities have high potential to act as surrogate for their habitats at structural, regional and landscape level management (O'Connell et al 2000; Canterbury et al 2000; Lindenmayer et al 2000). However, these surrogates or indicators could be used for different purposes and objectives (Lawton et al. 1998). Bird communities were mainly found to be used for the forested landscape with mosaic of habitat; we do need to consider their habits and guild while doing these interpretations. Such species could be separated from the analysis while working for small areas. There has been instance where Honrbills were used as indicator species which need a broad habitat such as raptors.

Defining 'biodiversity assessment' correctly

Keith Rennolls, CMS, University of Greenwich, UK.

Ariel Bergamini, in the Opening statement on Scaling problems in biodiversity assessment considers many important issues in an interesting way, and I am sure his opening statement will prove stimulating to many discussants. However, early in his statement he proposes the following: "as a general definition, the following might be useful for this session: A biodiversity assessment describes the state of biodiversity in a certain area (ranging from hectares to millions of square kilometres) such as number of species, describes patterns of biodiversity within the considered area, and, if repeated, should deliver accurate and precise estimates of trends in biodiversity."

A lot of the ideas, concepts, methods associated with biodiversity often remain vague and intuitive and are not defined precisely. Hence it is good to see an attempt at a definition of a "biodiversity assessment".

Unfortunately I find the definition to be rather vague and misleading, as described below. I hope after considering the limitations of this definition I can offer a better alternative.

1. "biodiversity assessment describes the state of biodiversity"

Any assessment procedure (of biodiversity, or whatever) may be regarded as a "measurement instrument" of the attribute of the entity being measured. The measurement instrument might involve the use of sample survey inventories, ground sampling and remotely sensed methods of data collection. The data collected will reflect the population entity being sampled/measured and the design and structure of the measurement instrument used. This data alone does not provide a measurement or assessment of the population diversity. The data has to be processed or analysed in an appropriate way for a measurement or assessment to have been made. Such data analysis may take into account the sampling design used (eg. Stratification and variable probability sampling etc) make use of models, etc. The point is that one cannot complete a measurement or assessment until one defines exactly what attribute of the population entity it is that is being measured/assessed. To say that an assessment "describes" the state of diversity is rather too general and vague to be of much real use in defining what constitutes a biodiversity assessment. On the one hand "describes" could be regarded as a very ambitious aim, if interpreted as a complete characterization for the complete biodiversity state-space-structure of the population, as revealed by the collected data. On the other had "describes" may be interpreted in a vague an intuitive sense. Both interpretations of "description" have their scientific value in biodiversity research, but possibly not ideally placed within a definition a biodiversity assessment.The biodiversity of a population (entity) is multi-dimensional (and patterned) and there are many associated (population) biodiversity measures-parameters-attributes which may be estimated using estimators or indices based on use of the data collected.To me it seems that the definition of a biodiversity assessment should be rather more constrained to a precisely defined mensurational context, and in this context it is necessary to define exactly the biodiversity state-variable/parameter, (attribute, structure or feature) that one is attempting to assess.Ariel does give an example of what he means by "state of biodiversity", as "number of species", the most popular of species biodiversity measures (i.e. species abundance). It is this specific example that is worthy of following in the general definition rather than the undefined "state of biodiversity".

2. "in a certain area" The data-collection for biodiversity measurement-assessment always involves a sample of area, or of population elements. Sample statistics can be calculated from the sample data as indices of biodiversity in the sample units themselves. Though this may have some (regional) comparative value for a simple study in which the sample units are standardized. However, this will not be achieved between biodiversity assessments, which will invariably choose their own sampling units. Hence, it would seem that estimation (from the sample data) of the chosen population biodiversity parameters/attributes of the region (in which sampling is carried out) should invariably be the aim of a biodiversity assessment. Ariel asks some wide-ranging and challenging questions in relation to how to do such extrapolation. He mentions in particular rarefaction curves, which I suppose is the same as species-area curves for plant population studies. Such considerations are often terms the scale-dependence in biodiversity sampling. Car needs to be taken between this kind of scale-issue, and the scaling that occurs when the nature of the objects observed changes qualitatively, and in-scale as the scale changes.On this point, it seems to me that there is some confusion in the ecological world on the use of such curves. They sometimes seem to be regarded as objects of study themselves and not estimators of population biodiversity parameters. However, they are in fact constructed from the sample data; the curves and models fitted to them are just ways of estimating from the sample to the population, just as in estimators such as the Chao, Jack-knife and more recently used Horwitz-Thompson type estimators of species abundance or other measures of diversity. I would also mention that the limited area sample is invariably a measurement instrument which is also limited by its minimal measurement size, or the granularity, as mentioned in the opening contributions. Hence all forest tree biodiversity inventories have their own minimal size of tree that will be included in the assessment. Of course different studies choose different minima, and this makes a very large difference to the recorded data, eg. The number of species recorded. Hence if we are always to consider extrapolation to the large area population, we ought to also model and extrapolate our estimates to small diameter cut-off thresholds, or adopt a common minimum if comparisons are to be made between studies.

3. patterns of diversity: also rather too general for precise definition purposes. Connotations would seem to include the various relationships that that might be discernible or

discoverable in the biodiversity state of the population. Again I would rather the general definition required the pattern being assessed to be specified. However, I see that such a view does mean we can only notice changes in those aspects which we are looking for. I suppose in general, when concerned with monitoring of biodiversity, rather than assessment, a looser definition might be more appropriate.

4. "and, if repeated, should deliver accurate and precise estimates of trends in biodiversity". Any assessment will deliver its estimates with a measure of uncertainty. This uncertainty, (or standard error maybe) will depend on the size of the sample, the estimators being used, and what is being estimated. If one is only assessing the proportion of the dominant species, then accuracy will be high even with relatively small samples. However, if the focus of concern is with the aspects associated with the very rare species in the population, then even large samples will have relatively large errors of its assessments/estimates. Hence it is not reasonable in a general definition of a biodiversity assessment to require accurate estimates, let alone accurate estimates of change, between re-assessments, let alone estimates of trends in biodiversity. Such words seem to be inserted out of the natural desire to obtain accurate results, which has little to do with the actual accuracy which will be obtained from an actual biodiversity assessment of a specific biodiversity measure.

So much for my analysis of Ariel's Definition. Let's now try a modified definition: DEFINITION. A Biodiversity Assessment is a process which measures/estimates a well-defined biodiversity (state or pattern) measure, or measures, on a target population using data sampled from that population and estimators which scale appropriately from the sample to the target population.

Apologies to those readers who find my considerations rather too pedantic. However, I feel much care is needed in the early definitional stages if one is to avoid undue and unfulfilled expectations from those who might have a political and strategic involvement in biodiversity conservation, an do not have the interest or the inclination for the technical assessment matters.

Re: Defining 'biodiversity assessment' correctly

Ariel Bergamini, Swiss Federal Research Institute WSL, Session I Chair.

Keith Rennolls discussed my definition of 'biodiversity assessment' (the opening statement was written by me and Lisandro Benedetti, although the definition was my idea) very thoroughly. Thanks for that! The definition was quite an 'ad hoc' creation and certainly not perfect. I will first shortly discuss the points listed by Keith:

-"biodiversity assessment describes the state of biodiversity": I completely agree that this is a vague statement. 'Biodiversity' and 'describe' can mean a lot of things and that's why I chose these terms. Furthermore, 'Biodiversity' has been defined uncountable times during the last fifteen years. Every biodiversity assessment is different in the methods applied, which depend on the precise aims and on the money which is available. Some assessments may only deliver the number of birds in a certain area (locality, region,..), other assessments may provide very detailed measures of diverse taxa in many plots distributed somehow in the study region. Other assessments may consider number of different ecosystems or number of genotypes of a taxon. Because of the variation of assessments and assessed things, I chose not to be too precise.

-"in a certain area": I simply meant, that by the end of the analysis something about biodiversity in the studied region is known. I did not use the term region because this term implies a certain spatial extent of the study. Thus, I used the neutral term area. I did not consider in the definition how to assess data or to analyse data to get a good estimate for the study area. Certainly, it should be the aim! Comparing assessments is a very difficult task. Mostly study areas vary in extent and the methods applied will rarely be identical. Keith Rennolls is thus perfectly right if he claims that an assessment should be done in a way such that comparisons with other assessments are possible. However, it's still a biodiversity assessment if such comparisons are not possible.

Next week we will have a keynote by Nicholas Gotelli on the use of rarefaction curves and other methods. Thus, I propose to postpone the discussion on rarefaction. Comparisons between assessments are certainly a problem because of different methods applied.

-"patterns of biodiversity": the patterns detected in a assessment certainly depend on grain size or other aspects of the sampling. The patterns one wishes to detect depend on the precise aim of the assessment.

-"and, if repeated, should deliver accurate and precise estimates of trends in biodiversity": I can't see what's wrong with that. Is imprecise better?

Altogether, I can't really see the improvements of the new definition. The assessment of biodiversity is a process, however, what it should deliver is a result (measures, estimates etc). And the value of an assessment clearly depends on whether the results are useful for the stakeholders.

Spatial complexity, power-laws, and issues of scale in marine benthic landscapes

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SUMMARY: Systems that are dynamic and that grow from nodes are often self-organized and spatially complex, with scale-invariant properties. Insights from theoretical and empirical investigations incorporating these concepts may be useful in understanding and managing biodiversity in marine benthic landscapes.

Issues of scale are important in the ecology and economics of marine benthic landscapes, including biodiversity issues. Scale has generally received less attention in marine than terrestrial landscapes. However, recent advances in the theory of complexity and self-organization have converged with improvements in remote sensing and empirical field methods in marine ecosystems. The result is a promising synthesis, although much additional work must be done.

Systems that are dynamic rather than static and that grow from nodes rather than at random are often self-organized and spatially complex. The signature of self-organization is the power-law frequency distribution of "event" sizes, with a negative slope straight-line on log-log axes. Such systems are scale-invariant. They look the same at all spatial scales. Famous examples include networks like the Internet and electric power grids. In the natural sciences, well-known examples include weather patterns, earthquake and avalanche size distributions, and groundwater and surface flow regimes.

Marine systems may be particularly well suited to this type of analysis. Some of the earliest examples of spatial power-laws in nature were fractal geometry descriptions of the coastlines of the United Kingdom and Norway. The great appeal of this approach is that it makes intuitive sense: the complex, jagged, irregular coasts shaped by glaciation and sea level rise have a larger fractal dimension than straight or gently curved shorelines.

What about biological systems? We know that the interplay between physical and biological processes creates spatial complexity in living systems. Examples include rocky shore and soft-bottom mussel beds, mudflat diatom mats, saltmarshes, and coral reefs. Often these systems display a strong power-law signature. Moreover, this complexity is revealed over a broad array of spatial scales: soft-bottom mussel beds are fractal at the millimetre to centimetre scale as well as at the scale of entire beds that are thousands of square meters in size. These power laws demonstrate that mathematically defined order exists in systems that appear visually "messy" or "random" and are sometimes mistakenly called "chaotic."

How useful are these insights to ecologists and managers interested in analyzing and protecting biodiversity? They have tremendous potential. First, these approaches are theoretically tractable. Much of the mathematics dealing with complexity is fairly accessible, and cellular automaton modelling can be performed easily because the model parameters and processes are often few in number. Second, these approaches are inexpensive. The necessary empirical data can be obtained at moderate cost with remote sensing techniques including aerial photography, satellite imagery, and side-scan sonar. Huge amounts of remotely sensed data are already archived by local, national, and international agencies, meaning that time-series and "before-and-after" investigations can be conducted for many marine systems, even going back in time before the present day.

What are some existing and potential applications?

Monitoring system health. Theoreticians have argued that fractal dimension should vary with the health of a natural system. For example, a benthic system that has been fragmented by harvesting pressure or pollution stress may have a fractal dimension that is different from that of a pristine system. Relatively rapid, low-cost remote sensing over large areas may detect such changes.

Determining the impact of dredging and trawling on the seafloor. Workers have used fractal geometry to describe the surface complexity of soft-bottom intertidal mussel beds and rocky shore assemblages of barnacles and mussels. Similar approaches were successful in quantifying the "smoothing" of the seafloor by trawling compared to more complex untrawled areas of the continental shelf.

Improving the efficiency of monitoring programs. The degree to which an ecological system is actually scale-invariant can theoretically be useful in designing the most efficient, least expensive monitoring program. A completely chaotic system needs a very intensive sampling effort. On the other hand, a perfectly scale-invariant system with a very strong power-law signal would need to be sampled at the most convenient spatial scale with a small number of samples (in the limit, only one sample!). Computer scientists have known this for a long time and have incorporated fractal geometry into their data compression designs to save space and money. Marine ecologists have not expressed much interest in using similar approaches to achieve these economies.

Understanding control processes. Theoreticians have predicted that exposed coastal systems may be controlled primarily by external forcing functions such as storm waves, whereas sheltered locations may be occupied by systems that are more internally self-organized by, for example, gregarious recruitment. Both are spatially complex, but they are controlled in very different ways that may be revealed by different power-law signatures.

Designing marine reserves. Theory predicts that complex, large-scale spatial patterns can be the result of very local, small-scale physical and biological interactions. In other words, marine systems exhibit self-organization over a variety of spatial scales. Only by working at all these scales can systems be protected with well-designed and well-managed marine reserves.

Predicting responses to global climate change. Storms, wave action, water current velocities, and sedimentation rates may all change locally in response to global climate change. These parameters are known to affect the growth and destruction of coral reefs, mussel beds, diatom patches, and saltmarshes. Modelling the feedback between physical and biological processes will help us predict how these self-organized systems will respond to global climate change.

To summarize, great theoretical and empirical advances have been made in the past few years that help us understand how landscapes are organized. So far, relatively few attempts have been made to apply these spatial analysis techniques to marine benthic landscapes. They merit our attention, especially with respect to protecting biodiversity.

On the stability of relationships between indicators and the indicated groups across spatial and temporal scales

Carlo Ricotta, Department of Plant Biology, University of Rome "La Sapienza", Rome, Italy

SUMMARY: Assessment and monitoring of ecosystems at the landscape scale requires characterization of spatial and temporal land cover patterns. Ideally, we want a quantitative basis for deciding when the relationships between indicators and the indicated groups undergo substantial changes over space and/or time. In this view, it is critical that one establishes a methodology for assessing multiscale landscape structure and its subsequent influence on ecosystem functioning.

As emphasized by Podani (1992), published work on landscape structure may be arranged along a hypothetical gradient, which reflects the degree of understanding the importance of scales in pattern analysis. At one end, total ignorance of spatial dependence is typical, whereas the problem of scales in landscape analysis is deeply understood at the other end. Nonetheless, since landscape structure is spatially correlated and scale-dependent, its quantification necessarily requires multiscale information, and scaling functions are the most precise and concise way for summarizing multiscale characteristics explicitly (Wu 2004).

The basic idea is that multiscale features of the landscape structure may be captured in a sequence of successively coarsened resolutions. Hence, instead of letting resolution be a problem, scaling functions employ the effects of variable resolution becoming even more informative about spatial pattern than characterizations based on a fixed scale. This approach is in line with hierarchy theory (O'Neill et al. 1991) which suggests that patterns observed at a given resolution will constrain to some extent the patterns observed at finer resolutions and will in turn be constrained by coarser resolution patterns. Thus, determining the range of scales over which landscapes exhibit hierarchically-nested spatial pattern provide a means for of identifying the scales at which fine-scale processes affect 'global' scale patterns and vice versa.

Finally, it is worth noting that the application of scaling functions is not limited to spatial and temporal scales. For instance, in ecological work, there is a plethora of less conventional but by no means less interesting topological data spaces that may be conveniently explored by scaling functions. Some interesting examples are: exploring the effect of different levels of nested classification schemes, such as CORINE Land Cover (Fuller and Brown 1996) on landscape structure, or the application of parametric landscape metrics that possess different sensitivities to rare and abundant land cover classes as a function of the selected parameter (Ricotta et al. 2003).

All these different scaling functions provide a different piece of information, thus being complementary rather than competitive within the context of a complex, plural and dynamic approach to multiscale ecosystem analysis.

On the use of higher-taxon richness and problems related to scale

Pedro Cardoso, Zoological Museum, University of Copenhagen

SUMMARY: The higher-taxa approach has been seen not only as a reliable but extremely useful method both for species richness prediction and in helping to define a network of conservation priority sites. The problem of scale may cause the approach to fail though, and a large assessment program may be needed to answer many fundamental questions.

Higher-taxa richness has been tested and proven as a possible and easy shortcut in biodiversity assessment and ranking of areas for conservation priority. Shortcuts like this are especially useful in taxa whose information is scarce, identification time-consuming and availability of taxonomists very limited. But this is the case of most taxa, namely the speciose ones like arthropods, which constitute the bulk of biodiversity. To identify individuals only to respective genera or families allows information to be obtained on a large number of taxa and sampled areas with relatively low effort and resource use. Another crucial advantage is the retention of broad biological information that allows the understanding of distribution patterns and more efficiency in the definition of conservation priority areas.

Although higher-taxa surrogacy has been tested in a variety of regions and taxa, a lot has still to be known. We don't know for what taxa it is really valid or for which regions it is especially useful (although in Europe the Mediterranean region would definitely be the most benefited). Also, the reliability of the approach can be affected by effort and consequently, scale. All this to claim that we really don't know how scale affects the higher-taxa approach, this scaling problem has never been tested to my knowledge. If we test the approach by comparing habitat patches in small areas we may have totally different outcomes from the ones we get if we compare different countries in a continent. Histories are different, biogeographical areas are different, and these factors can certainly influence the ratio between species and higher taxa. The change in scale leads to a change in this ratio which in turn may cause the approach to fail, if not taken carefully. The future may depend on a large biodiversity assessment program to be conducted in a variety of countries and with as many taxa as possible, probably by the application of a nested sampling design, so that several sampling areas are nested in each of different habitat types, regions, countries, major biogeographical areas and up the scale. Such a project would allow not only testing the validity of higher-taxon richness in biodiversity assessment independently of scale, taxon or area but also a number of other questions in macroecology and conservation.

Scaling in biodiversity: biomes, biogeography and life forms

Rob Jongman, Alterra, Wageningen UR, The Netherlands

Biodiversity is a complex issue to tackle; but we also have to accept that it is a world-wide issue. This also means that local regional, continental and world-wide issues have to be tackled to grasp biodiversity. The GEOSS report states that biodiversity is necessary for the sustained delivery of the goods and services essential for human well-being, as well as for the maintenance of life on Earth in general.

If we are to understand biodiversity and its loss, build global, regional and national baselines, make rational management decisions and assess the success of conservation measures, many sources of biodiversity observations must be pooled. Most biodiversity observations are, and will continue to be, made in situ. If you want to measure biodiversity you have to think about the level of aggregation. Then the question is important if a Jaguar does represent soil biodiversity or aquatic biodiversity and if can it be used as a regional or world level biodiversity indicator? That is not the case. If you want to measure biodiversity at the level of a biogeographic region a large scheme of indicators is needed if focusing on species.

According to the GEOSS report integrated biodiversity data is needed for local, national and international policy makers to develop science-based policy, establish priorities in biodiversity action plans and to implement legislation, especially in the context of international conventions. Requirement for upscaling and downscaling is that a sampling strategy covers all major ecosystems at the level involved and that it is done systematically without regional bias.

The clearest approach is to stratify the world in zones as has been done in classical ecology and make use of the classic indicators that are available. At the local and regional level these are representatives of local species groups; these differ between deserts and wetlands and between continents. When we want to compare biodiversity we have to search for common denominators for all parts of the world and these are plant life forms and its complexes. They are the common denominator used already in the past by Walter (Vegetation and climate zones), Raunkiaer, in many floras of the world, in moderate climates as well in tropical climates (Tutin et al, Pignatti, Aeschimann). Life forms can be used to identify habitats and these can be a proxy of the baseline for biodiversity. Different levels of detail are possible and it allows us to compare for instance the Mediterranean regions of the world in their complexity and completeness. In a testing on Europe it appeared hat different biogeographic regions have different combinations and differences in dominance of life forms. As the number of life forms is restricted and can be generalized over the whole globe as expected, this might be an entrance to worldwide biodiversity assessment.

On the use of higher-taxon richness and problems related to scale

Shonil Bhagwat and **Paul Williams**, Biogeography and Conservation Laboratory, Natural History Museum, London, and **Katherine Willis**, Biodiversity research Group, School of Geography and the Environment, University of Oxford

The estimates of global species richness vary widely. Although 1.75 million species are named by science (www.species2000.org (last accessed: March 2005), some experts believe that this number could be as little as 10% of the actual global species richness. The problems of estimating species richness of some hyper-diverse taxa such as insects and fungi are further compounded by the lack of sufficient taxonomic knowledge. This poses serious challenges to biodiversity assessment and conservation-priority setting. To circumvent these challenges, as well as to reduce the cost of sampling, one obvious shortcut is to carry out biodiversity inventories only down to the level of genera, families or orders rather than species (Williams 1996). A close correlation between species richness and higher-taxon richness has been demonstrated (Williams 1994, Balmford et al. 1996, Balmford A. et al. 2000, Grelle 2002, Prinzing 2003, Villaseñor et al. 2005).

While taxonomic classification systems reflect evolutionary processes, they are highly influenced by the ways in which humans have chosen to organise information. Therefore, higher-taxon approach to measuring biodiversity has been criticized in some quarters. Prance (1994) did a cross-continental comparison of species, genera and families of plants in tropical regions and concluded that species diversity of neotropics was much higher than in African or South-east Asian tropics, although the family richness was very similar. Anderson (1995) compared Australian ant faunas at 24 sites and concluded that ant genus richness was not a useful surrogate of ant species richness. A closer look suggests that the scale at which these studies were carried out might have influenced their findings. While Prance's investigation compared global patterns of plant richness across continents and in areas of different sizes (Williams et al. 1997), Anderson's study addressed local-scale patterns. All other studies that demonstrated a close correlation between species richness and higher-taxon richness were conducted at regional scale.

The factors influencing the distribution of biodiversity are delimited by scale (Willis & Whittaker 2002). The global-scale distribution of families between continents may have been driven by historical processes (continental plate movements, sea level change) acting over hundreds of millions of years. At regional spatial scales, environmental factors acting over time scales of 1-10 million years (glacial-interglacial cycles, water availability) may have had the strongest influence on the evolution of species within families. On a local scale, environmental events operating over periods of 1-1000 years (habitat structure, disturbance by fires or storms) may have influenced patterns of species distribution. A close correlation between species richness and higher-taxon richness at regional scale may be expected; however, at global scale this may be constrained by the precise understanding of phylogenetic biogeography of the group of organism in question; and at local scale by stochastic factors responsible for ecosystem dynamics in space and time.

It is obvious that higher-taxon approach may not be necessary for biodiversity assessment where higher-taxon richness is similar to species richness. It has been suggested that such approach will fail for groups of organisms in which numbers of higher taxa are extremely small relative to numbers of species – for example, on isolated islands (Altaba 1997) or in biotas with unusual taxonomic structures (Cronk 1997). However, it has also been shown for various regions and groups of organisms that the above factors may have an effect, but will not cause the higher-taxon-richness–specie-richness relationship to break down (for example, see Williams & Gaston 1994).

In conclusion, future studies seeking to use higher-taxon richness for biodiversity assessment will need to pay a careful attention to the group of organisms in question; and taxonomic, spatial and temporal scales.

Congruence of diversity patterns at the genetic and species level?

Felix Gugerli, Swiss Federal Research Institute WSL, Birmensdorf, Switzerland

SUMMARY: Large-scale studies on multi-species variation at all three levels of biodiversity, and the relationships among them, are necessary to better understand the processes and driving forces that underlie the patterns of diversity observed in space and time.

Biodiversity is still mainly looked at from the species perspective! Habitat heterogeneity is sometimes considered, whereas the third level, i.e. genetic variation within species (interspecific diversity), is strongly neglected in assessments of biodiversity. However, genetic variation is one of the key components in evolution and thus in the species' adaptation to the ever changing environment. There is an urgent need for describing gene diversity over wide geographic scales, with a sufficiently dense sampling. We need to search for spatial patterns in genetic diversity, for correlations with species and habitat diversity, and to find the key factors that allow us to explain how these patterns have evolved.

One may assume that intra- and interspecific diversity should be linked because the former may be considered the precursor of the latter. Further including the third level of biodiversity, it seems plausible that heterogeneous habitats invoke heterogeneous species assemblies and, likewise, genetically variable species. So why should not all three levels of biodiversity be correlated? To date, I am not aware of any study that has addressed this issue at the multi-species level, be it on small or large geographical scale (but see e.g. Vellend 2004). Not to mention the identification of the driving forces!

A multi-national consortium tackles this novel aspect of biodiversity assessment in the EU-funded project INTRABIODIV (Tracking surrogates for intraspecific biodiversity: towards efficient selection strategies for the conservation of natural genetic resources using comparative mapping and modelling approaches; http://intrabiodiv.vitamib.com/). In two well-delineated geographical areas, the European Alps and the Carpathians, all three biodiversity levels will be studied on a common, regular grid system across these mountain ranges. Plant species richness in high-mountain taxa will be identified, environmental variability modelled on the basis of biophysical parameters, and intraspecific genetic diversity measured in 30 widespread plant taxa. This comprehensive data set will allow us to search for potential correlations among the three levels of diversity. Even more so, it may be possible to pin down those factors that best explain the diversities found and that may underlie the biodiversity observed in the two study areas. Future work should expand into other organismal groups besides plants, and also include diversity in interactions such as competition/facilitation or interactions among trophic levels.

When interpreting the patterns of genetic diversity, attention has to be paid to respect the fundamental differences between neutral and adaptive variation. To date, we are able to assess only neutral genetic variation by applying molecular markers – with only few exceptions. These are the markers of choice when aiming at tracking processes in space and time, such as gene flow via pollen or seed. Evaluating selective genetic variation, on the other hand, requires labour-intensive common garden experiments. At the molecular level, functional genomics that study variation in the expression of genes, and ecological genomics, searching for sequence variation within selective genes in relation to different environments, are still in their infancies. However, these studies are limited to experimental plots and far from expanding into populations or even landscapes.

Towards a unification of methodologies to assess biodiversity at landscape scale

Ortega Quero, Centro de Investigación Forestal. Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria. CIFOR-INIA, Madrid, Spain

SUMMARY: Understanding and monitoring spatial distribution and patterns of biodiversity is not a simple question because includes not only knowing species diversity of different biological groups, some of them with very complex spatial and time distribution patterns, but also the problem of scaling in biodiversity assessment.

Landscape scale then acquires notable significance for biodiversity assessment, but research on indicators about biological and landscape biodiversity has not advanced enough to provide results (COM(2001)619). Usefulness of indicator species has been proposed and verified (Noss, 1990, 1999), but although the analysis of landscape structure has been developed remarkably (Turner & Gadner, 1991), an approach linking landscape structure with biodiversity assessment is still necessary.

To assess biodiversity at landscape scales, an emergent parameter is needed which would be easily analysed at territorial scales and would have a close relationship with the biodiversity of the ecosystem. This parameter should also maintain its bias, accuracy and precision as much as possible when used at different scales (Hellmann & Fowler, 1999). Plant communities replies rather well at some of these requirements and have been used for a long time like biological indicators (Mueller-Dombois & Ellemberg, 1974). More recently, plant communities have been proposed to describe biodiversity at a landscape scale (Noss, 1987) and they have been tested as parameters to discriminate and classify ecosystems by remote sensing techniques (see Nagendra & Gadgil, 1999a,b).

Recently, our research team has published a paper that describes a methodology to assess plant diversity at landscape scale linking landscape structure and proved it in three Spanish rural municipalities chosen as landscape units because this administrative scale involves both social and economic factors, and cultural landscapes are a result of these factors combined with natural conditions (Ortega et al., 2004). This methodology was based on:

- Territorial identification of plant cover and land uses types (CUTs) based on remotely sensed information including environmental factors (such orientation or altitude) to detect compositional differences within CUTs, because the Mediterranean region have heterogeneous landscapes and frequently shows topographic and climatic variability (Cowling et al., 1996).
- Multi-scale field techniques to asses plant diversity in CUTs, using a sample design as defining by Stohlgren (1995), and
- Quantification of plant diversity by means of an additive model, proposed by Lande (1996) and used by Crist et al. (2003), that partitions gamma diversity (or diversity at territorial scale) into its alpha (mean diversity of CUTs) and beta (diversity similarity between CUTs) components. We proposed to weight plant communities in relation to some metrics of landscape (area of CUTs, number of patches, spatial distribution of patches, etc) in order to link landscape structure to assessment of diversity.

This type of territorial analysis methodology could predict any change of use or perturbation in a piece of territory. So it would be a good tool for the sustainable management of land uses. But in any case it is only an indicator of true plant diversity and therefore of biodiversity. An accurate assessment of biodiversity at landscape scale doesn't exist nowadays. So other approximations to assess biodiversity at landscape scale are necessary in order to contrast results.

But what is the size or scale of landscape that we have to consider for evaluating biodiversity? For plant diversity we proposed a administrative landscape unit, but for mammals it would consider other physical boundaries, such as roadsides for example and for birds with large habitats it would be necessary to aggregate land cover. Landscape heterogeneity measured by the number of land cover type/hectare is an important factor affecting the species richness pattern of some animals, while other animal groups are more affected by abundance of particular land covers or favourable habitats (Atauri, De Lucio, 2001). The regression analysis provided by generalized additive and generalized linear models are used to study the distribution patterns of species (Guisan et al, 2002). Both approximations could be used to predict species' potential habitats and to evaluate the loss of biodiversity.

Another important goal would be to get a standardisation of methods to evaluate biodiversity at the landscape scale taking into account a unique classification of species' habitats in order to detect and protect the "hotspots" of biodiversity. These methods could also be implemented in the planning of sustainable management of land uses.

Scale change in non-linear systems with feedback, spontaneous self-organisation and emergence in biodiversity

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

I think that Dr. John Commito's contribution ('Spatial Complexity, Power-Laws, and Issues of Scale in Marine Benthic Landscapes') raises a particularly important issue. I'd like to take his comments on non-linear, self-organising systems a little further.

I suspect that many of the problems related to scale in understanding biodiversity are really problems of emergence. We all agree, I think, that at any level 'genetic, organismal, population, species or ecosystem' we are dealing with complex biological, ecological or cultural systems that are influenced by so many variables that it is rarely possible to model the system comprehensively. At every scale, elements of the system interact in non-linear ways that include both positive and negative feedback and randomness. Non-linear interactions with feedback is a hall-mark of both (mathematically) chaotic and emergent systems.

When we look at an ecosystem we can talk happily (although sometimes perhaps unthinkingly) about ecosystem processes, or functions, or goods and services; in other words, we see an organised and somewhat predictable entity. Where does this organisation come from? In a sense, from nowhere - the components of the ecosystem interact to produce spontaneous self-organization.

'Spontaneous self-organization' is the classical definition, if there is one, of emergence. What else characterises an emergent system? Three things: Simplicity: We can describe the properties of the ecosystem in simpler terms than we can the properties and interactions of the thousands of species that compose it. New concepts: The vocabulary we use to describe the properties of the animal is different from the vocabulary we need to describe the liver and its interactions with the rest of the organs. Autonomic behaviour: The behaviour of the cell arises from, but is in some sense autonomous from, the behaviour of plasmids, Golgi bodies and the endoplasmic reticulum.

When we change scale in the study or assessment of biodiversity, we often cross an emergence boundary. In looking at increasing scales, the whole we see at the new scale is different from the sum of the parts we saw at the previous scale. The change of scale confounds us because the rules that operate at one scale are no longer visible at the larger or smaller scale. Like Heisenberg's uncertainty principle, this is a fundamental property of nature, not a limitation to do with our conceptualisation of reality.

In changing spatial scale, we see that each ecosystem, from the landscape down to the microscopic, is an assemblage of smaller ecosystems. The properties of the larger ecosystem is not necessarily predictable from the various properties of the smaller ones. Where this is the case, reductionist science no longer helps us. I am increasingly convinced that to improve our understanding of biodiversity we must make a concerted effort to understand the implications of the fascinating topic of emergence.

In closing, I suspect that much of Alan Feest's disquiet with the conventional definitions of biodiversity may stem from the way "biodiversity" changes its nature as we cross boundaries characterised by emergence.

Sharman's emergence

John Commito, Professor of Environmental Studies and Biology, Environmental Studies Department, Gettysburg College, USA.

Thank you very much, Martin, for your comments on emergent properties in self-organised systems. I particularly appreciate what you said about simplicity. Using just a few organizing rules, very complex patterns can emerge. This is a heartening conclusion because to monitor and make predictions about the real world, we empiricists don't have to measure every single abiotic and biotic factor and all their possible interactions!

For example, my work with a very simple model of mussel bed spatial pattern development relies on two rules: (1) beds grow, and (2)growth is from random recruitment to bed edges. The resultant spatial patterns are fractal and very complex. They match real world beds very nicely.

Mussel beds can both promote and reduce biodiversity, so understanding their spatial dynamics is crucial to understanding marine benthic biodiversity patterns wherever beds exist in rocky shore and soft-bottom habitats. I suspect that the same is true for other biogenic structures like worm reefs and coral reefs. Because they are fractal, they are scale-invariant over the spatial scales that have been studied thus far. As I said earlier in my first posting, scale-free spatial structure has many implications for studying and managing biodiversity.

The synergetic approach of environmental issues for a better integration of the ecological, political and economic scales

Christian Kleps, Romanian Academy of Agricultural and Forestry Sciences, Bucharest, Romania

SUMMARY: Analyzing the environment problems in an integrated approach always offers technical, scientific and financial advantages.

Before approaching biodiversity problems on European or planetary scale, I consider as a logic priority the elucidation of all relevant aspects at a national scale. This item is necessary both following the different share on ecosystems types and of the existing differences concerning the involved forces in biodiversity research, generated by political, social and economic factors specific to each country. For the most efficient development and capitalization of this research, whenever it is possible, we consider it useful to develop a synergetic approach, which should meet the efforts to solve various environment complementary problems. This approach always represents the advantage of offering supplementary information concerning both the existing common drawing-backs and inter-influence among these factors, for example the ones resulted from the inter-relation between climatic changes, land degradation and biodiversity. In Romania an experimental project on this issue was recently developed with good results.

In January 2004, the United Nations Program for Development together with the Romanian Ministry of Environment and Waters Management issued out a program to support the implementation of those three Conventions from Rio (the Framework Convention of the United Nations on Climatic Change, the Convention on Biologic Diversity and the Convention on Land Degradation and Desertification). The Project entitled "National Capacity Self-Assessment for Global Environment Management" intends to identify national priorities and needs in the field of the institutional capacity of those three Conventions for an integrated approach to global environment problems. The catalysis of internal and external actions in this field in a coordinated and planned way was followed in this project, by exploring the synergies among those Conventions and the necessary priorities and selections of those common actions and measures that can contribute to support those implementations in Romania with efficient financing.

Following the project analysis and responses at questionnaires and interviews of the owners' interests, 6 main priority problems occurred, which group needs for capacity developing for conforming to CBD and Cartagena Protocol integrally. These priorities concern the fields of legislation, institutional, education, financial, science and public support. They group all requirements of the CBD, Cartagena Protocol and the two UE directives regarding nature conservation. It is certain that, by approaching integrally the three Conventions for global implication, the accomplishment of their requirements demands mainly the same capacity functions:

- Capacity to formulate intersectorial policies by respecting the proportion between availability and needs, dimensioning adequately the capitalization of renewable and non-renewable resources that they don't spoil the nature and pawn the future of our descendants.
- Capacity to implement the present regulations by imposing sanctions where necessary, but also the capacity to apply other instruments of market economy, as complementary policy of the one "command and control".
- Capacity to draw funds, mobilize amounts of money courageously in the prospect of their multiplication and the renouncing to the allocated funds expecting behavior.
- Capacity to democratize continuously the process of decisions making by informing, communication and public participation.

Remote sensing and biodiversity assessments: On what scales and on which organisational level (genetic diversity, species richness, ecosystem diversity) are remotely derived variables useful predictors/indicators for biodiversity?

Harini Nagendra, Center for the Study of Institutions, Population, and Environmental Change (CIPEC), Indiana University, USA and Ashoka Trust for Research in Ecology and the Environment, Bangalore, India.

SUMMARY: In the biodiverse tropics, a multi-scale approach that integrates remotely sensed data with field investigations has been very useful for mapping biodiversity at ecosystem, habitat and species levels.

Advances in the spatial and spectral resolution of sensors have made it increasingly feasible to map biodiversity distributions. Yet this task remains most difficult in the tropics, where the spatial and temporal dynamicity and complexity of biodiversity distributions present a challenge of altogether different magnitude. Our research in the Western Ghats biodiversity hotspot in India has integrated remote sensing with field assessments for mapping biodiversity at multiple spatial and organizational scales (Nagendra and Gadgil 1998, 1999 a and b; Nagendra 2001, 2002; Nagendra and Utkarsh 2003). We utilize a nested hierarchical classification of ecological entities from biosphere to individual organisms. This assumes the existence of certain relationships between organizational scales. Entities at any level in this hierarchy (e.g. vegetation types) should be capable of differentiation in terms of the composition and configuration of their constituent entities at the next lowest level (e.g. species). Furthermore, indicators of richness at higher organizational levels (e.g. landscape diversity) are potential surrogates for biodiversity at the next level (e.g. species diversity).

Our investigations were conducted at three nested scales that differed in grain, extent and ecological organization. The first scale encompassed the Western Ghats and western coast of India. We utilized the Normalized Difference Vegetation Index (NDVI), correlated with green biomass, to map this ecoregion covering more than 170,000 km2 into 205 patches of 11 ecomosaic types. Ecomosaic patches ranged from rare high montane evergreen-grassland complexes, to frequently encountered complexes of degraded forest, plantation and agriculture, and varied in size from 100-10,000 km2. This map was used to locate 13 representative sample landscapes, ranging in size from 10-50 km2, in dominant ecomosaic types. 24 vegetation types were identified in these landscapes, ranging in size from 103-106 m2. Finally, these vegetation maps were used to locate 1-100m2 field plots to sample the distribution of plant species. We were able to establish linkages between different scales of information collection. Landscapes belonging to different ecomosaic types differed significantly in spatial pattern metrics known to impact species distribution – including landscape diversity, patch shape and inter-patch distance. In turn, vegetation types identified based on supervised classification of imagery significantly differed in species diversity and in species composition. A high degree of field input was essential for image classification, and ‘rapid’ unsupervised assessments derived without field training did not work well. Neither remote sensing nor field investigations are sufficient in themselves, but together they provide a powerful, efficient and cost-effective tool.

At this resolution, we were unable to establish direct correlations between the differences in spectral intensities and species composition of vegetation types. However, given the advent of new sensors with increasing spatial, spectral, temporal and radiometric resolution, attention needs to be paid on using this increasing amount of information to directly map biodiversity distributions at species and possibly even genetic scales (Bawa et al., 2003). Another new area of investigation is the development of methods to integrate remote sensing and geographical information on key institutional, biophysical and ecological indicators that influence the distribution of biodiversity, and to model and extrapolate the information so obtained in both time and space (Nagendra et al. in press-a, in press-b, 2004, 2003; Schweik et al. 2003). Through partnership with the DIVERSITAS Core Project on bioDISCOVERY, we are engaged in the development and validation of such approaches for the tropics.

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On the use of morphospecies for rapid biodiversity assessments: Potential and pitfalls?

Peter Duelli, Swiss Federal Research Institute WSL, Birmensdorf, Switzerland

SUMMARY: Using morphospecies (or parataxonomic units) for rapid biodiversity assessment is a valuable and economic method for a very special purpose: comparing and monitoring species richness. It is useless for qualitative assessments of species composition and conservation value.

Biodiversity evaluation is a priori hampered by the fact that only single aspects or entities can be measured with reasonable effort and costs. In fact, most current large-scale evaluation schemes do not even quantify biodiversity itself, but surrogates of biodiversity. An impressive example is the measurement of the number or extent of agri-environment schemes in agriculture, where it is assumed that they are positively correlated with biodiversity. If they are not, as in the case of the Netherlands (Kleijn et al. 2001), the surrogate indicator value is misleading.

Moreover, we are always faced with the compromise between spatial representation (e.g. for national inventories) and compositional (taxonomic) representation. We can either cover large areas with a stochastic sampling scheme, or we can cover a large taxonomic spectrum – both together would be impossible, because the costs for the identification of the numerous species would be prohibitive.

In such a situation it is tempting to resort to counting morphospecies, i.e. not to come up with species names, but simply count the piles of specimens that look as though they are the same species. That approach was termed Rapid Biodiversity Assessment by the Australians (Oliver and Beattie 1996), who sometimes have the problem that their species are not yet described. The method is of growing importance in practice, whenever the required goal is species numbers only, but the limitations are obvious (see also Krell, 2004):

- All species, big or small, count for the same: there is no differentiation with regard to conservation value, category of threat, pest status, profit, etc.
- No accumulation of species lists is possible, nor is calculation of beta- or gamma-diversity.
- No similarity studies, no temporal or spatial species turnover can be measured.

On the other hand, if only species richness is the target issue (which in fact is very often the case in monitoring schemes and comparative risk assessments), Rapid Biodiversity Assessment with a morphospecies (or parataxonomic) approach can enlarge the taxonomic spectrum considerably.

Species richness as such can be used as an indicator for ecological resilience (Duelli & Obrist 2003). We have developed a sampling scheme ("Swiss RBA"; Duelli & Obrist, in prep.), where we have collected flying and crawling arthropods with standardised trap devices since 2000 at 45 localities in Switzerland. Per trap station and year (4 optimised sampling weeks) we have collected 90-450 morphospecies. Fourteen taxonomic groups were counted separately to allow for specific analyses and for calculating rough ratios of e.g. carnivores vs. herbivores, pollinators, etc. We monitored changes in managed forest plots, on agricultural sites, and compared them with control stations in unmanaged ("wilderness") areas. The material is kept for eventual later identification (pest outbreaks, immigrants, etc). We found a slight underestimation of species numbers (y=0.93), and the correlation with true species numbers was R2=0.83, and R2=0.89 without arachnoids.

We consider RBA to be a valid and economic method to get standardised estimates of local species richness. It is a relative measure, most useful for monitoring species richness over time, but it is useless for evaluating qualitative differences or changes in species composition. We suggest using RBA when the financial resources are not sufficient to consider a broader faunistic spectrum. Ideally, RBA should be used in addition to inventories of identifiable but species-poor groups such as birds, butterflies or grasshoppers.

Re: Morphospecies for RBA: some statistical comments

Keith Rennolls, University of Greenwich, UK

Peter Duelli wrote: "Moreover, we are always faced with the compromise between spatial representation (e.g. for national inventories) and compositional (taxonomic) representation. We can either cover large areas with a stochastic sampling scheme, or we can cover a large taxonomic spectrum – both together would be impossible, because the costs for the identification of the numerous species would be prohibitive."

Peter, I do not understand this comment. A random sampling scheme can be stratified according to likely contribution of each stratum to the overall species abundance measure. Also it is usually the case in Sample survey design, that the costs of "measurement" of the various sample units, within strata, (i.e. species identification in this case) will be included in the full cost model of the survey. Then the optimal sampling configuration is determined by maximizing the resulting information (maybe the coefficient of the abundance estimate in this case) for given overall cost, or equivalently to provide the information required at the desired precision, at minimum cost.

Peter Duelli suggests the use of morphospecies counts instead of species richness (with the possibility of using the latter to estimate the former), and lists some of the pros and cons of the approach in the RBA context.

It seems there might be a scale-dependent bias in doing so. I assume that common evolutionary history in the same or equivalent environments (Niche?)(for the same evolutionary time) is the dominant reason for two distinct species being morphologically equivalent. Hence the number of species within a morphospecies will vary with the size (area) of the shared environment/niche. Since the relative sizes of the various environments/niche within any particular target population being inventoried will vary between inventories and regions, the scaling ratios between morphospecies counts and species counts will not be consistent between such studies or regions. Hence it is hard to see how the results of a RBA based on such methods really could be used as a valid quantitative basis for the comparison of species richness between studies and between regions.

Re: Re: Morphospecies for RBA

Peter Duelli, Swiss Federal Research Institute WSL, Birmensdorf, Switzerland

In answer to the comment by Keith Rennolls:

1.) RBA is a "quick and dirty" method for average alpha-diversity in cases where the funds are not sufficient for species identification of a large taxonomic spectrum - which is, I guess not only in Switzerland, the normal case in practice. Given 200 000- EUR a year, you can count birds and plants in stratified samples all over the country. And you assume that these taxa represent local species richness, i.e. correlate with the other 95% of local biodiversity. I bet they don't. Or with 200 000- EUR/a you sample a broad taxonomic spectrum at 5 stations for several months. And you assume that the five stations represent average local biodiversity in the rest of the country. Which I bet they don't. Obviously, both measures are far from representative for average local species richness. Even with all the scatter and systematic errors, RBA certainly is a much more reliable estimate of average local species richness; for 200 000- EUR a year you run at least 100 stations and record an average of 260 taxonomic units per station, plus the flowering plants.

2.) Given the above mentioned scatter and systematic errors (which can be calculated) it really doesn't matter if two species are looking more similar in one habitat than in the other. Anyway, the RBA-approach is a relative measure, most useful for monitoring changes in time, not for comparison between different habitat types. But indeed, we realized that with higher species numbers (> 300) the morphospecies approach underestimates species richness, most likely because then chances are higher that two species look very similar.

As I mentioned in a former e-conference: It would be great to have a European test area for comparing the performance of competing biodiversity indicators! Maybe for the Countdown 2010?

Re: Re: Re: Morphospecies for RBA

Alan Feest, Water and Environmental Management Research Centre, University of Bristol, UK

In answer to Peter Duelli

1. I too think that only by defining closely the study area can we extrapolate from the known to. Sampling processes are paramount in obtaining good information. The problem is that one single site will not necessarily allow a scaling up of the conclusions. Take the following case that we have researched in the last year. A large woodland site had at its core a modern coniferous plantation and this was surrounded by deciduous woodland (Oak, Beech and Ash). The new owner wished to convert the coniferous woodland to an open space as it is also an Iron-age Fort. We were asked to establish baseline biodiversity indices of the macrofungal species evident this year. The result was emphatic; almost all of the interest and biodiversity was underneath the coniferous trees. The conifers also had Goshawks nesting. For most other taxa the deciduous woodland was superior in biodiversity. So what can we say about biodiversity? Certainly only by getting the scale right would the whole biodiversity quality interest have been understood and yet these were two different habitats.

2. Our work on biodiversity quality has included using a range of "biodiversity indices" and I can say that of all of the indices the Shannon-Wiener is the least informative! Simpson's gives a better result as it has a wider amplitude and Berger-Parker Dominance is a lot simpler to calculate and nearly always reflects the other two closely. Species Richness can be misleading in biodiversity quality as all of the species may be common and ubiquitous so other measures are needed and I have used a a "Species Value Index" to great effect. This can reflect the RDB status of species or any perceived scale of rarity. If individuals are counted it is also possible to obtain a biomass index. Combining these allows biodiversity quality to be indicated. If the survey is conducted properly it should be possible to estimate the total number of species by the Chao formulae but given the above evidence even total number of species will not indicate the whole of the biodiversity quality.

3. The EEA (EU) programme to reduce biodiversity loss by 2010 is a very laudable aim but given that somehow the decision has been made that this should be based on historic evidence and that no unification of sampling process is intended are we going to find that scaling problems are totally confounding any conclusion. Is this going to be another case of "good money chasing bad science"?

Morphospecies concept or monitoring of morphotypes? The use of aggregate taxa

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Götz Heinrich Loos, Biological Station of Western Ruhrgebiet, Oberhausen, Germany

In floristic botany, so-called "aggregate taxa" or "collective species" are used. These parataxonomic units are very useful as it is obvious from a priori that there are more than one taxon implied and as they are not treated as species (the term "collective species" is a priori not a taxonomic species term). Aggregate taxa can also contain subspecies, varieties and forms (see Loos 1997). The aggregate concept is a "quick and dirty", too, but it seem to be more honest. For zoological questions, Krells "parataxonomic units" could be as useful as aggregates in floristics.

Taxa are also not really existing entities, but they should be used as "standard" terms with clear-cut definitions, while aggregates and "parataxonomic units" are less clear-cut and only operational, but not metatheoretic conceptional.

Choice of Biodiversity Index and other issues....

Keith Rennolls, University of Greenwich, UK

Alan Feest, in responding to Peter Duelli discussed some specific biodiversity indices, including Species richness (abundance), Shannon-Weaver Entropy/Diversity, Simpson's Index, and the Berger-Parker Dominance Index, and expressed his preferences based on his own experiences and objectives.

Presumably other Biodiversity researchers might have other experiences and objectives and hence will have other reasonable preferences. One might ask how to proceed if one does not have any prior preferences. There is a very obvious and clear answer, do all the indices, and see what they say about the data. However, this might seem rather arduous without software support, (which does exist, but I have not bothered to track references of sources for this contribution Chao has a package Estimate S; see the web reference given below). There is however another side to this approach, since all the above mentioned indices can be subsumed to a single functional diversity index. Equivalent equations for such an index have been presented for ecological analysis independently by Renyi (1961) and by Hill (1973). A functional diversity index on a single dataset may be presented as a graph of Diversity of a dummy parameter, (alpha say), (Orloci, 1991) . Then particular values of alpha correspond to all of the various specific indices mentioned above, and more, of course. (Alpha= 0(Richness), (1) Shannon-Weaver, (2) Simpson, (infinity, but about 3 is enough) Berger-Parker.

If one has multiple sites that have been assessed then one can examine the alpha-diversity plots and it can be fairly obvious which of the various indices are most effective in discriminating effectively between the sites and their "diversity types". Hence Analysis of biodiversity data, though diversity indices can become data driven rather than preference driven. Rennolls and Laumonier (2000) sample the Renyi diversity curves for 20 values of alpha between 0 and 3, and do a PCA based ordination of the sample sites. For their particular data, (sub-plots of a 3ha plot of tropical rain forest at Batang Ule, Sumatra), that TWO dimensions are all that are needed to adequately summarize the diversity data, and also to adequately (and sufficiently) distinguish between sites in terms of species diversity. This idea, that of the dimensionality that is appropriate to a diversity dataset collected from multiple sites has not been considered much (or at all) in operational biodiversity studies. It might be fruitful to do so in the hope of characterizing "the whole of the biodiversity quality".

Alan mentions the Chao Estimator of Species Richness. A references is Chao (1984). However, Alan's qualifier "if the survey is conducted properly" for the use of the Chao estimator to be valid is not the best perspective, in my view. Many estimators are actually based on rather idealized and impractical theoretical assumptions, and the survey practice could never really match them, and in fact should not try. The sampling of biodiversity data should be done first to obtain wide coverage, and as much representative (as well as extreme) relevant information as possible, and the estimation theory should try to move towards to the practice rather more than practice should try to move towards the theoretical assumptions.

I would mention that the jacknife estimators (of various orders) are somewhat similar to the Chao estimators. For an interesting modern review of abundance estimation see Chao (2004). I would also mention that Rennolls & Laumonier (1999a) introduce a new estimator of species richness, obtained by regarding the sample as a variable probability sample of species, and use of the Horwitz-Thompson estimator. Essentially this method estimates the number of unseen species (termed virtual species in the Rennolls and Laumonier paper) from the number of observed species (and their frequencies). The estimator has a common sense interpretation, not something that can be said about the purely technical Chao and jacknife estimators. The Good coverage estimator also provides a crude estimate of the number of unseen species, (Good, 1953) Chao and Shen also uses the H-T estimator on species (with an adjustment based on the Good coverage estimator), but to estimate the Shannon-Weaver diversity in the population (Chao & Shen 2003) . Rennolls and Laumonier (1998) actually apply the H-T estimator to the Renyi alpha functional statistic, and in so doing are analysing multiple diversity views of the (estimated)population diversity structure. A paper following on from Rennolls and Laumonier (1998) is currently submitted to JTE on the (coverage-adjusted) H-T estimators of species richness in particular, and Renyi's alpha diversity in general, and the associated ordination of the "diversity space".

While mentioning references, in a previous contribution to this e-conference I mentioned the importance of extrapolating diversity indices on diameter-measurement threshold as well as on sample area. A reference in relation to this is Rennolls & Laumonier (1999b). I have to say, in concluding this rather technical contribution, that my impression of the whole Biodiversity assessment area. (this e-conference included), is it that it seems not to have a solid foundation in terms a widely accepted coherent and deep views of the various estimation issues (both sampling and estimation) associated with Biodiversity assessment, (across scales). There is obviously a lot of data being collected. But how much of it is open access so that the results presented can be subjected to real scrutiny. But in addition to sharing of data, we need to make sure we share theory and methodologies(the Chao site is a good resource bases here, and there are others). However, these two possible pooling activities, of data and theory should not be conducted separately, by different people. They both need to inform and influence each other. Hence, I hope, if this e-conference were to lead to the proposal for shared or joint studies/data that biodiversity theorists, biometricians, modellers should also be included in the enterprise.

Renyi biodiversity & scaling

Gabor Lovei, Department of Crop Protection, Research Centre Flakkebjerg, Slagelse, Denmark

I fully support the recent contribution by Keith Rennols. One small clarification, perhaps: I have also used the Renyi -curves to compare diversity in different insect assemblages and found that to analyse the profiles up to alpha=3 is not always sufficient. The mutual relationship between assemblages is not obvious, in cases, up to alpha=16, so I usually calculate the profiles to alpha=20.

A new version of the program to calculate different generalised diversity functions, written in R, has recently been published (Tothmeresz 2005).

Scaling up: Estimating numbers of species from probability of finding species in samples

M.G. Chapman and **A.J. Underwood**, Centre for Research on Ecological Impacts of Coastal Cities, Marine Ecology, University of Sydney, Australia

SUMMARY: For most sets of species, particularly invertebrates and other small cryptic taxa, the number of species in any site/habitat can only be estimated from the numbers that one gets in a sample (of n replicates) taken at some spatial scale in that site/habitat. Here we explore methods to scale up the estimates of numbers of species from a set of samples (about 1.25 m2) to a site (5 m x 4 m) to a whole rocky shore (say 150 m x 50 m).

If one knows, for any single site (i), the true number of species (Ni) and the number one gets in a sample of a particular size (ni), one can calculate the probability of finding a species in a sample of size n (p = ni/Ni) and use this probability to calculate the number of species in any other site (j) from any similar sample (Nj= Ni/ni*nj). This procedure assumes (a) the relationship between Ni and ni is consistent among times of sampling and among sites and shores (i.e. that spatial relationships among species do not change), (b) that n is large enough to minimise sampling error and (c) that the probability of finding a species in a sample is well-measured by whether a species is or is not found in a single sample.

It should be possible to improve this estimate by taking several samples (say 4) in a site. These can be used to estimate the probability of a species being found in all 4 samples(p4), 3 of them (p3), only 2 (p2) or only 1 sample (p1). These can then be used in other sites to estimate the total number of species in the site, based on the number found in a sample. This sort of method takes into account the ways different species are distributed (spatial variance) and therefore their frequency of occurrence in samples. It should provide a more reliable estimate of the relationship between the true number of species and the number sampled. One can predict therefore that this method should under- or overestimate the true number of species to a smaller degree than a method that assumes all species are equally likely to be sampled. There is, however, more effort (and thus cost) in establishing the relationship between Ni and ni using multiple sets of samples. If this is substantial without much gain in accuracy, then it may not be worth the additional effort.

This was evaluated by examining relationships between N (number of species found on 4 intertidal rocky shores), Ni (number of species per shore), Nij (number of species in each of 4 sites on a shore) and nijk (the number of species found in a set of replicate samples in each site on each shore).

Preliminary results indicate that there can be substantial error in scaling up from samples of organisms on rocky shores to estimate numbers of species. Using the first method of calculating probabilities, the number of species was under- or overestimated by > 20 %, whether one was estimating species richness at the scale of sites or the entire shore. Although increasing the number of samples from which the probabilities were calculated decreased the magnitude of the over- or underestimation, this decrease was not very large, even when the effort was increase more than 4-fold.

As expected, the second method of calculating the probabilities estimated the number of species more reliably, at the scale of sites and shores. Nevertheless, a substantial increase in reliability seems to require very large sample sizes.

Re: Scaling up

Rainer Waldhardt, Justus-Liebig University, Giessen, Germany

In a way, the outlined upscaling- approach ‘Estimating numbers of species from probability of finding species in samples’ reminds me of our approach to estimate the plant species richness in a mosaic landscape (published last year in Landscape Ecology 19: 211-226; AUTHORS: R. Waldhardt, D. Simmering, A. Otte, Division of Landscape Ecology and

Landscape Planning, Justus-Liebig-University Giessen, Germany; http://dx.doi.org/10.1023/B:LAND.0000021722.08588.58).

Please, note the abstract of our publication in Landscape Ecology 19: Traditional agricultural mosaic landscapes are likely to undergo dramatic changes through either intensification or abandonment of land use. Both developmental trends may negatively affect the vascular plant species richness of such landscapes. Therefore, sustainable land-use systems need to be developed to maintain and re-establish species richness at various spatial scales. To evaluate the sustainability of specific land-use systems, we need approaches for the effective assessment of the present species richness and models that can predict the effects on species richness as realistically as possible.

In this context, we present a methodology to estimate and predict vascular plant species richness at the local and the regional scale. In our approach, the major determinants of vascular plant species richness within the study area are taken into consideration: These are according to Duelli's mosaic concept the number of habitat types and of habitat patches within area units. Furthermore, it is based on the relative frequencies of species within habitat types.

Our approach comprises six steps: (i) the determination of present habitat patterns within an observation area, (ii) the creation of a land-use scenario with simulated habitat patterns, (iii) the determination of species frequencies within habitat types of this area, (iv) a grouping of habitat-specific species, (v) the estimation of the probabilities for all species (or habitat specialists) to occur, either in stepwise, exponentially enlarged landscape tracts (local scale), or in the entire observation area (regional scale), and (vi) the validation of the estimated species numbers. The approach will be exemplified using data from the municipal district of Erda, Lahn-Dill Highlands, Germany.

Species richness, area and sampling

Philip Roche, University Paul Cezanne, Institute of Mediterranean Ecology and Paleoecology, France

Scaling problems in sampling of community are major problems that ecologists have faced for several decades. If we focus only on the problems of the species number as an estimate of biodiversity (species richness), there are still plenty of pitfalls. One of the first problems is to delineate/identify the entity being sampled (i.e. a community, a habitat, a vegetation patch, etc.) and then determine a correct sampling effort (both in time and space).These two points, the structure sampled and the sampling effort are obviously clearly related.

Determining an adequate sampling effort for a given community is a tractable problem that can be solved using SAR (Species Area Relationship) or species sampling intensity relationship, and then from a pre study or from literature determine a minimal sampling area. Once this stage is done, one can use some species number estimator in order to improve species number sampling (Jacknife, Chao...). A much more delicate problem is to scale up sampling done at one scale to larger scales (i.e. from 10m2 -> 1ha). Some attempts have been done using SAR and models used to fit to the observed SAR. Here some attention must be taken as to the method used for creating the SAR. Two main methods are used: 1/ hierarchical nested sampling (true SAR), 2/ Species accumulation curves obtained from resampling techniques of several fixed size samples dispersed within the community or from several similar communities dispersed within a landscape or a region (SAC) (see Ugland et al. 2003, Thompson et al. 2003, Flather, 1996).

The shape of the SAR can vary according to several points: is the area sampled extensible to infinity? Is the species richness potentially finite (theoretically or practically)? Is the SAR shape contingent of area or of sampling protocol? In the case of infinite species richness and infinite area; the SAR would not have asymptote and thus power models or exponential models can be used (they have been the more commonly used up to today without theoretical ground most of the time...). Recent results indicate that within a given region, both species richness (S) and area (A) cannot be considered infinite, thus sigmoid models are best (logistic function, Weibull cumulated function, beta-P cumulated function ...) (see He and Legendre, 1996; Williamson, 2001; Lomolino, 2000, 2001; Thompson et al. 2003).

Once all these problems are more or less solved can we still used these models to extrapolate between scales? I don't think so. First, the most used SAR are in fact SAC (for practical reason, SAR are very time consuming and somewhat difficult to do when area is getting large (>400 m2). SAC results from dispersed samples, explore a greater environmental variability than the nested protocol and as a consequence overestimate actual species number of a one block area. Things get worse when the number of rare species is large and beta diversity between samples is large too (Le Mire-Pecheux 2004, PhD). Secondly, SAR are not observed for all taxa : Species richness of Carabs is correlated to isles area, but not forest plant species or landsnails (Nilsson, 1988), Bird and bat species richness is correlated to area but not for reptiles and amphibians (correlation with environment diversity) (Ricklefs, 1999). Species richness of native plant species is related to area but not species richness of introduced species (Granados et al., 2001). Thirdly, when extrapolating through scales we must acknowledge that the nature of the object under study changes. For small scales, we may accept that we remain within a given community, but for larger scales, it's no longer a community that is being sampled but a landscape (i.e. a mosaic of community and habitats) and at higher scales a Region. The ecological factors constraining and determining species richness change accordingly:

- At small scales: microenvironment, species interactions
- At intermediate scales: mesoenvironment, land uses, successionally dephased communities
- At larger scales, climate, species migration patterns, biogeography …

Area and habitat diversity both contribute to species richness (Rosenzweig, 1995). But since things are never simple, the respective effect of both causes vary according to the ecological requirement of species: "Presumably, habitat generalists are less sensitive to habitat diversity than are habitat specialists "(Ricklefs & Lovette, 1999. Simberloff (1988) considers that large areas host more species not because they are large, but because they have more habitats than smaller ones.

At the landscape level, the species richness might depend mainly on landscape structural patterns (Duelli 1992; Wagner et al. 2000; Ortega et al. 2004). Species richness is a function of area and landscape complexity. Nevertheless at this level, few studies propose convincing solutions: Roy and Tomar (2000) proposed to use landscape diversity as a surrogate for species richness, Tjorve (2002) proposed to compute SAR models for habitat mosaics (but consider only 3 habitats), Le Mire Pecheux (2004) proposed to use a hierarchical model including SAR model for community and Bayesian prediction of species occurrence to predict species richness at landscape level, but SAR proved to be unreliable. Roche et Le Mire Pecheux (under revision) proposed to use landscape complexity and ecological disparity of habitats as surrogates of species richness. Guisan & Theurillat (2000), Araujo & Williams (2000), Segurado & Araujo (2004) used regression models (GLM, GAM) in order to predict species occurrence of a large number of species but failed to estimate species richness because of the small number of species used.

Scaling problems in biodiversity assessment: Summary Week 1

Ariel Bergamini and **Christoph Scheidegger**, Swiss Federal Research Institute WSL and **Lisandro Benedetti-Cecchi**, Dipartimento di Scienze dell'Uomo e dell'Ambiente, University of Pisa, Session I Chairs

This first week of the e-conference was characterised by a multitude of different contributions. In the following, I try to shortly summarize to most important topics.

One topic in particular seems to be of outmost importance, namely how to estimate biodiversity (presumably mostly seen as species richness) in a larger region. That means, how to scale up from plots to the entire habitat or landscape (terrestrial or marine), or in more general terms, to the system. Approaches reached from linking remotely sensed data with field investigations, to models using the detection probabilities of species in small plots to extrapolate to a larger region, to the development of scaling functions. It was emphasized to use the fractal dimensions of different systems to monitoring system health. Furthermore, to develop a cost-effective monitoring programme, it was putting forward, that theoretically, it may be useful to determine the degree to which an ecological system is scale-invariant. Martin Sharman emphasized that to improve our understanding of biodiversity, a concerted effort is necessary to understand the implications of the topic of emergent systems.

Other contributions discussed the usefulness of different indicators. Given the constraints of the high costs of a biodiversity assessment of a larger region it was emphasized to focus on the most important species such as red listed taxa. Other approaches, which were critically discussed, included the use of higher-taxon richness and the use of morphospecies as indicators of species richness. However, while both approaches may be very useful in some circumstances, they may also have severe shortcomings. For both approaches, it seems unclear how they are affected by spatial scale. Higher-taxon richness was presumed to be quite useful at regional scales, but not at the global or local scale. But, the approach may critically depend on the evolutionary history of the studied region. For example, it may not be useful on islands with very few higher-taxa. The use of life forms as a geographically independent indicator of habitats which in turn could be a proxy of biodiversity was also emphasized. Although the concept of umbrella species was criticized, it was also stressed that using different umbrellas at different scales may be a valuable approach.

Focusing on the genetic level of biodiversity it was recognized that genetic diversity is still strongly neglected in biodiversity assessments. Thus, a strong need was recognized to search for spatial patterns in genetic diversity, for correlations with species and habitat diversity, and to find the key factors that allow us to explain how these patterns have evolved.

Furthermore, it was also stressed that we should begin to use the large amount of data sampled so far (Red Lists, diverse monitoring programs) and that we have to develop methods to use the many different sources of biodiversity information to monitor biodiversity.

Next week, there will be keynotes on patterns of biodiversity within regions and on the comparison of biodiversity between regions. Topics such as 'optimal' grain size, rarefaction and alpha/beta diversity patterns will be discussed. I am looking forward to thought-provoking, lively discussions!

Patterns of biodiversity and grain size: are there any general rules for grain size in biodiversity assessment?

Carsten Rahbek, Center for Macroecology, Zoological Museum, University of Copenhagen, Copenhagen, Denmark.

SUMMARY: We know today that macroecological patterns and the influence of processes on these is highly scale-dependent. For a thorough discussion of these aspects, I will refer to the recently published, attached, review-paper (Rahbek, 2005). Given the title provided to me by the organizers this essay deals with conservation aspects of scale effects — with a focus on area-selection. However, I like to stress as forcefully as possible, that if we are to manage biodiversity globally and nationally, we must embark on the challenging enterprise to describe biological patterns, reveal mechanisms, and put processes on the map to achieve effective identification and management of areas of importance to biodiversity.

Today, we have a wide range of powerful tools in terms of quantitative techniques for assessing conservation priorities based on the principle of complementarity (Pressey et al., 1993; Williams, Burgess & Rahbek, 2000). Area-selection techniques have been applied to identify conservation networks at a variety of different spatial scales, both in terms of area extent (e.g., local, regional, continental) and grain size of selection units (see Pressey & Logan, 1998 for a review). Recent studies have shown that spatial scale in terms of area extent (Erasmus et al., 1999; Rodrigues & Gaston, 2002) and grain size of selection units (Pressey et al., 1999; Larsen & Rahbek, 2003; Warman et al., 2004) can influence conservation priority-setting.

At first glance, one would think that the identification of priority areas for conservation should rely on data (e.g., distribution) at the finest possible geographic resolution (grain size) in order to provide the best possible guidance for the identification of actual reserves on the ground. The smaller the grain size of selection units, the more efficient (in terms of amount of area required) one may select a network of areas that provide the best species representation. However, choice of selection units (grain size) is not that straight forward:

- There are several biological reasons why larger biodiversity-managed areas are preferable to smaller areas in terms of long-term persistence of biodiversity
- Many biological processes (e.g., speciation, meta-population dynamics), environmental services (e.g., freshwater quality and quantity, nutrient cycling incl. carbon uptake and release) and landscape values require a certain size to operate, be maintained and captured.

There are also important practical considerations. Global and continental, even national and regional, priority-setting is limited by the lack of reliable distributional data for most groups. Priorities are currently based on a subset of taxonomic groups, such as vertebrates (Brooks et al., 2001; Balmford et al., 2001), birds (Stattersfield et al., 1998), subsets of plants (WWF & IUCN, 1994–1997) sometimes combined with habitat loss (Myers et al., 2000). Using fine-grain sizes as selection units prevent the incorporation of biodiversity data for the organism groups that constitute the vast majority of species on Earth (e.g., insects) simply because our knowledge about these groups are so “coarse-scaled” and fragmented.

So perhaps using a larger grain-size at an extent of scale, where we do have information about these organisms provide a more preferable balance between the lack of geographical precision (due to coarser grain size) and better taxonomic precision (due to the inclusion of more phylogenetically and ecologically divergent taxa) than archived in the currently vertebrate-dominated broad-scale approaches to conservation priority-setting. An alternative approach is perhaps the higher-taxon approach (see Larsen & Rahbek, In Press for a discussion of scale effects using that approach)

We simply do not know where the fulcrum of that balance is. Our understanding of the effect of scale on priority-setting is still rather limited both from a theoretical perspective and in terms of insight gained from empirical studies, especially when it comes to the assurance of persistence of biodiversity.

One thing we do know: there are no universal guidelines for deciding on the extent of scale and grain size when designing biodiversity assessment. The best choice depends on the nature of the question asked and the focal region and taxa. In studies depending on information on species distribution, a rule of thumb could be to use a grain size as small as the smallest range size among all the species to be included within the study area. But often it is impractical – biodiversity is a discipline in a desperate need of more hardcore "on-the-ground" information. Following the general guideline from macroecology (Rahbek & Graves, 2000; Rahbek & Graves, 2001), ideally, the scale of analysis (extent and/or grain size) can be varied systematically from the original scale (finest possible) to coarser scale in order to obtain the optimal resolution of pattern in results for a given analysis. Re-sampling procedures of data have much un-explored power to unravel insight vital for successful biodiversity assessment and a more thorough understanding of the effects of scale.

Rarefaction as a unifying concept for quantifying biodiversity

Nicholas Gotelli, Department of Biology, University of Vermont, Burlington, USA

SUMMARY: Rarefaction methods allow for rigorous quantification of many biodiversity patterns at the landscape scale.

Ecologists still struggle to quantify landscape patterns of species richness and evenness and alpha, beta, and gamma diversity. However, most diversity metrics are sample-size dependent and do not have a solid statistical footing (Magurran 2004). Diversity can be understood in terms of a rarefaction curve, which plots the number of individuals on the x axis and the number of species accumulated on the y-axis (Gotelli and Colwell 2001). Although these sampling curves have been in the literature for decades (Sanders 1968), only recently has a comprehensive framework of biodiversity based on rarefaction emerged (Olzewski 2004).

A useful measure of species evenness is Hurlbert's (1971) Probability of an Interspecific Encounter (PIE), which itself is the complement of Simpson's (1949) D. This index of evenness turns out to be the slope of the rarefaction curve measured at its base (Olzewski 2004). PIE is sample-size independent and measures an aspect of diversity that is distinct from total species richness (the asymptote of the rarefaction curve). Lande's (1996) partitioning of alpha and beta diversity can be realized by comparing PIE for an aggregated set of samples to the average PIE calculated for the individual samples. Differences between these two curves quantify beta diversity, the change in species composition among patches. The same strategy can be used to analyze species-area curves and diversity at the landscape scale, because patches of different area accumulate individuals and contribute to within and between patch species richness (Brewer and Williamson 1994).

Although rarefaction analysis requires data on abundances, the statistical framework has recently been extended to accommodate incidence-based presence-absence data (Colwell et al. 2004). Finally, a family of asymptotic estimators can be used to estimate total species richness (Colwell and Coddington 1994). Whereas rarefaction involves interpolation of data to smaller sample sizes, asymptotic estimators require extrapolation beyond the limits of the sampled data. For this reason, variances associated with asymptotic estimators may be large, but they are still the best approach for trying to estimate diversity in speciose taxa that cannot be sampled exhaustively (Longino et al. 2002).

The traditional measure of species richness is species density, which is the number of species per unit area (James and Wamer 1982). However, species density is actually the product of species richness (species number/number of individuals) and total density (number of individuals / unit area). Rarefaction allows one to decompose these elements and to understand the contribution of both species richness and total density to observed patterns of species density. Although ecologists have rarely paid attention to the distinction between species richness and species density, the difference between these two metrics is profound, and they may yield completely different answers for the same data set (McCabe and Gotelli 2000). Rarefaction and sampling curves provide landscape ecology with a more solid footing for quantifying diversity patterns at multiple spatial scales, and for understanding the effects of abundance and area on biodiversity measures.

Biodiversity intactness index: independent of scale?

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

I may have missed a reference in this conference to the recent (3rd March) article by Bob Scholes and Oonsie Biggs (Nature 434 45-49) entitled "biodiversity intactness index". In the introduction to their paper, they echo much of what has been said in this discussion concerning the "complex, multidimensional nature of biodiversity, which can be defined in terms of composition, structure and function at multiple scales". They remind us that the CBD indicators are intended to "be amenable to aggregation and disaggregation at ecosystem, national and international levels". They point out that many methods of measuring biodiversity are scale-dependent, and propose a new index that they claim is scale-independent. The paper is available on-line http://www.scidev.net/pdffiles/nature/nature03289.pdf. For your convenience I shall try to summarise their method, and I hope I do them justice.

Divide the area you are interested in into a small number (say 10 or fewer) land use categories. Find at least 3 experts who are knowledgeable about one or more of the taxa present in the area of interest, and keep finding experts until, taken together, they cover a wide range of taxa. Ask each of them independently to consider modern conditions relative either to those in large local nature reserves or at some historical moment, and for each of their specialist taxa, estimate the changes in population abundance brought about by each of the land uses of concern. Aggregate the results across species (grouped by ecological similarity), ecosystems, and areas of land use. The resulting "biodiversity intactness index" (BII) is, they claim, a sensitive, realistic and useful measure of biodiversity loss that has the same meaning at all spatial scales. In an accompanying comment in the "News and Views" section in the same issue of Nature, Georgina Mace FRS (Nature 434 32-33: http://www.scidev.net/pdffiles/nature/434032a.pdf) remarks that policy makers think of "biodiversity" as the number of species on a list. Scientists, she says, think of "biodiversity" as variability within and between genes, genomes, individuals, communities, traits and ecosystems across many scales of space and time. She states that the BII is a robust, sensitive and meaningful indicator that allows trends in biodiversity to be monitored over time and space.

Can anyone help me to understand why the authors claim that the BII is independent of scale? It seems to me that an index based on land use is inherently dependent on a grain size that can never be smaller than the areas of land used in a particular way. Furthermore, when you interview an expert, you will presumably ask her to consider changes that have occurred in a particular area - say, a country, or catchment basin. To change spatial scale, surely you would have to interrogate her again, asking her to estimate changes at the new scale? A similar remark holds for scales in time. Is an index "scale independent" if you have to re-assess it when you change scale?

Re: Biodiversity intactness index

Alan Feest, Water and Environmental Management Research Centre, University of Bristol, UK

I have now reviewed the two papers you refer to and find that the original paper by Scholes and Briggs is fatally flawed in the following ways:

1. They only include plants and vertebrates (for reasons that relate to the practicality of using past databases) but, apart from amphibians and reptiles, these organisms are far less susceptible to environmental change than are poikilotherms (mostly composed of invertebrates in terms of biomass and species numbers). Amphibians are problematic in that whilst they are poikilothermic their breeding requirements are very specific and these limit

their distribution. Maybe reptiles are possibly the only useful organisms but there are not many of them in the tundra!
2. They use "expert opinion" without any apparent understanding of observer bias which in most informal population estimates are a source of gross error. My own work on macrofungi shows this well where some obvious large species are regarded as much more common than in fact a systemic recording shows them to be.
3. They use species richness data for some of their calculations and then on the next page indicate the insensitivity of species richness as a measure.
4. Their index is susceptible to the overwhelming influence of plants and so they then had to apply a "functional" version of the index to cover this.
5. It seems to consider all species as equal!
6. It will be very slow to respond to change but change is now so rapid that this is a severe disadvantage.
7. They provide a single number index but state that it is clear that a single number is not sufficient for all purposes.

The paper by Georgina Mace is much more sensible and clearly states that the understanding of biodiversity is completely different between scientists and politicians; the latter consider lists of species as being biodiversity.

Georgina states that real data is much to be preferred and that the Scholes and Biggs paper does not rely on real data. I would say that it relies on guesstimates and that this limits its application.

Scaling up and down: aggregation and disaggregation of biodiversity indicators

Allan Watt, Centre for Ecology and Hydrology, Banchory, UK

Earlier in this e-conference, Martin Sharman and Alan Feest discussed the Biodiversity Intactness Index. I don't want to discuss it in detail, although I share Alan's concerns about it, but I'd like to focus briefly on the issue of scaling indicators, which is central to the main theme of this e-conference.

"A user's guide to biodiversity indicators" will be published on 22nd March by the European Academies Science Advisory Council. It includes an assessment of indicators, including the focal indicators selected for implementation in the EU.

One of the questions considered for each of these indicators was whether or not they could be readily aggregated (or disaggregated) to provide meaningful information at a range of spatial scales. The report concludes that most of these indicators can be aggregated or disaggregated. Undoubtedly, many of these headline indicators were selected for implementation because relevant information could be readily aggregated. In some cases, however, the information normally collected makes disaggregation difficult: nitrogen deposition, number and costs of alien species and the marine trophic index were all thought to be indicators where disaggregation would be particularly difficult. Furthermore, it was recognised that in many cases aggregation can lead to the loss of useful information, particularly for local decision-makers.

Reference: "A user's guide to biodiversity indicators" can be found at www.easac.org. The project group that prepared this report was chaired by Georgina Mace and also included Ben Delbaere, Ilkka Hanski, Jerry Harrison, Francisco Garcia Novo, Henrique Pereira, Allan Watt and January Weiner.

An insidious problem: inadequate sampling continues to hamper the identification of local through regional patterns in marine habitats

Sean Connell, Southern Seas Ecology Laboratories, The University of Adelaide, Australia

SUMMARY: I believe that most ecologists are intrinsically interested in the existence of broad scale patterns (and process). Surprisingly, however, many of us work at scales where complexity is greatest (i.e. local) or make premature attempts at generality by comparing vastly different areas with few samples. An unsurprising result, therefore, is that we tend to be captivated by the description of local variation (and publish idiosyncratic patterns which probably represent the outcome of many special and unique events) whilst being pessimistic about the existence of broader patterns.

My central concern is how we space our sampling units (i.e. coverage and distance apart) and place our sampling units (i.e. simple random sampling v. stratified random sampling). Of the multiple decisions we make about initial sampling, these two decisions have major effects on our ability to detect patterns.

(1) On spacing samples: One frequently used, but poorly employed method focuses on nesting samples within successively larger scales that span the range of areas of interest (i.e. hierarchical sampling). While the use and advantages of the hierarchical approach are widely accepted (and debated), there remains much needed discussion on its use in extending ecological knowledge beyond high context-dependency and low predictability in local phenomena (e.g. Noda 2004).

Many broad-scale tests compare replicate sites (sometimes ordered into a hierarchy) between widely separated regions (e.g. 1000s and 10,000s of km apart) with the expectation that similarity between distant regions provides powerful inferences for generality (i.e. between regions not studied). While these approaches create the opportunity for rapid progress, interpretation of spatial generalities are hampered because of lack of insight into the scales and places where similarity ends (e.g. spatial extent of generalities). In many cases, differences are detected between the regions and we are left with little to no understanding of what they represent (again because we have little understanding of the similarity/dissimilarity of those samples to the surrounding regions). Would it, therefore, be better not to advocate comparisons of a few sites that are widely separated, but encourage replication to span entire regions? This way we may be in a better position to understand the spatial extent and nature of generalities (similarities).

A real example. In the search of regional patterns, two universities independently sampled the bio(diversity) of morphology of subtidal kelp (Ecklonia radiata) across Australia (> 5000 km of coast). One team emphasised high local variation and the lack of pattern, while the other emphasised high local variation imbedded within regional patterns. This difference in interpretation cannot be explained by differences in response variables, locations and timing of study (both studies are remarkably similar). Instead, the difference centres on how the regions were sampled. The team that emphasised biogeographic differences made their interpretation from samples that broadly spanned the regions of interest using a hierarchy of four spatial scales, whilst the team that emphasised no differences made their interpretation from locations that were often sparsely separated (e.g. distances were up to 1000 km apart between paired sites).

The message is clear. Large variation across several spatial scales indicates that comparisons among local studies, even if done at several sites within a single location, provide a difficult basis to understand the generality of pattern. This situation arises because the within-region variation has not been adequately estimated by the replicate samples (i.e. too few samples and too narrowly spread), preventing interpretable comparison among regions.

Other marine research continues to support the idea that large-scale patterns can emerge from apparent stochasticity at small scales, and that unaccountable variation at local scales need not impede tests for similar patterns at broader scales. For example, in a study of variation in the diversity of invertebrates that inhabit kelp, Anderson, Connell et al. (in press) revealed a lack of significant variation in the proportional abundances of phyla at large spatial scales and suggest that some consistency of pattern may emerge at larger scales (spatial and/or taxonomic), even in the presence of high small-scale variability. It is encouraging, therefore, to observe that patterns can emerge from complexity at local scales to provide new opportunities to answer some of the more interesting questions about the relative importance of processes across the vast parts of the world's coast. By understanding the proportion of total variation that is attributable to each scale, we are in a stronger position to identify the scales at which general patterns, rules and laws may emerge.

(2) On placing samples: There are costs involved in the comparison of large areas. These costs often sacrifice specific information for breadth and ignore some special feature of the environment which, when taken into account, could improve predictive power. The problem confronting ecology is not whether one should test for the existence of general or specific phenomena, but what balance should be sought between the two and what costs are involved in favouring one aspect over the other. If some local variable is strongly associated with an unrecognised feature of the environment, then tests of broad scale patterns may be compromised.

In conclusion, we are becoming increasingly aware that ecologists are working at scales (i.e. local) where complexity is often greatest. At these scales patterns are likely to represent special and unique events that incorporate variation from broad to local scales. For those interested in the existence of broad scale patterns, it is encouraging to observe that patterns can emerge from complexity at local scales. This realization, together with the need for a renewed effort for carefully planned sampling across broad scales suggests that there are opportunities to test some of the more interesting questions about the relative importance of processes across the vast parts of the world's coast.

On the interpretation of alpha-beta-diversity patterns within regions

Helene Wagner, WSL Swiss Federal Research Institute, Birmensdorf, Switzerland

SUMMARY: Research on the scaling of biodiversity from sampling plots and ecosystems to landscapes or regions should go beyond the quantification of diversity components at different scales and provide ecologically relevant interpretation. The main focus may be on one of the following: (1) testing of ecological theory; (2) testing of the effect of landscape structure on biodiversity; or (3) monitoring of changes over time. Here, I argue for a spatially explicit integration of these approaches.

Ecologists have studied the relationship between local and regional diversity to test whether communities are saturated or proportional samples from the regional species pool. Loreau (2000) showed that for the additive partitioning of species richness, a large alpha compared to beta diversity indicates proportional sampling, whereas a small alpha relative to beta diversity indicates community saturation. Gering and Crist (2002) proposed a quantification of the "alpha-beta-regional relationship" to determine at what scales alpha diversity is limited by local interactions and habitat availability.

The testing of ecological processes is often addressed in a spatially implicit manner, without considering the exact location and the specific landscape context of each unit. However, I think that we can do more than comparing the relative size of mean alpha and beta diversity at a scale of interest. For instance, we could use the mean alpha and the overall beta diversity as a null model that assumes that all units are comparable. We could then try to explain deviations from the expected unit alpha diversity and its share in beta diversity by the characteristics of the unit (e.g., size) and its landscape context. For this, beta diversity in terms of similarity of species composition between units needs to be studied on the level of individual units or pairs of units.

Conservation biology emphasizes the effects of patch size, shape, or connectivity in terms of the number of nearby patches on local alpha diversity. This tradition is strongly rooted in the theory of island biogeography, assuming that patches of the habitat type of interested resemble islands in an ocean of a matrix of non-habitat. In terrestrial landscapes, however, the matrix may reflect various levels of habitat suitability or resistance, so that connectivity is likely to depend both on distance and on the nature of the area separating two habitat patches. A quantification of beta diversity between pairs of samples similar to a resemblance matrix used in gradient analysis, or an integration of diversity components with habitat specificity as quantified by Wagner and Edwards (2001), would allow the spatially explicit testing of alternative ecological processes, such as niche-assembly and dispersal-assembly rules, and provide an avenue for the spatially explicit testing of landscape effects, including matrix effects, at different scales.

Our theoretical knowledge on which processes determine species diversity at different spatial scales and how landscape structure affects the diversity of local ecosystems are paramount for interpreting observed changes through time at various spatial scales. Basically, we don't even need to start setting up a monitoring scheme if we don't know how to relate observed changes to ecological processes and their drivers. However, such interpretation often depends on implicit equilibrium assumptions, which need to be made explicit. This may not be enough, as most terrestrial landscapes are subject to changes in land-use and/or climate, so that it may be increasingly important to understand transient dynamics.

Scale and biodiversity indicators

David Vackar, Agency for Nature Conservation and Landscape Protection of the Czech Republic, Prague, Czech Republic

The concept of scale is somewhat critical to ecology and nature conservation (see e.g. Allen & Hoekstra 1992 or Peterson & Parker 1998). And when talking about multi-dimensional and hierarchical concepts such as biodiversity undoubtedly is, scaling issues are much more critical. Allen and Hoekstra in their excellent book discern between hierarchical levels (population, community, ecosystem, landscape) and scale (i.e. scaling-up and –down), which are often confused even by ecologists. In many cases change in scale requires also consideration of new levels of organization. As a rule of thumb, when using the term scale, we should be able identify physical units of measurement.

Biodiversity indicators as information tools summarizing biodiversity status and trend have to cope with different levels of biodiversity at different scales. Genes, species and ecosystems can be differently scaled in time and space. The basic perspective is an evolutionary one, with time scale of eons and millennia. There are already proposed "ultimate" indicators – e.g. index of evolutionary potential (Santini & Angulo 2001) or indices of phylogenetic diversity. However, evolutionary time scale is sometimes insensitive to everyday decision-making. Also the record about dynamics of extinction and speciation is far from complete. That's the reason why the IUCN, The World Conservation Union, is developing Red List Indices (RLI -Butchart et al. 2004). The index aims to measure genuine shifts in the Red Lists categories, which indicate the probability of extinction in the time scale of decades.

What is the appropriate time-scale for the construction of policy-relevant indicators? There are developed indices based on time-series, such are Living Planet Index or species assemblages trends indices (e.g. common bird index). Are we able to discern trend from the population fluctuation? These questions are not readily answered and require further monitoring of trends with appropriate sampling design and frequency. Even in Europe, deficiency of time-series data exists. At the century time-scale, we can assess biodiversity changes mainly using grid data. However with respect to the 2010 target, abundance based measures should be promoted.

Spatial scale is critical for the sampling design of a monitoring programme that is the prerequisite for indicator construction. Biodiversity assessment is sensitive to sample size and area surveyed. Biodiversity data can be aggregated by land-use or ecosystem type (see for example BII-Biodiversity Intactness Index or NCI –Natural Capital Index). In the process of indicator development, aggregation of fine-grain information into coarser scales of resolution can produce errors. Some changes at the fine scale of resolution can be of very specific nature ("noise"), while some can reflect general trends. The challenge is to find optimal levels and scales for biodiversity indicators to assess changes in ecosystem integrity, health and services realistically. There already exist attempts to find the appropriate spatial scales (BII) and time scales (RLI) for the assessment of state and trends of biodiversity.

Further development of scale-relevant biodiversity indicators, mainly with regard to the 2010 biodiversity target, will require a common venture of scientists, conservationists, managers and decision-makers. Finding appropriate scale for aggregation of information is really the critical issue of indicator construction.

Patterns of alpha and beta diversity within regions: Measurements and tests across space and time

Thomas Crist, Department of Zoology, Miami University, Oxford, Ohio, USA

SUMMARY: Additive partitions of alpha and beta diversity provide an operational framework for analyzing and testing patterns of species diversity at multiple spatial and temporal scales.

Scale-dependent patterns of species diversity are often linked to Whittaker's concepts of alpha (within community), beta (among community), and gamma (regional) diversity (reviewed by Veech et al. 2002). He viewed these components as a multiplicative relationship (beta=gamma/alpha) so that beta diversity was expressed in dimensionless units of species turnover, which is a common view of beta diversity today (Vellend 2001). More recently, Lande (1996) suggested that gamma diversity may be additively partitioned (beta=gamma-alpha) into the species richness or diversity (as measured by an index) found within and among communities. Operationally, alpha diversity may be defined as the average number of species (or diversity) found in a set of sample units or areas, and beta may defined as the average number of species that is absent from a randomly chosen sample (Veech et al. 2002). Hence, beta diversity is expressed in units of species richness or diversity.

If sampling designs are hierarchically scaled, then species diversity may be partitioned across multiple sampling scales in which beta diversity is further decomposed into additive components among plots, habitats, or land uses (Wagner et al. 2000). The value of such an approach to regional biodiversity assessment and monitoring should be clear: sets of habitat areas that comprise the largest beta diversity are those that deserve our attention in formulating land-use practices or in prioritizing areas for protection. In fact, most hierarchical partitions of species richness to date have shown that the largest components of beta diversity are due broad-scale landscape heterogeneity and land use (DeVries et al. 1997, Wagner 2000, Fournier and Loreau 2001, Gering et al. 2003).

Additive partitions of diversity may also inform us about sampling designs or monitoring strategies by identifying the sampling scales that contribute most to beta diversity. Difficulties in the interpretation of additive partitions can arise because observed diversity partitions may reflect the allocation of sampling effort or incomplete sampling. Crist et al. (2003) implemented randomization tests on diversity partitions to determine whether diversity components from multiple sample scales are greater or less than those expected by chance. Recently, Couteron and Pélissier (2004) extended this approach to ANOVA-like comparisons among different habitat types. Further hypothesis testing on diversity partitions may assist in interpretation and identification of habitats or land uses associated with high beta diversity within regions.

Diversity partitioning holds considerable promise in our understanding of the spatial and temporal distribution of biodiversity. For instance, samples taken from multiple time scales may also be partitioned into weekly, seasonal or yearly components, which are important in biodiversity monitoring. Additive partitioning may also inform other approaches used in the analysis of local and regional diversity (Gering and Crist 2002). Recently, I extended the additive concepts of alpha and beta to estimate the fraction of the beta diversity that is explained by the species-area relationship; a meta-analysis of over 100 species-area data sets shows that less than 25% of the beta diversity is due to habitat area. This suggests that habitat and landscape heterogeneity may of overriding importance to regional diversity.

Patterns of rare and common species: congruence, scale and implications

Jack Lennon, Macaulay Institute, Aberdeen, UK

SUMMARY: One of the most conspicuous features of biodiversity is that it varies spatially. It has been know that species richness varies from place to place for a very long time – the latitudinal gradient, for example (Wallace 1876) - yet surprisingly, very little attention has been given to how this variation in diversity is driven by different kinds of species.

In considering this strangely neglected topic, the most obvious point of departure is the question of whether rare or common species drive diversity patterns. The handful of studies comparing species richness of common and rare species have found that they form significantly distinct spatial patterns (Prendergast et al. 1993, Jetz & Rahbek 2002, Lennon et al. 2004). Moreover, the diversity patterns formed from the rare and common species of an assemblage are correlated with the pattern of the full assemblage with different strengths. A recent study of British and southern African bird diversity patterns has suggested an additional feature. A species diversity pattern formed from a number of common species is much more like the entire assemblage diversity pattern than is a species diversity pattern made from the same number of rare species (Lennon et al. 2004). If generally true, it suggests that the multitude of studies examining determinants of diversity patterns have inadvertently concentrated heavily on the common species, and that explanations of variation in species richness may really be explanations of variation in common species richness. Rare species diversity patterns may not be subject to the same set of environmental constraints (e.g. Jetz & Rahbek 2002), mainly because the difference between rare and common species diversity patterns makes a single explanation for both impossible.

The effect of spatial scale, in the sense of sampling resolution, on these differences is an open question. We know that spatial scale can have large impacts on species richness patterns, such that a species richness pattern at one scale can look very different from one measured at another scale, calculated from the same set of data (e.g. British bird data again - Lennon et al. 2001). So, it seems likely that the respective contributions of common and rare species will vary at different spatial scales.

These observations, if generally applicable, have clear implications for biodiversity assessments. First, studies using species richness as a measure of taxonomic diversity will not identify factors uniquely important for the rarer species. Second, given the often arbitrary choice of sampling scale and the scale-dependence of diversity patterns, environmental factors important for diversity at one scale may not apply at other scales. Third, the extent to which species richness reflects the commonness and rarity of the component species may also be scale dependent to an unknown degree. Finally, the complexity inherent in apparently simple distribution data suggested by these analyses indicates that there are some fundamental research questions still to be addressed. The good news is that they are clearly amenable to analysis.

Re: Patterns of rare and common species: congruence, scale and implications

Kajetan Perzanowski, Carpathian Wildlife Research Station, Polish Academy of Sciences, Poland

Jack Lennon is perfectly right in pointing out that diversity patterns of common and rare species are different and that explanations of variation in species richness may really be the explanation of variation in common species richness.

However when we are talking about the loss of biodiversity the common species are not the first to be affected, and I strongly doubt whether short term changes in environment quality can be detected by monitoring common species. Subtle and rapid changes may rather be reflected by altered incidence, abundance or spatial distribution of rare species. By the time we can detect changes in common species, many rare species will probably already be lost.

Therefore, although common species may be responsible for a major part of the variation in species richness, they cannot serve as indicators of biodiversity changes, and protection of biodiversity should begin with the most sensitive rare species.

Scaling problems in species diversity measures

Jim Mallet, UCL London

Studies of biodiversity and biodiversity conservation efforts seem to find it hard to get away from the use of species richness as a metric in comparisons of biodiversity in different areas or different taxonomic or functional groups.

It seems to me that the problem is language. We need words such as "species" to express concepts. However, these concepts may be fuzzy, little different from criteria for overlapping clusters of things. Problems crop up when users of words such as "species" begin to read too much into the concept, and start to perceive the species as a "real thing".

Now of course, I don't mean that robins and blackbirds are not distinct, or that field guides don't work; what I mean is that species are not real in the sense of being connected together evolutionarily, so that there is actually no a-priori, or god-given location in the biodiversity hierarchy where the term "species" must logically apply. It is more like a cake: you cut it where it is convenient to make bite-sized chunks.

Beginning around 1890, until around 1980, a steady consensus had been building among animal biologists (at least) to adopt a more inclusive "polytypic concept" or "biological species concept". If there was some doubt whether two divergent populations, if overlapping, would remain distinct, that is they were suspected or observed to form intermediate or "hybrid" populations, they would be placed often as subspecies within a large Rassenkreis or polytypic species.

Unfortunately, instead of justifying this by a practical argument of consensus or international agreement, Theodosius Dobzhansky and especially the late Ernst Mayr, justified the polytypic species on the grounds that reproductive isolation was the "real" essence of species, so that the absence of blending in overlap was the true, underlying reality of species, rather than merely a cause of species distinctions in overlap, and a convenient stopping place in taxonomy. Nonetheless, this worked fine from the 1940s to about 1980, when Hennigian and other ideas became more prevalent.

Quite suddenly, in the last part of the last century, fixed morphological or genetic differences began to be used to distinguish species, with proponents arguing that the phylogeny is, if you like, the "real" basis of species, instead of reproductive isolation.

We are now in a situation where conservationists, biodiversity specialists, and other users of taxonomy wouldn't be able to tell how many species there are, even if all the taxa were known. There has been recent exponential growth in the numbers of "species" of groups such as primates, while groups that have received less recent taxonomic attention, such as the carnivores, have not increased at all (Isaac et al. 2004, Trends in Ecology & Evolution 19(9): 464-469). Opinions differ as to how much difference this species concept change will make; some estimates give as little as two-fold change, but my belief, partly based on my knowledge of butterfly taxonomy, is that each of the current species can very likely be divided still further, so that an order of magnitude won't be improbable.

In the changes in species richness in vertebrates, there have been relatively few descriptions of new taxa; mostly what has happened is that taxa formerly considered as subspecies have been elevated to species status. The new species are usually although not always) related to the older more inclusive species via hierarchical links. The supporters of the biological species concept and supporters of the more finely-divided phylogenetic concepts both agree on the evolutionary explanation for hierarchy, but merely disagree on the precise level of "real" species.

This seems a strange situation to be in. There are scaling issues here, and it seems likely that the phylogenetic species diversity and the biological species diversity will be approximately, although not perfectly correlated, as has been found in other biodiversity scaling issues, and perhaps it doesn't matter too much which one we use so long as everyone agrees to use the same criteria in comparisons. (But this should be very much under study, as underscored by Jack Lennon's and Jean-Luc Solandt's contributions in this series). The problem is that we can't do this; different groups, and organisms on different continents are currently classified differently.

I don't know what the answer is, but it does seem to me that it might be useful to come to an international agreement about species level and stick with it. The trouble is that there really are a lot of people who feel that their particular reality of species is the correct one and that using some other criterion would be wrong, and would misinform in biodiversity studies. My own personal preference is for a more inclusive, polytypic style species, because it makes sense in two areas:

(1) what speciation is -- it doesn't mean much more than "evolution" if it simply means fixing a new genetic or morphological marker in a separated or partially separated population.

(2) local, alpha-diversity at the species level -- is only enhanced by the ability of two populations to overlap; it doesn't increase just because of genetic divergence across a taxon's range.

Furthermore, there might often be jobs for the more finely-divided phylogenetic species; but these needs could easily be accommodated by the use of the existing subspecies rank in Linnean nomenclature. It would just need acceptance that sometimes there will be paraphyletic polytypic species that have budded off local species that are more closely related one or more of the parent's subspecies -- of course this is anathema to phylogeneticists.

But of course there are exceptions and intergrades here as well. For example, in spite of the fact that we know they hybridize and blend together freely in zones of contact, I have yet to find any modern herpetologist who would designate Bombina bombina and Bombina variegata as subspecies within a species (even among supporters of the biological species concept). So we are always going to have taxonomic anomalies, that may differ in different groups even if we had a strong international agreement to iron them out.

So, even if we had a perfect world where all nations agreed to use the same type of species concept (which I strongly doubt will happen any time soon!), we therefore would still need a second approach:

Hendry et al. (2000, Conserv. Genet. 1, 67-76) have suggested doing away with species in conservation altogether, and using genetic metrics to express biodiversity. While I understand this argument, we are a word-based culture, and words like "species" can be useful even though they do not express underlying realities. I can't see any reason why we can't continue to use the word "species", more carefully, and I strongly doubt that the word will go away in conservation, and especially in the public mind, any time soon.

Instead, we ourselves have to become more educated, and we have to educate all those politicians and the public better, about the fuzzy nature of our most widely-used metric of biodiversity, the species. In our comparisons and counts of biodiversity, we must make this clear at every opportunity, and try to make sure that our analyses in macroecology and conservation science are robust to precise estimates of species numbers. If we fail to emphasise to users the lack of certainty of this particular biodiversity metric, errors will certainly be made.

Species number bashing

Peter Duelli, Swiss Federal Research Institute WSL, Birmensdorf, Switzerland

Counting species numbers really seems to be the most disputed issue when discussing biodiversity evaluation. However, instead of species number bashing, why don't we come up with viable, practicable alternatives? The Countdown 2010 will be an ideal opportunity to propose and compare indicators at different scales: global, national, regional, local. All it needs is a clear statement on the scale and aspect or entity of biodiversity one intends to evaluate, and a realistic proposal for a measurable and affordable indicator.

If nit-picking on proposed indicators was only allowed in connection with a better proposal, we would have a rather calm discussion. And quite a number of the resulting proposed viable and affordable indicators might be based on species numbers. It's always a matter of alternatives!

Re: Species number bashing

Alan Feest, Water and Environmental Management Research Centre, University of Bristol, UK

Thank you for getting us to understand that the politically understandable species lists are not the answer and a more sophisticated approach is needed (Mace, 2005). The real problem of species lists is that they regards all species as the same and they are not!!!! Species have different niches, population sizes, metabolic rates, biomass and frequency both individually and collectively. This adds up to each species and population of species having qualitative characteristics and describing biodiversity without accessing these features in some way is a bit like listing the letters of the alphabet. No real sense in it and the real story is missed!

Practically I measure species relative biomass, relative rarity or conservation value, populations and relative frequency (Shannon-Wiener etc.). I also estimate whether the species list for a site is complete or not and use the Chao formulae to indicate the true number of species. Only by regarding the suite of characteristics is the biodiversity value of a group of organisms described. Different groups are more prominent in different habitats so no one group will serve as universal indicators (despite the political need to simplify things in this way). This work is done on the basis of structured surveys of defined areas; normally less than 10 hectares.

What I want to know is how do I scale this up for a landscape scale biodiversity estimates? My first thoughts are that this is simply an additive activity (adding several surveys together) but this might become too labour intensive to be practical. And this is the problem in scaling from genes to landscape: practically gene analysis will relate only to the population studied and no real confidence can be placed in an extrapolation made from this analysis to a larger scale or other populations. Similarly the practicalities of landscape scale analysis seem to imply a great deal of work and may not reflect the "grain" of variation. Having been a soil microbial ecologist I well understand that two neighbouring crumbs of soil can have a greater difference in the species and populations present than two crumbs a kilometre apart. In soils the grain of biodiversity is microscopic!

Re: Species number bashing

Jim Mallet, UCL London

Alan Feest wrote: "Thank you for getting us to understand that the politically understandable species lists are not the answer and a more sophisticated approach is needed (Mace, 2005). The real problem of species lists is that they regard all species as the same and they are not!!!!" Perhaps I should emphasize that I feel, personally, that "politically understandable species lists" (or counts or estimates of species diversity of some sort) probably ARE the only answer, in spite of their shortcomings. I seem to remember Georgina Mace feels pretty much the same way, and is involved very much with listing species with different levels of threat status. But we should all try to make politicians as well as ourselves more aware of the shortcomings of the listing approach, that advice based on these lists will be for guidance only, rather than an oracle.

"Practically I measure species relative biomass, relative rarity or conservation value, populations and relative frequency (Shannon-Wiener etc.)". Of course these approaches are important as well; evenness is important as well as species richness. But ultimately diversity cannot increase above a certain amount if there are no more species, so extinction seems more important, more final than a reduction in evenness. I think this is why the latest work in this field has mostly gone back, in recent biodiversity studies, to lists of species rather than more sophisticated metrics of diversity.

"I also estimate whether the species list for a site is complete or not and use the Chao formulae to indicate the true number of species". Clearly, most of what I was saying above is inapplicable if species have been sampled incompletely, as is the case with the vast majority of biodiversity in invertebrates and microbes, for example. In those cases, asymptotic estimates based on sample saturation would have to be used; but as we all know, they do have their problems!

"What I want to know is how do I scale this up for a landscape scale biodiversity estimates? My first thoughts are that this is simply an additive activity (adding several surveys together) but this might become too labour intensive to be practical." I immediately think of species-area curves here. Rosenzweig's book is a great resource on this, and rightly very influential. It is species-area curves that predict how many species, after re-equilibration, will remain after reducing a given biome such as neotropical rainforest by 1/2 or 3/4. And species-area curves lead to the idea that there will be an extinction deficit: we have already doomed many species to go extinct, even though they're still around and looking healthy today.

But of course, this probably isn't what you meant; instead you were saying, I think, that how should we value additional habitat area? Would it be worth spending twice as much on conservation if we double the area conserved, because we have twice as much of the same stuff we had before? Or should we go with the species area curve, and scale our value system to the numbers of extra species (which will typically decline per each added increment of area) conserved per dollar? Well it is a hard one, that, but I think most would go with the numbers of species, rather than the overall area.

Species number bashing and scaling in a sea of ignorance

Allan Watt, Centre for Ecology and Hydrology, Banchory, UK

This has been an extremely interesting e-conference and I have enjoyed the recent discussion between Alan Feest, James Mallet and Peter Duelli, agreeing with most of what they have written.

I would like to endorse the comments made by James Mallet about Shannon-Wiener and other indices. I don't see their value either and agree with him and Peter Duelli that it is to be expected that many studies use species richness (or lists of species) rather than apparently sophisticated measures of biodiversity. I have to confess to using methods such as the Chao formulae as I feel they are extremely useful in trying to predict the number of species present from the inevitably inadequate samples usually taken. But while on the subject, I have never forgotten the warning by Ian Woiwod that estimates of species richness are biased by the abundance of the species concerned and therefore inadequate too.

I would also, however, like to take issue with the points made by Alan Feest and James Mallet about scaling up from local samples to landscape level biodiversity estimates. I agree with James that this is not an additive process but I believe that species-area curves should be used with extreme caution (if at all). The theory of island biogeography was developed from research on true islands surrounded by sea. Terrestrial habitat islands are different – the surrounding "sea" often supports many of the species found in the habitat island.

Because scale is the main theme running through this e-conference, I would like to hear other views about scaling up from local samples.

Re: Species number bashing and scaling in a sea of ignorance

Alan Feest, Water and Environmental Management Research Centre, University of Bristol, UK

As will be clear from my contributions so far I am approaching the topic of scaling from a the viewpoint of a practitioner involved in measuring biodiversity quality rather than a theorist. I have so far worked on macrofungi, bryophytes, beetles, butterflies and spiders since all of these have potential as indicators of the biodiversity status and change of an area; either alone or jointly.

My problem on moving from area assessments of biodiversity are that the areas studied are chosen as relatively uniform habitats and as such show good approximations to an asymptote in species accumulation curves indicating that the qualities described for the measured biodiversity quality represents most of the species of the study group present. In the UK, habitats are normally (at least in lowland areas) of small scale and an intimate mixture. My experience has been that two neighbouring tracts of land with similar geology and climate but differing vegetation can have completely different biodiversity quality. Therefore on a landscape scale how does one scale up from the individual sample areas to the landscape scale and is this a worthwhile activity? Should we just review the different habitats as subsets or are they additive? How many of these possible subsets represents the landscape? When does one subset of habitats finish and another begin?

In answer to James Mallet's question about additionality of biodiversity; how big does the area have to be for a useful species area curve to result? An example of this problem was a beetle biodiversity quality assessment of a golf course. Eight different habitats were identified on the golf course and a thorough survey failed to demonstrate any reduction in the slope of the species accumulation curve whereas an upland area in NW England gave identical results for three concurrent surveys in two years and a good indication of the asymptote of the species accumulation curve. The latter of these cases covered a far more extensive area than the former but had a clearly measurable biodiversity quality. Therefore I am compelled to conclude that a GIS definition of the landscape scale and "grain" is necessary before a sampling protocol can be proposed. But would this be able to detect the "grain size" of a golf course?

Scaling problems in biodiversity assessment: Summary Week 2

Ariel Bergamini and **Christoph Scheidegger**, Swiss Federal Research Institute WSL and **Lisandro Benedetti-Cecchi**, Dipartimento di Scienze dell'Uomo e dell'Ambiente, University of Pisa, Session I Chairs

The main topic of the second week of this session of the e-conference was on patterns of biodiversity and scaling problems. We had six key-note contributions (five from terrestrial ecologists, one from a marine ecologist), but rather few other contributions on the topics.

Discussions on the use of morphospecies in biodiversity assessment, which was the topic of last Friday, dropped in the second week (or was posted on the net after the first week summary was written). It became clear that the use of morphospecies may have some shortcomings, but nevertheless may be very useful especially in cases with short money and if the aim is not to compare different regions but temporal changes at a given station.

Below, I summarize some key research questions, conclusions or problems that were identified and discussed during this second week:

- The selection of conservation-relevant areas (in terms of biodiversity) is scale (or grain) dependent. However, smaller grain size must not be better. For example, fine-scale data on insect diversity is rarely available. Thus, small grain size may prevent the inclusion of very species-rich groups. However, there are no general rules to apply to grain size or extent (Carsten Rahbek).
-There is a trade-off between geographical precision and taxonomical precision (Carsten Rahbek, see also contribution of Peter Duelli on mophospecies last Friday).
-Rarefaction was discussed as the method for quantification of biodiversity patterns at the landscape scale since most biodiversity indices are strongly sample-size dependent. Furthermore, a profound distinction was identified between species density and species richness. These two metrics may yield completely different answers to the same question (Nicholas Gotelli).
-Conclusions or biodiversity pattern detected, critically depend on the design of the study, i.e. how sampling units are spaced (i.e. coverage and distance apart) and placed (i.e. simple random sampling v. stratified random sampling, Sean Connell).
- Research on the scaling of biodiversity from sampling plots and ecosystems to landscapes or regions should go beyond the quantification of diversity components at different scales and provide ecologically relevant interpretation. Moreover, we don't even need to start setting up a monitoring scheme if we don't know how to relate observed changes to ecological processes and their drivers (Helen Wagner).
-A lack of long-term monitoring data exists, even in Europe. It is thus not easy to differentiate between population fluctuations and real trends (David Vackar).
-The usefulness of methods or indicators depends on the considered time scales. While grid data may be useful to monitor species richness over centuries, abundance based measures should be promoted when considering shorter time-scales. Furthermore, it is a challenge to find optimal levels and scales for biodiversity indicators to assess changes (David Vackar).
-Additive partitioning of gamma diversity may help to identify sets of habitat areas that comprise the largest beta diversity. These areas deserve special attention in formulating land-use practices or in prioritizing areas for protection. Additive partitions of diversity may also inform us about sampling designs or monitoring strategies by identifying the sampling scales that contribute most to beta diversity. Temporal partition of diversity is also possible and may be important in monitoring biodiversity (Thomas Crist).
-Richness patterns of rare and common species are not congruent and richness patterns of common species are more similar to overall richness patterns than patterns of rare species are. The contribution of rare and common species to overall diversity pattern may differ at different spatial scales. Thus, if drivers of species richness are identified, factors uniquely important for rare species will not be identified (Jack Lennon).

Furthermore, the recently developed 'biodiversity intactness index (BII)' by Scholes & Biggs (2005, Nature 434, 45-49) was critically discussed and some weaknesses were identified (all species are equal, slow response to change, large influence of species-rich groups etc., Alan Feest)

In the last week of the e-conference, we will have at least two keynotes (one by Robert Colwell and one by Niklaus Zimmerman & Antoine Guisan) focusing on estimation and on predictive modelling of species richness, respectively. I'm looking forward to this last week of the e-conference and I hope we will have some interesting discussions!

Spatial scaling and modelling of biodiversity patterns

Niklaus Zimmermann, Swiss Federal Research Institute WSL, Birmensdorf, Switzerland and **Antoine Guisan**, and University of Lausanne, Switzerland

SUMMARY: Understanding and predicting patterns of biodiversity across a range of spatial scales profits from a conceptual reflection of the driving forces and scale dependent features. Hierarchy theory may help to understand such pattern and suggest scaling techniques. Two examples are discussed on how to scale from smaller to larger spatial scales and from single species to multiple species predictions.

Biodiversity arises at many different scales, along axes of evolutionary development, hierarchical complexity, structural diversity, or spatial and temporal gradients. A multitude of theories has arisen attempting to explain such patterns at one scale or another. Here, we discuss a few points regarding the spatial scaling of diversity patterns relevant to the conservation and management of species richness. The points are linked to the question: Which of the two goals behind spatial scaling of biodiversity pattern is more important: accuracy or understanding (e.g. Mac Nally 2000)?

If accuracy is most important, then any spatial predictor can be used that helps to bridge across spatial scales. Often, derivatives of fine grained digital elevation models are used. However, models (or explanations) built on predictors that reflect relevant mechanisms are expected to provide more robust predictions under new environmental conditions (Guisan & Zimmermann 2000). But what theory addresses what scale? E.g. species-area relationships (SAR), intermediate disturbance, energy-diversity or heterogeneity-diversity concepts all are potentially useful explanations. Using an explanatory approach may suggest testing a series of competing models based on conceptual grounds instead of fitting just one global model. Successful models may reflect the "emergence" of pattern at a target scale [see M. Sharman's contribution and C. Rahbek's comment and paper (2005) on scale-[in]variance].

Hierarchy theory (e.g. Allen & Hoekstra 1992) allows addressing how lower scale mechanisms interact with the scale of interest, and an array of scaling methods has been applied (King 1991). For model-based scaling of biodiversity, it is essential to scale the predictors appropriately in order to maintain the predictive power at the next higher level. Thus the successful emergence (or maintenance) of the predictive power of one conceptual approach depends also on the scaling of the predictor, not only on the concept.

Let's assume we attempt to explain/predict the biodiversity of large spatial units (say 100km2). Is the SAR concept or the habitat diversity concept useful here? Conceptually yes, but practically it is rarely done. First, alpha-diversity plateaus at smaller areas, and second, habitat diversity pattern are not always available. However, large scale climate maps (e.g. temperature) allow testing the energy-diversity theory. Often, this is done by linking the "average climate per spatial unit" with the diversity. This a simple form of predictor scaling from a small spatial grain (say 25m-1km) to large sampling units. Are there more appropriate ways to proceed? E.g., can we incorporate the finer scale variability and species-environment relationships into the coarser scale explanation?

Instead of simply averaging temperature per larger sampling unit, we can use an array of additional statistical properties (variance, min, max, fractal dimension, etc.) that summarizes environmental heterogeneity at a higher scale. Linking this idea with SAR further suggests including additionally properties related to the area covered by each environmental domain (ED) per larger sampling unit. An ED combines areas of similar environmental conditions, and thus represents a "potential habitat". By doing this, we include some of the mechanisms of the lower scale into the explanation at the larger scale. This approach is currently investigated within the IntraBioDiv project (http://intrabiodiv.vitamib.com).

Such a "hierarchy view" can further be extended to different ways of predicting and understanding diversity patterns. Are the individual components of diversity (species, etc.) invariably dependent on the same environmental predictors? In other words, should each species be predicted individually, so that diversity patterns emerge "a posteriori" (Guisan &

Theurillat 2000, Ferrier 2002)? Alternatives include calibrating diversity models of (functional) species groups independently (Pausas and Austin 2001).

Integration of different biodiversity data sources into a landscape level approach

José M. García del Barrio, CIFOR-INIA. Madrid. Spain

SUMMARY: The knowledge of biodiversity at three main axes (biological groups, spatial and temporal scale) could be improved on the basis of a minimum territorial scale that integrates the different biodiversity data sources.

During the fist two weeks of the BioPlatform e-Conference "Landscape scale biodiversity assessment: the problem of scaling" there have been a large number of contributions focused on the central issue of scaling. A detailed knowledge of biodiversity at the three main axes (biological groups, spatial and temporal scale) seems to be our present scope, but the progress during the last years would allow us to reach the 2010-year with a definition of minimum requirements to assess biodiversity at the European level. These minimum requirements could be described as follow:

a) Biological groups: The main effort should be put into those groups with a more deep level of knowledge (plants, vertebrates, butterflies, etc.), given that we do not have enough data on the spatial distribution covering broad areas. We should implement RBA procedures, as the one described by P. Duelli, for the other type of organisms.

b) Territorial scale: a basic territorial unit should be defined in which most of the information on the different biological groups is available. In the previous Bioplatform conferences, we have argued on the suitability of the municipality as this Basic unit, and also because municipality could be considered as a basic landscape unit. Following the scheme by H. Nagendra, for each unit, we should establish a Land Use Cover by remote sensing (eg. CORINE land Cover) and a Plant Habitat Cover (eg. EUNIS classification) to determine the species richness of the reference biological groups. Based on these covers, it should be necessary to develop cost-efficient sampling strategies to estimate biodiversity parameters (eg. species richness, etc.) for the main biological groups. Those municipalities included in the Nature 2000 Network, ZEPAs or some other conservation units, should be focused to complete the biodiversity estimations for all the biological groups being considered.

c) Temporal scale: It would depend on the timing on the different projects from which data are obtained (national inventories, CORINE land cover, vertebrate species atlases, etc.), but 10 years seems to be an appropriate actualization period. However, this period should be reduced when important land uses changes are observed in some of the municipalities. A monitoring network with different landscape type of municipalities should be established to detect those changes.

To implement this procedure it is necessary to reach a broad agreement on different items: (sampling procedures, uses of GIS technologies, statistical software to define and aggregate biodiversity data at the landscape level, monitoring procedures, etc) and last but not the least, political agreement. Would it be possible to agree on an agenda for reaching a territorial knowledge of base-lines of biodiversity from European municipalities to counties, provinces and regions, in the 2010 perspective?

Estimation of species richness at large spatial scales: Potential, procedures, pitfalls

Robert Colwell, Department of Ecology and Evolutionary Biology, University of Connecticut, USA

SUMMARY: I argue for sampling schemes stratified by discontinuities in the biotic mosaic. Richness in large spatial domains can then be approximated by estimating alpha richness within 'patch types' (including undetected species) and the number of species shared between them (a function of beta richness). The union of detected species lists for all patch types represents a lower-bound estimate of total domain richness, whereas the sum of all patch-type richness estimates (including undetected species) is an approximate upper bound estimate for total domain richness.

As several contributors to this e-conference have rightly pointed out, biodiversity as both a scientific and a practical concept encompasses far more than simple species richness, yet richness nonetheless remains a key component of biodiversity for biologists as well as for decision-makers. The e-conference organizers have asked me to discuss the estimation of species richness at large spatial scales, which I take to mean (for macro-organisms) scales that span landscapes or eco-regions, although the same scheme should apply, in a hierarchical manner, to continents, ocean basins or even the planet. To keep the concepts general, I will refer to such areas simply as 'domains', leaving scale unspecified.

With the exception of very well known groups in very well known places (for which we already have good estimates of total richness anyway), richness must generally be estimated based on samples. I first consider sampling schemes and then discuss estimation. Throughout, I assume that we are talking about some limited taxonomic or functionally defined group of organisms subject to detection by well-defined sampling methods. The concepts and suggestions in this contribution have been shaped by my participation in an ongoing regional inventory of arthropods in Costa Rica (Longino et al. 2002). Please note that this rather personal essay makes no pretense of offering a balanced review of the literature that bears on the many topics it touches, nor does it claim any original insights. My objective is simply to bring together concepts, provide some guiding principles, and offer some tools.

The testimony of our own eyes and the distribution of organisms confirm that the biosphere is not organized as a set of smooth continua in space, but rather as a complex 'biotic mosaic' of variably discontinuous assemblages, with the discontinuities driven in the shorter term by topography, hydrology, recent disturbance history, dispersal limitation, species interactions, and human land use patterns and in the longer term by climate and earth history. Over larger spatial or climatic scales the 'patches' of the mosaic can be better viewed as ordered along gradients (in either physical or multivariate space). Unfortunately, the geometry of the biotic mosaic is remarkably idiosyncratic (although it may be properly fractal for some organisms at some scales), which means that designing a scheme for estimating richness at large spatial scales is more like designing trousers for an elephant than finding yourself a hat that fits.

One effective way to deal with idiosyncratic biotic patterns is to take advantage of biotic discontinuities to define 'patch types' in the mosaic for sampling purposes; an alternative is to select sampling sites along explicit gradients. Both strategies represent forms of stratified sampling, in which the strata are the patch types or gradient sites, and multiple samples within them are treated as approximate replicates (perhaps nested: samples within a patch, patches within patch types; preferably adjusted for spatial autocorrelation).

Needless to say, any particular definition of patch types (e.g. habitats, land-use types, eco-regions, elevations, or landscape units of some other description) and the scale that underlies them is ultimately somewhat arbitrary. A seemingly less arbitrary alternative would be spatially random sampling over the entire domain, with a multivariate approach to assess the relation of richness and species composition to underlying environmental and historical factors. But given limited resources (is they ever otherwise?), random sampling over heterogeneous domains is highly inefficient because of the uneven relative abundance of patch types: the biota of common patch types are oversampled compared to the biota of rarer patch types, which may even be missed entirely. If one accepts a within-and-between-patch-type design framework, the definition of patch types (or sample spacing on gradients) might best be made at the design phase based on expert advice and whatever prior data exist, with the possibility of later iterative adjustment (resources permitting!) based on what the biota has to say. (Sean Connell and Carsten Rahbek discussed this problem earlier in the e-conference.)

Estimation of both within-patch richness (alpha diversity) and between patch turnover (beta diversity) from sampling data is bedeviled by 'Preston's demon': the larger the samples (or the greater the number of samples), the more species one detects, as Preston's 'veil line' (Preston 1948) penetrates further and further into the rare end of the relative abundance distribution.

In the case of within-patch richness, the number of species detected by sampling is a non-linear, saturating function of the sampling effort, where sampling effort is best measured by the accumulated number of individuals sampled (for groups in which 'individuals' make sense) (Gotelli and Colwell 2001). Because samples from different patches or patch types vary in size, rarefaction is one way to allow valid richness comparisons (see Gotelli's contribution to this e-conference). Because samples within patches or patch types are usually heterogeneous in composition and in number of individuals, despite being treated as replicates for the purpose of richness estimation, comparison of richness among patch types is usually best done on the basis of sample-based rarefaction scaled to individuals (Colwell et al. 2004, implemented in Colwell 2005).

Although comparisons of 'alpha richness' among patch types by rarefaction are interesting in their own right, they fail to take us towards the goal of estimating 'gamma richness'. For this objective we need estimates of total richness for each patch type, including undetected species. Nonparametric richness estimators attempt this feat by considering the pattern of rare species in the samples, yielding minimum estimates of total richness (Colwell and Coddington 1994; Chao 2004; Magurran 2004; methods implemented in Chao and Shen 2005 and Colwell 2005); mixture models can reliably extend the reach of sample sets twofold to threefold by extrapolation (Colwell et al. 2004, Mao et al. in press).

If we had full knowledge of the biota of all patch types within a domain, it would be simple to determine the total biota for two, three…all types combined, computing some measure of (average or pair-specific) beta richness (species turnover) along the way. For sampling data, the problem is much more difficult. Undetected species within patch types are not only undetected, they are unidentified, so that that we do not know whether the same or different species remain undetected in different patch types. Nonetheless, we can define lower and upper bounds for domain richness. The union of detected species lists for all patch types, pooled, provides a lower bound estimate of total domain richness, on the assumption that every species undetected in one patch type is detected in some other patch type. The sum of total richness estimates over all patch types (including undetected species from each patch type) is an approximate upper bound estimate of total domain richness, assuming that undetected species are entirely different for each patch type. The truth doubtless lies between these bounds.

To estimate the true domain richness, we need information about the true pattern of shared species among patch types. Statistical tools for estimating the true number of species shared by two sample sets, including species undetected in one or both sets, are scarce, and this is an area in which more work is needed. Chao et al. (2000) developed coverage-based estimators for shared species, based on patterns of rare, shared and rare unique species in the observed species lists (implemented in Chao and Shen 2005 and Colwell 2005), but these pair-wise tools cannot directly assess the number shared species among N sample sets.

Many studies have attempted to address the problem of estimating beta diversity, or pooling samples (patch types or random samples) by using similarity indexes, such as the Sørensen or Jaccard indexes. Unfortunately, the number of observed, shared species is almost always an underestimate of the true number of shared species, because of the undersampling of rare species. This means that species lists based on samples generally appear proportionally more distinct than they ought to, similarity indexes are routinely biased downward, and slope estimates for the ‘distance decay of similarity’ (e.g., Nekola and White 1999, Pyke et al. 2001) are overestimated. Recently, Chao et al. (2005, in press) developed an estimation-based family of similarity indexes that greatly reduce under-sampling bias and promise to help correct this longstanding dilemma (implemented in Chao and Shen 2005 and Colwell 2005). These indices are based on the probability that two randomly chosen individuals, one from each of two samples, both belong to species shared by both samples (but not necessarily to the same shared species). The estimators for these indexes take into account the contribution to the true value of this probability made by species actually present at both sites, but not detected in one or both samples.

Nailing down the questions

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

This conference has been running for some time now, and to me at least it has not always been clear what the key research issues are. Can you suggest one or more one-sentence questions that identify the vital, strategically important bits of information that must be known about scaling if we are to slow or halt the loss of biodiversity?

Re: Nailing down the questions

James Mallet, UCL, London

Many people have argued that we need good taxonomic databases for biodiversity assessment. At the European level, however, funding in this area is a very low research priority at the moment, and those funds which do exist seems to focus on parochial issues of European biodiversity, particularly with relation to agriculture and climate change in Europe.

We have the world's richest resources in terms of world collections of specimens and expertise in our museums. My vision for all of this data and expertise is that we should make these collections, and systematics information such as classifications and checklists, available as far as possible on the internet. To some extent, the model would allow data repatriation of information generated from these collections from countries with much higher diversity than our own in the tropics.

We have begun a trial project entitled Global Butterfly Names, funded by GBIF-ECAT program, in the Natural History Museum, London, to document the entire worldwide published scientific nomenclature of butterflies (around 17,500 species, 60,000 names), and to classify these names according to validity, and current classification as to genus, species, subspecies etc. The main systematist involved is Dr Gerardo Lamas, who has masterminded a checklist of the entire Neotropical butterfly fauna, a substantial fraction of the world's species. The data will be updating the already established LepIndex project (http://www.nhm.ac.uk/entomology/lepindex/), a nomenclature database which holds an electronic copy of the original NHM card index for the entire Lepidoptera of 290,099 names.

Although data delivery in LepIndex is still rudimentary (most of the names are not currently classified), the ability to deliver checklists, and the image delivery for the card index can easily be adapted and improved to provide online delivery for all information in the form of images of type specimens associated with the names and other information such as distribution, biology and conservation status, and linked to specimen databases with individual data on these topics. Rapid lists at any arbitrary level of classification could be produced for any biodiversity and conservation status purpose, which would considerably aid taxonomic research as well as projects requiring rapid biodiversity assessment. We envisage this system, and other systems built to take advantage of such data, such as Species 2000, to provide the kind of complete taxonomic information that GENBANK/EMBL provides for DNA and genomics data.

Unfortunately, the funding for this work is extremely difficult to find at UK level and at European level. We are running on a rudimentary budget level; and can employ Dr. Lamas for only 4 months on our project. The problem is not so much the database development (although it is hard to find money even for that), but the taxonomic data and image data collection itself -- in other words the basic taxonomy underlying the project. If anyone knows of any way of moving this project forward more speedily, we would be very interested to hear from them.

A brief addendum: Perhaps I should explain why I feel basic taxonomy has great importance to biodiversity assessment and scaling problems. As already mentioned, the problem is that species-level taxonomy is heavily used by biodiversity scientists. However, as we have seen, different definitions of species may lead to more- or less-inclusive entities.

The problems could be obviated if users of taxonomic decisions in biodiversity assessment could tailor the level they consider species to their own personal philosophies. Hence the need for fully synonymic checklists. Very often (but not always, of course) valid subspecies of polytypic species could be used as species names by those using more finely divided phylogenetic species concepts.

Furthermore, anyone basing decisions on particular taxa or groups of taxa may find, on looking up the taxa in a field guide, that the name is not there, or that they belong to a different genus. A fully synonymic list, containing all names, both currently considered valid and those considered invalid, would enable one to look up any validly published species or subspecies and find out something about the group of taxa surrounding or included within the taxon of interest.

Another argument that I have not mentioned, but should have done earlier, is that DNA data (for example in DNA-based identification strategies such as DNA barcoding) should be linked to specimens that would ultimately be placeable in taxonomic hierarchies like those proposed here. In addition, studies of genetic aspects of biodiversity should likewise be linked to specimens and ultimately scientific names.

Ultimately, nomenclatural and taxonomic databases should provide the fundamental taxonomic basis for all biological information, including that involved in biodiversity studies and implementation of conservation. Only in this way will the taxonomic impediment be removed.

Scaling problems in biodiversity assessment: Summary Week 3

Ariel Bergamini and **Christoph Scheidegger**, Swiss Federal Research Institute WSL and **Lisandro Benedetti-Cecchi**, Dipartimento di Scienze dell'Uomo e dell'Ambiente, University of Pisa, Session I Chairs

In this last week of the e-conference we had only two keynotes. Nevertheless we also had some very interesting other contributions.

The most important topics were the following:

- The use of species lists as indicators of biodiversity was questioned. One of the main problems identified with species lists is that on such lists all species are regarded as equal which, of course, they are not. However, species lists were also seen as very useful because they are politically understandable
- Estimating of species richness within large domains (i.e. areas) based on species occurrences within plots was discussed by Robert Colwell. One of the main problems with that approach seems to be how to get an estimate for an entire landscape because undetected species are also unidentified. That means it is not so easy to get an overall estimate because it is not known if two undetected species in two habitats are the same or not.
- Niklaus Zimmermann and Antoine Guisan emphasized in their keynote that predictors which reflect relevant mechanisms are expected to provide more robust predictions across spatial scales under new environmental conditions and should therefore be preferred over other predictors
- There is a lack of basic taxonomic agreement in various groups. It is of upmost importance to find a consensus and to compile full synonymic checklists.

Furthermore, 'A user's guide to biodiversity indicators' published by the European Academies Science Advisory Council (EASAC) this week can be downloaded at http://www.easac.org.

Biological scales and conservation

Introduction to Session II: Biological scales and conservation- Terrestrial perspective

András Báldi, Animal Ecology Research Group, HAS, Hungarian Natural History Museum, Session II Chair.

This session on "Biological scales and conservation" intends to explore effects of scale on various issues related to conservation. "Acts in the ecological theatre are played out at various scales of space and time" (Wiens, J.A. 1989. Functional Ecology 3:385-397). In other words, research results from different scales may show different patterns; a pattern detected as relatively homogenous on a scale with coarse resolution might disappear when a finer resolution is applied or vice versa. For example, increase in biodiversity with the available energy is linear on large spatial scale, while hump-shaped on the local scale (Chase. & Leibold 2002. Nature 416:427-430). Scale is a fundamental concept in ecology, yet most investigations are restricted both in time and space, rendering generalisation more difficult (Báldi & McCollin 2003. Global Ecology and Biogeography 12:1-3.). If there is little research on scale effect, if the available knowledge is scarce and fragmented, how can we guide nature conservation practice to maintain biodiversity?

The main aim of the session is to give an essence of the generality of scale effect, and relate it to conservation. Picking up key issues, and putting them together may help to understand better how scale effect is present in ecology and conservation. Therefore, for the three-week conference three broad topics were chosen. First, the idea is to show that although scale effect is present in most patterns, its inclusion into practice is largely missing. Prof. Paul Opdam argues that it is necessary to link ecological scales from metapopulation to biogeography, then to link ecology with planning - "a huge challenge". In deed, even if adding the non-linearity of ecological systems, presented by Dr. Jorgi Bascompte, or the dependence of general conservation guidelines on temporal and spatial scales, as showed by Prof. Bill Kunin. Dr. Michel Baguette argues to address the landscape scale for biodiversity conservation. Several other interesting contributions are expected, and I think that these will give the essence of diversity of scale issue in ecology and conservation.

The two other topics address experts of various ecosystems (forest, farmland, freshwater, etc.), and various taxa (plants, birds, amphibians, insects, etc.). Marine issues are equally important and are missing simply due to practical reasons. The experts on terrestrial systems are usually specialised to a given ecosystem or taxa. Due to the huge variety of these ecosystems and taxa, most experts have different views on spatial or temporal scales. It is obvious if considering tree line shift due to climate change, frogs in temporal ponds, or mites in the soil; the important scales are decreasing by several order of magnitudes from centuries and continents to weeks and centimetres.

Although it may be a naive expectation from a single e-conference, it would be nice to explore some general patterns. Then, it would be easier to give guidelines for policy-makers on how to treat the scale issue in regulations and administration, or at least to show the major gaps in our knowledge. The diversity of views and approaches of the contributions, and the high level expertise involved, may help to achieve this goal.

Introduction to Session II: Biological scales and conservation- Marine perspective

Keith Hiscock, Marine Biological Association of the UK, Plymouth, UK, Session II Chair

Conventions, directives and national statutes usually implement conservation through the protection of sites for particular habitats, and species. There are features of marine ecosystems that question whether such a 'traditional' approach will provide the protection needed in the sea. This part of the e-conference will help to catalogue the advantages and disadvantages of different biological scales for biodiversity conservation in the marine environment.

Marine ecology considerations: Marine systems are often considered much more 'open' than terrestrial systems. Contaminants and nutrients can be spread over large areas by water currents. Most marine species have propagules that are readily dispersed in the water column and connectivity is therefore considered to be high. Fish and other swimming species have few barriers to prevent their migration. Habitats, even in restricted areas, may rely for their integrity and health on large scale processes. For instance, offshore aggregate extraction may have consequential effects for inshore sediment recruitment and therefore habitat maintenance. Such considerations question the value of small-scale localized action to protect biotopes or species.

Practical issues: Measures that protect a species may be particularly valuable when there is targeted collection of that species (for instance, edible limpets) or non-targeted by-catch (for instance, of porpoises). Measures that protect habitats may be particularly valuable when a habitat or biotope or biotope complex is being damaged by targeted human activities and is unlikely to recover (for instance, horse mussel beds, maerl beds). But where do 'marine landscapes' or 'ecoregions' fit into management to protect biodiversity?

What is the 'best' scale (or is there a best scale?) to use in developing the 'Network of marine protected areas' to be established by 2012 as an outcome of the World Conference on Sustainable Development in 2002? And, is there anything about 'scale' that produces or informs what constitutes a 'network'.

New initiatives abound – not least spawned by discussions towards an EU Marine Strategy. We are being implored to use the 'ecosystem approach' in managing human activities in the marine environment – an approach that entails taking into consideration all elements that make up the ecosystem as well as activities taking place there in order to ensure that the biodiversity, health and integrity of the marine environment is maintained. What is the most practical scale to implement this 'ecosystem approach'?

Separate topics will be introduced as follows:

1. Species – Jean-Luc Solandt, Marine Conservation Society, UK
2. Biotopes and biotope complexes (habitats). OSPAR approaches and criteria. Hein Rune Skjoldal
3. Marine Landscapes. Paul Robinson (Joint Nature Conservation Committee, UK).
4. Ecoregions. Stephan Lutter, WWF.

Their brief is to promote the practical advantages and the advantages to conservation of biological resources of their 'level' and 'type' and to ask questions of e-participants so that any points they raise or that are raised by correspondents can contribute to the debate.

Biological scales and conservation

Paul Opdam, Wageningen University Department of Land Use Planning, Alterra Landscape Centre, The Netherlands

SUMMARY: From the point of view of biodiversity conservation, biological scales are expressed in terms of space and time. We need to understand how biological processes across a range of spatial scales are expressed in biodiversity patterns. Ecosystem networks are a key concept in linking local, regional and biogeographically scales in a dynamic world under the pressure of climate change.

Ecosystem networks. To ensure persistence, populations need enough habitat area. In fragmented landscapes under heavy human pressure, effective conservation depends on whether small ecosystems can be functionally linked into ecosystem networks supporting metapopulations. Because larger species need more space, but can cover larger distances, ecosystem networks for large species extend over larger spatial scales than networks for small species. Although metapopulation ecology recently developed rapidly, most knowledge is at a theoretical level. We lack methods to determine ecosystem networks and ecologically relevant indicators for sustainable ecosystem networks that can be used in practice (including practical minimal thresholds for viability). We also need diagnosis methods to determine sustainable ecosystem networks for biodiversity, which integrate species differences.

Linking ecological scales is essential if the cause of a conservation problem is at a different level of scale than its solution. This is especially true in the light of a change in climate, which is a process at the scale of a continent, but requiring adaptation of fragmented landscapes to climate change effects at the local and regional scale level. It is necessary that biogeographically approaches of range patterns and range dynamics are linked to metapopulation dynamics in ecosystem networks.

Time scale: responses to changing land use. Predictions on the effectiveness of conservation measures for populations and communities should include a time dimension. The interaction between spatial patterns, at a range of scales, and the dynamics in the distribution of species is very complex and hardly explored. For example, the effects of fragmentation are treated as static, but in reality the spatial effect of fragmentation on the distribution of a species may be apparent only at low population levels. Spatial responses of metapopulations to local and regional disturbances must be understood and quantified, at a range of spatial scales, in relation to the scale level, the frequency and the intensity of the disturbance.

Linking ecology and planning. Because conservation of biodiversity is implemented in a complex and multipurpose world, ecologically significant spatial scales have to be related to administrative scales. EU, nations, regions, and local municipalities encompass a range of administrative scales, responsible for taking action when it comes to conservation of species. For most species, ecosystem networks encompass scale levels well above the municipality size, and some may even require networks on national and international levels. Because most decision making in land development is by local actors at the local level, there lies a huge challenge in developing spatial planning concepts and methods to link the different scale levels at which decisions are made before developing land, and to make sure that local actors implement in their planning process the need for improving large scale ecosystem patterns.

Scale and nature reserve design

William E. Kunin, Faculty of Biological Sciences, University of Leeds, UK

SUMMARY: At the core of many specific issues in nature reserve design is a scaling difference: with intrinsic tradeoffs between the maximisation of a diversity and of b diversity, and between short-term collecting and long-term maintenance of biodiversity.

In 1975, Jared Diamond proposed a series of general principles for nature reserve design. In subsequent decades, almost all of Diamond's principles have been called into question (e.g. Simberloff 1988), but his list nonetheless provides a useful summary of some key issues to be addressed. I would argue that in almost every case, the answers to these questions depend on temporal and spatial scale.

These scaling issues are best illustrated by the most famous of Diamond's principles: that a single large reserve is preferable to a set of several smaller reserves with the same total area. Many have wearied of the SLOSS ("Single Large Or Several Small") debate, but the confusion may be resolved by when scale is considered. Diamond argued that if the number of species initially contained in a reserve (or network of reserves) is determined by its area, the Single Large and Several Small reserve strategies ought to begin with the same number of species. After isolation, however, smaller reserves should lose more of their species, and more quickly, than large ones, so the Single Large strategy appears preferable (Fig 1a).

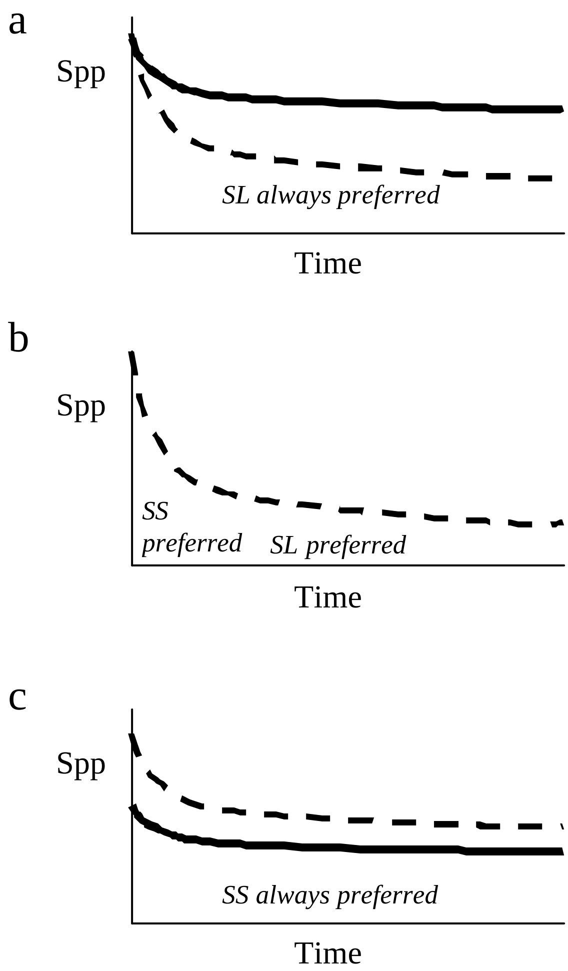
But the two strategies are not likely to begin with the same number of species. The total number of species in a set of several reserves depends not just on their total size, but on how far apart they are in space and in environment. A Single Large reserve is of necessity all in one place, but Several Small reserves can be some distance apart. In general, the further apart they are in space or environment, the more dissimilar their biotic communities become, and so the greater the total number of species they collectively contain. If the Several Small strategy begins with more species than the Single Large reserve, but if they lose those species more quickly (Fig 1b), then the preferred strategy depends on the time scale considered: Several Small wins in the short term, Single large in the long term. Given the very long relaxation times noted in e.g. land bridge islands (e.g. Heaney 1986), the "long-term" may be very long indeed.

But extinction is a non-linear function of area. As the size of an island grows, it is expected to lose progressively smaller fractions of its biota, and more slowly. Eventually, we can attain a size where even the "Small" reserves are big enough to be relatively secure, so that the extinction cost of subdividing habitats is less than the benefit of sampling greater diversity in the first place (Fig 1c). Beyond this scale, the Several Small strategy preserves more species than a Single Large reserve, even in the long run. Thus when large amounts of land are being discussed, subdividing it may be preferred, whereas when less land is available, it might best be kept as a single reserve (at least in the long run). Identifying these critical cut-off scales is a key question for future research.

These same scale issues haunt most of Diamond's other principles. His idea that multiple small reserves should be sited as close together as possible, and should be clustered in a circle rather than arranged in a line are both trade-offs between greater b-diversity sampling where sites are far apart or lined up, versus lower species loss when they're close together or clustered. As in SLOSS, dispersion should be preferred at short time scales and at large spatial scales. The recommendation that reserves be as nearly circular as possible presents a similar trade-off between scale-specific edge effects favouring circular reserves at small scales, but increases in b-diversity sampling come to outweigh this at larger scales, giving elongated reserves the advantage (Kunin 1997).

It should not surprise us that the answers to these questions are scale-specific. After all, if a Single Large strategy were always preferable to Several Small reserves of the same total area, the optimal global conservation strategy would be to gather all of the World's conservation areas into one spot. To do so would ignore the obvious benefit of preserving different species in different parts of the globe. Conservationists need to learn to take scale seriously.

Figure 1. Hypothetical time series of species protected over time under a "Single Large" (SL) and a "Several Small" (SS) reserve strategy of the same total area. (a) As envisioned by Diamond 1975, both strategies begin with the same number of species, and the SL strategy is always favoured. (b) Where SS samples greater initial diversity, it may be favoured in the short term, but slip behind SL in the long term. (c) At large spatial scales, however, the increased extinction in SS may still not be enough to overcome the initial advantage of a larger species sample, so that SS is preferred even in the long term.



Marine biotopes and habitats

Hein Rune Skjoldal, Institute of Marine Research, Bergen, Norway

Terminology: In the UN Convention on Biological Diversity: www.biodiv.org/welcome.aspx, "habitat" means the place or type of site where an organism or population naturally occurs (Article 2). The OSPAR Convention (the Convention for the Protection of the Marine Environment of the North-East Atlantic, www.ospar.org) refers in its Annex V to this definition of habitat. From this definition, habitats can range in size from small to large, as can also ecosystems. It is convenient to consider marine ecosystems as larger units (Large Marine Ecosystems (LMEs) – www.lme.noaa.gov) with interacting populations tied together in food webs, consisting of a mosaic of smaller scale habitats constituting the underwater landscape or seascape. Habitats can be taken to be synonymous with biotopes.

Classification and mapping. Nature is continuous but with more or less distinct discontinuities in environmental conditions and distributions of organisms. This forms the basis for classifying and mapping of biotopes or habitats. EUNIS (the European Nature Information System) is a hierarchical classification system that at the first level separates aquatic from terrestrial systems and at levels 2 and 3, separates broad habitat categories based on physical environmental conditions (depth, substrate, salinity, exposure). From level 4 and onwards, biological information is taken into account in more and more detailed form. Thus kelp forests may be one habitat type, with sub-types dependent on dominant species and species composition. The hierarchical classification is related to scale, going from broad scales at the first levels to finer scales at higher levels of classification.

Habitats may be seen in the generic sense as specific habitat types (e.g. kelp forest or deep-water coral reef) and in the geographic sense as the location or distribution of a given habitat type.

Quantity and quality of habitats. The distinction between quantity and quality is dependent on the hierarchical level of the classification system of habitats. Dominant species and species composition, which are used in classification, may change due to natural or anthropogenic causes. Such changes may first be registered as changes in habitat quality in finer scale classification, but they may subsequently lead to a change in classified habitat type and thereby in the quantity of habitats.

Management and conservation. For environmental managers, habitats or biotopes may be suitable for use at an 'operational level'. They can be linked to measures of 'quality' (relevant to implementation of the Water Framework Directive in the EU), and monitoring can reveal whether extent and quality is increasing or declining. Particularly sensitive biotopes that are likely to be damaged by human activities can be identified (see http://www.marlin.ac.uk) and made candidates for protection. Habitat conservation has traditionally been used to effect species conservation – safeguard the habitat and the component species will thrive. This is also likely to be true for marine environments for which the full range of species may not yet have been appreciated and described. We are faced, however, with a serious lack of information on the distribution of habitats in most European seas. Detailed habitat mapping should therefore be given the highest priority to promote conservation and sustainable management practices.

Non-linearities, scale and conservation

Jordi Bascompte, Estación Biológica de Doñana, CSIC, Spain

SUMMARY: Non-linearities and thresholds in ecosystem processes dictate abrupt, discontinuous scales for conservation. While previous work has predicted appropriate scales for the conservation of single species, almost nothing is known about scales for the maintenance of whole networks of ecological interactions.

Defining the spatial scale of appropriate conservation units is an important issue in conservation biology. In the last few years, theoretical ecology has explored the spatial component of population and community dynamics (e.g. Bascompte and Solé 1995, Hanski and Gilpin 1997, Tilman and Kareiva 1997). This "last frontier" has shown to what extend non-linearities require minimum areas for the generation and maintenance of ecological processes. If ecological processes were linear, small spatial samples would contain miniatures of the whole ecosystem. However, we know that this is not the case.

At the level of two-interacting populations, simple models of host-parasitoids and species competition show how species coexistence is a qualitative result not predicted by non-spatial models. This coexistence is oftentimes associated with spatial self-organizing patterns (Hassell et al. 1991, Bascompte and Solé 1995). Examples are spiral waves in which high population densities are produced at focal points and spread towards the periphery in the form of travelling waves. These spiral waves require a minimum spatial size. Understanding these processes of self-organization, thus, dictates a minimum spatial scale for the persistence of populations.

Similar considerations can be made in relation to the problem of habitat fragmentation. Extinction thresholds are critical values of habitat destruction at which metapopulations go extinct even when some habitat is still available (Lande 1987). When considering localized dispersal in spatially explicit landscapes, such thresholds become more abrupt, and are related to sudden, discontinuous changes in the structure of the landscape (Bascompte and Solé 1996). These changes give place to a sudden breakage of previously continuous habitat into an archipelago of isolated patches. Near these critical destruction values, the spatial scale undergoes a transition.

The question of scales for conservation becomes more complicated when we consider whole ecological networks. Ecosystem stability depends on the structure of the network of interactions (May 1973, Pimm and Lawton 1977, McCann et al 1998), and this network is organized at a particular scale. This calls for a convergence of spatial dynamics and ecological networks. For example, food web studies focus on aggregated values over a given spatial area, and little is known about their spatial dimension. Partly, this is a consequence of the trade-off in the level of complexity of the models used so far. Ecologists have studied either the dynamics of one species at multiple patches, or the dynamics of many species at a single patch. How does network structure change as we increase spatial scale? In other words, what is the minimum scale for the conservation of ecological networks? This is a major gap that requires attention. One potentially useful strategy is to use integrative studies that combine the few data available on the spatial dimension of food webs with spatial models of multi-species interactions.

Species-scale conservation

Jean-Luc Solandt, Marine Conservation Society, Ross-on-Wye, UK

SUMMARY: Is species protection complementary to the other scales being considered in this e-conference? Or, is it a different category entirely and not a 'scale'?

Protecting species is a fundamental part of conservation and, in the UK, there are about 400 marine species listed as 'of conservation concern' by the UK Joint Nature Conservation Committee. However, identifying which of those species are under immediate threat becomes difficult if the IUCN 'Red list' criteria are used – marine species usually come-out 'data deficient'. This is also the problem with any scale of assessment of marine species, be it international, UK, national, or regional because gaining accurate assessment of marine species populations is very difficult.

There are decline criteria that 'work' for marine species. They are based on studies undertaken by OSPAR (The Oslo & Paris Commissions for the Protection of the Marine Environment of the North-East Atlantic), and list threatened and / or declining species and habitats (Accessible from: http://www.ospar.org/eng/html/welcome.html). At a country level, the UK published the Biodiversity Action Plan in 1994 (which added approximately 19 purely marine species and 19 purely marine habitats in 1999) in order to conserve threatened and declining species in the UK continental shelf area.

For environmental managers, species are suitable for use at an 'operational level'. The location where nationally rare or scarce species are found can be protected from potentially damaging activities and can be mapped so that they are protected as far as possible in the case of an accident (e.g. oil spill). Particularly sensitive species that are likely to be damaged by human activities can be identified (see http://www.marlin.ac.uk) and made candidates for protection.

Whilst rare or threatened species that are sessile or sedentary in nature or that have particular locations where they breed may be protected by identifying areas, the question of scale becomes a significant one for migratory species (for instance, the charismatic megafauna that constitutes a high proportion of species listed for protection).

Protecting marine species may require protection of ecosystem processes at a large scale – affecting, for instance, water quality. Species may become extinct if their food source or habitat are removed: 'pinning a badge' on a species saying "protected" is not enough. Their habitat needs to be protected and food sources maintained. However, focused conservation efforts on particular species can be useful, as some species are ecologically 'representative' either for the quality status or type of habitat in which they occur.

What about species that are rare in a country only because they are at the geographical limits of their range or because their habitat is very restricted there. On the scale of north-east Atlantic, they are not rare or restricted. Should they be subject to conservation at a local scale when they are widespread at a north-east Atlantic scale?

So, what is your interpretation of the importance of species-scale conservation?

Re: Species-scale conservation

Keith Rennolls, University of Greenwich, UK

Jean-Luc Solandt says: "identifying which of those species are under immediate threat becomes difficult if the IUCN 'Red list' criteria are used – marine species usually come-out 'data deficient'".

However, experience with the use of species area curves tells us that even with large samples there will be a significant number of unobserved species. In fact the Good/Turing estimator of the proportion of unseen species (of all species that exist), is R1/n, where R1 is the number of singleton species observed and n is the number of observed individuals (Engin, Stochastic Abundance Models ,Chapman and Hall, p32).

Presumably if data is cumulated then the number of singleton species will be very small, and n will be very large, so almost all species in existence will be known. In any case it is often likely to be the case that those species most likely to be lost, (and hence under threat) are the rarest, and these will almost always be data deficient.

Jean-Luc Solandt also wrote: “focused conservation efforts on particular species can be useful, as some species are ecologically ‘representative’ either for the quality status or type of habitat in which they occur.”

I am afraid that this approach would fail. If a species is taken as a indicator-species because it is representative of the quality or type of its habitat, then focused efforts to conserve that species will make it un-representative of its habitat, and apparent indications of habitat conservation, using the same indicator species would be completely misleading.

IUCN Red List criteria

Keith Hiscock, Marine Biological Association of the UK, Plymouth, UK, session II Chair

Keith Rennolls perhaps misunderstood the IUCN criteria that Jean-Luc Solandt referred to. It is criteria such as the example below for “Endangered” that we marine biologist do not have sufficient information (except perhaps for seals) to apply: “An observed, estimated, inferred or suspected population size reduction of 70% over the last 10 years or three generations, whichever is the longer, where the causes of the reduction are clearly reversible AND understood AND ceased, ...”.

See: http://www.redlist.org/info/categories_criteria2001.html#critical

But, help is at hand and the criteria developed by OSPAR (the Oslo & Paris Commission for the Protection of the Marine Environment of the North East Atlantic) work well in a practical way and give us that oh-so-valuable commodity - terms that politicians can relate to (“threatened”, “endangered” etc.). See http://www.ospar.org/eng/html/welcome.html

Marine species scale conservation

Jean-Luc Solandt, Marine Conservation Society, Ross-on-Wye, UK

Regarding the debate above- unfortunately it is the case that some politicians and people at the ‘sharp end’ of conservation (the stakeholders, general public and local managers) identify more readily with the species that can be ‘flagships’ of habitat and ecosystem protection at a number of geographic scales. This issue is currently being discussed within a UK priority review of marine species and habitat conservation.

Species conservation and monitoring in the marine environment is also recognised within an academic framework. The excellent conference held by Ian Boyd of the Sea Mammal Research Unit called ‘Management of Marine Ecosystems: Monitoring change in upper trophic levels’ at ZSL, London, April 2004, showed that incidents of monitoring key changes to environmental variables and biological parameters could be described by the population fluctuations of those species at the top of the food chain (seabirds, seals, cetaceans etc). This is particularly pertinent where food chains are ‘uncomplicated’ and have few levels (such as algae - krill - penguin - leopard seal).

Furthermore, many species in the marine environment form monospecific frameworks (or reefs), such as horse mussel beds, maerl beds, and Lophelia deep waters reefs, which have significant impact on local biodiversity. Therefore consideration of conservation of these habitat-forming species is important to overall benthic biodiversity - this has been recognised within the UK Biodiversity Action Plan, and at OSPAR level, and within the definition of ‘habitats’ under the Habitats Directive.

At what scale should conservation efforts and planning be undertaken?

Alessandro Gimona, Macaulay Land Use Research Institute, Aberdeen, UK

The main point made here is that the processes producing biodiversity operate at very large scales. These, in theory, should be the focus of international efforts for the protection of biodiversity.

The question of scale in this forum is implicitly spatial, but I would like to draw attention to the fact that (as we all know) there is both a spatial and a temporal dimension of the preservation of biodiversity, and that the two are linked. Certainly patterns change with the scale of observation. Conservation advice, for very good practical reasons, is often given with a short time horizon in mind (for instance, 100 years of viability are a very short time for a species) and to a spatial scale which might not be relevant to the global extinction probability. One cannot help wondering if all the attention to details of "local" patterns is unwittingly distracting both ecologists and "society" from seeing the 'big picture' i.e. that -by reducing the global area of natural habitats- each generation is accumulating a large global extinction debt and passing it on to future generations. On the one hand, due to political constraints, both the spatial and [even more] the temporal scale at which problems are tackled need to be highly pragmatic. On the other, it is useful also to keep in mind the more abstract (and therefore less constrained) aspects of the debate, to be aware of the effects of 'pragmatic solutions' on long-term trends.

To answer the question of the appropriate scales of conservation it is valuable to keep, at least at the back of our minds, what has been learnt in macro-ecology (see e.g. Rosenzweig, 1995) and most participant are familiar with. This, by providing an upper limit to the 'extent', might help putting in prospective questions such as the "scale dependence of patterns" and "scale and conservation policy". Species diversity is generated and maintained by processes that occur at a wide scale, and simply cannot be recreated at the scale of nature reserves, and often, not even at the scale of national landscapes. Unfortunately, in the long run, nature reserves are unlikely to work for a great deal of species. The reason is that area is a fundamental attribute of ecosystems. Reserves can certainly slow down the pace of extinction, and, to be sure, are very useful and important in the short term. So is the protection of hot spots areas. However, if one considers a time horizon of centuries, reserves, even if connected by corridors, are unlikely to stop the trickle of extinctions, unless they are part of a wider strategy of landscape extensification. Species diversity, results from the balance between speciation and extinction, and in general, larger areas have more speciation and less extinction than smaller ones. Biogeographic provinces [self contained areas whose species originated only by speciation within them], are the right 'scale' at which, ultimately, the dynamic processes responsible for biodiversity occur, and should be protected Nature reserves which, collectively, occupy just a fraction of the area of various biogeographic provinces and biomes, are unlikely to maintain the dynamics at work. (In particular it is hard to see how speciation can keep pace with extinction). In other words, as others have already hinted at, the diversity of these vast systems cannot be stuffed into the (comparatively) small area covered by the world's protected areas, probably not even if these were to encompass all the known hot spots. Because human activities, and in particular land use change , have shrunk the area available for such natural processes to occur, speciation is bound to plummet and extinction to climb (and very probably climate change will add to this). Eventually, the number of species will tend to match the new species-area relationship. This means that, even with a series of connected nature reserves, a large (mass?) extinction event is probable in the next few hundred years, because of the species-area relationships within biogeographic provinces. Ecologists can point out to decision makers the problem of "lack of viable area" at the global scale. I wonder if they (we) have done enough in this respect. If a solution can be found, this has to include provision of more habitable landscapes worldwide. Politically this might or might not be feasible, and can only happen with a huge societal commitment.

Integrated conservation and sustainable landscape planning at the right biogeographic scale might be a part of the solution.

Re: At what scale should conservation efforts and planning be undertaken?

Kajetan Perzanowski, Carpathian Wildlife Research Station, Polish Academy of Sciences, Poland

Alessandro Gimona has tackled a very important aspect of biodiversity conservation ..."Species diversity is generated and maintained by processes that occur at a wide scale and cannot be recreated at the scale of nature reserves and ...even... national landscapes" and ..."Because human activities and land use change ... speciation is bound to plummet and extinction to climb".

An implication of those statements is that most probably we will be unable to prevent (even within a relatively short time-scale) the loss of a number of species from areas considerably altered by man. The first to disappear will be the highly specialised species having a narrow ecological niche, but in turn it can be expected that such areas will become dominated by a smaller number of opportunistic species, with probably a high number of individuals.

Subsequently it means that even if do succeed in providing a temporal refuge for rare, vanishing species within protected (usually small) areas, those species will be additionally threatened by a highly competitive species swarming around and infiltrating our nature reserves. It would be unreasonable to think that especially within overcrowded continents like Europe there is still a potential to extend the size of protected areas. However those small islands are at the moment the only living space left for certain species and where natural processes can occur without our intervention and "improvement". Therefore, it is probably high time to say that the most important function of protected areas is not public education, recreation etc. but simply the protection of certain components of the biosphere that cannot survive nowhere else. Although the majority of strictly protected reserves cannot be probably enlarged, there is still a possibility of introducing on a larger scale the concept of zonation of protected areas, as in case of biosphere reserves. That could be a "soft" way to extend the size of area being under some sort of protection and to "cushion" the most valuable habitats from a direct collision with anthropogenically transformed environment.

Re: Re: At what scale should conservation efforts and planning be undertaken?

Keith Rennolls, University of Greenwich, UK

Alessandro Gimona wrote: 1. "SUMMARY: The main point made here is that the processes producing biodiversity operate at very large scales. These, in theory, should be the focus of international efforts for the protection of biodiversity." 2. "One cannot help wondering if all the attention to details of "local" patterns is unwittingly distracting both ecologists and "society" from seeing the 'big picture' i.e. that -by reducing the global area of natural habitats- each generation is accumulating a large global extinction debt and passing it on to future generations". 3. "Species diversity is maintained by processes that occur at a wide scale, and simply cannot be recreated at the scale of nature reserves, and often, not even at the scale of national landscapes. Unfortunately, in the long run, nature reserves are unlikely to work for a great deal of species. The reason is that area is a fundamental attribute of ecosystems." 4. "Species diversity, results from the balance between speciation and extinction, and in general, larger areas have more speciation and less extinction than smaller ones".

I accept the above points which I have extracted as what I understand as the main thread from Alessandro's argument. The conclusion, in 1. Summary, is based on the assumption that conservation sites should be left alone to speciate and to suffer increasing extinction and eventually to find their own stable location on the species-area curve.

The second law of thermodynamics is that the entropy of a closed system must necessarily increase with time, http://www.panspermia.org/seconlaw.htm. This also applies to natural populations. The Shannon-Weaver formula for species diversity (as a measure of disorder) is of the same form as the entropy measure as used in thermodynamics. It is tempting to assume that nature, left to its own devices will maximize entropy pretty effectively, even "optimally". However, in terms of the Shannon Weaver definition of species diversity this seems not to be the case.

By going into a forest and thinning the dominant species we increase the relative proportions of the rarer species, and hence we increase the Shannon-Weaver species diversity of the forest (of course there are potentially many other aspects of the system dynamics; dispersal, seeding, survival and mortality and speciation that might be affected... but of these I can say little or nothing). Hence by "interfering" in the forest and creating gaps for the rarer species, and increased potential speciation potential for the rarer species, it might be possible to manage the "conservation" of biodiversity better than nature can do; maybe we can push local reserves up the species-area curve! This seems a bit of a heresy in conservation ecology, i.e the idea of "increasing conservation value by improving on the best that nature can do".... but this does seem to be possible, and we would just be going along with the Second Law!

Second law of thermodynamics

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

The second law of thermodynamics is often misused, and I think that we should be extremely cautious about borrowing it to describe ecosystems or evolution or how species behave.

The law states, in short, that entropy tends to increase in any closed system. But there are, essentially, no closed systems. Under some assumptions about its geometry, the entire Universe is a closed system, but almost every other natural system can accept or lose energy across its boundaries. Quite obviously the Earth is an open system since it constantly receives energy from the sun and radiates energy into space. Plants use the sun's energy to reverse entropy by making leaves and things, and animals reverse their own entropy while decreasing that of plants by eating them. The net result is that something extraordinarily disorganised - the sun's surface - is used to generate something much more organised - you and me.

I therefore completely disagree that "nature, left to its own devices will maximize entropy pretty effectively" when it comes to natural populations. The evidence is entirely against this statement; in its place we have natural selection, and the gradual increase in complexity of the biodiversity of our planet as more and more niches are created and exploited as species compete, co-operate and slowly evolve. At least, natural complexity increased until humans evolved, when the complexity started to decrease in the natural world to drive greatly increased complexity in our societies and technologies.

Ecosystem approach-ecological process

Alejandro J. Rescia Perazzo, Universidad Complutense de Madrid, Madrid. Spain

The issue of scale has demonstrated to have great interest in ecology. Based on the literature about this issue, we can distinguish two fundamental aspects: one of them related to spatial management and planning –conservation strategies will be included here- (strictly spatial) and the other related to the resources exploitation by the organisms –organism's perspective-. These two aspects deal with the interrelationship among observation scale (grain and extent), organism's perspective (cross-scale resources exploitation) and temporal and spatial scales of phenomenon occurrences (disturbance regime and fast-slow variables).

Currently, we can express some assertions:

- Ecosystem stability and ecological processes vary directly with diversity in species and functional type
- Species properties expressed as functional types exert a greater degree of control on ecosystem function than species diversity (richness) And we achieve a consensus with regard to biodiversity and ecosystem= function:
- Some minimum number of species is essential for maintaining ecosystem function under constant conditions
- A larger number of species is probably required for maintaining ecosystems in changing environments
- Determining which species have significant impact, on which processes and in which ecosystem remains an open empirical question
- Mechanisms for generating determined processes may range from systems with a few dominant species or functional types (low diversity) to systems with high diversity, low level dominance and high complementarity

This consensus emerges from a 'change of vision' on ecosystems' stability which emphasizes on function (i.e. functional diversity) more than on structure (i.e. species diversity). In any case, there are different ways to explain ecological organization of ecosystems. Concomitantly, the ecosystem approach in landscape management, in conservation biology and restoration ecology is growing up.

Therefore, is important to highlight a land management from an ecosystem approach in order to maintain ecological processes ensuring the ecosystem function. This approach will correspond with a 'horizontal vision' of conservation -based on the interrelation among species and ecological processes- that is different from a 'vertical vision' -based on certain species conservation-. This idea suggests that the presence of an emblematic species or an endangered species (anyone susceptible of conservation) involves a good degree of conservation of their habitats and ecosystems in general.

Scales and metapopulation

Michel Baguette, Université catholique de Louvain, Unité d'écologie et de biogeography, Louvain-La-Neuve, Belgium

SUMMARY: Landscape is the relevant scale for nature conservation in man-shaped landscape. Metapopulation models are useful guides for the design of efficient networks of suitable habitats allowing the long-term persistence of target species.

There is a growing insight that space has to be explicitly taken into account in the analysis of ecological and evolutionary processes. This recent trend generated powerful advances, leading to the development of the metapopulation and metacommunity concepts, which consist in now well-defined interacting levels of biological integration with their own properties. However, although both concepts were developed to incorporate spatial aspects in dynamics and evolution of populations and communities respectively, the spatial scale to which these concepts apply are not yet comprehensively defined. Metapopulations are considered as "sets of local populations, within some larger areas, where typically migration from one population to at least some other patches is possible" (Hanski & Simberloff 1997; Hanski 1999 p11). Metacommunities "consist of all trophically similar individuals and species in a regional collection of local communities" (Hubbell 2001 p5). "Some larger areas" or "regional collection" refer to vague spatial delimitation, whereas according to the hierarchy of ecological and evolutionary systems (Blondel 1987), each level of the biological hierarchy corresponds to restricted domain of space, time and change.

Here I propose that landscape is the relevant spatial scale corresponding to metacommunity and metapopulation dynamics and evolution. I refer to a global, biogeographical definition of landscape, namely a portion of the geographic space showing homogenous geomorphology and determinate climatic features. According to this definition (refined from Forman & Godron 1986; Blondel 1995), landscapes have no precise spatial scale; however, it does not mean that landscapes are dimensionless. Every region of the biosphere may be easily partitioned into various-sized landscapes, usually between 1000 and 10.000 km² on the basis of their particular geomorphology and climate regimes. Natural landscapes are far from homogenous but should be rather considered as a dynamic, shifting mosaic of various stages of the ecological succession. The position of succession stages is not fixed in the landscape but depends on natural disturbances, which perturb locally the climatic stage and re-start ecological successions. Natural landscapes submitted to a disturbance regime may be considered in a dynamic equilibrium: every stage of the succession exists somewhere in the matrix, according to the place where the perturbation occurred. Ecological successions progressively restore climatic stages until the next perturbation.

The current challenge of conservation biology is to conciliate biodiversity and human-shaped landscape management. Key research questions for nature conservation are (1) to what extent can landscapes be managed by humans while their natural functioning remains and (2) in man-shaped landscapes, how can the natural functioning be mimicked, to allow the persistence of all the species of the metacommunities?

The first question is no more relevant to the European context. The second question needs researches on (1) what are the species with a conservation concern, (2) what are their habitat requirements (including the management of these habitats), (3) what are their (meta)population dynamics and genetics. Research keywords are habitat quality, habitat management, metapopulation modelling, dispersal, landscape connectivity, ecological networks. The final aim is to design networks of suitable habitats in man-shaped landscapes, allowing the long-term persistence of the target species.

Marine ecoregion conservation

Stephan Lutter, WWF. Germany

SUMMARY: This keynote contribution addresses advantages and limitations of large-scale ecoregion conservation in the marine environment.

Working on nature's terms: ecoregion conservation is about conservation beyond political and geographical boundaries. It is about thinking big scale and over the long-term. It is about conserving or even restoring the full range of natural diversity. It provides the most efficient way to deal with threats to the world's most important natural systems, and the context to work at all levels - practical local level to high level policy.

How to define ecoregions: in the highly dynamic marine environment, the concept of Large Marine Ecosystems (LME) appears to be most appropriate to set ecoregion boundaries. It allows to encompass critical habitats and biodiversity features in compartments to which similar patterns of threats and management responses apply. However, the concept still needs to be further extended and systematically applied to the open ocean and deep sea with its three-dimensional seascapes.

From theory to practice: bearing in mind that it is human activities that we aim to manage not ecosystems, we may have a number of pragmatic considerations before finally taking a decision on the scale of marine ecosystem research and conservation: for example, WWF's North-East Atlantic Shelf Ecoregion was defined as a merger of the Biscay, Celtic and North Seas, following the shelf contour and including part of the continental slope, as well as the adjacent Baltic Sea. Due to the distinct differences between shelf and deep sea realm and in light of homogenous socio-economic and political conditions this seemed to be the most appropriate solution at operational level. This modification was of secondary importance because we continued to scale both our biodiversity vision and our efforts to reduce human impacts around ecologically meaningful areas within the ecoregion chosen, such as: the continental margin, the Western Approaches, the Celtic Shelf, the Irish Sea, the southern North and Wadden Sea and the Baltic Sea.

From a practitioner's point of view, the issue of appropriate biological scales and marine conservation is most relevant to human activities or mitigation measures to be tackled at sea such as fisheries, extractive industries, maritime transport - as well as the associated marine conservation and spatial planning responses be them recovery plans for populations and stocks, networks of marine protected areas, migration corridors, fisheries closures or no-take-zones. Where fish stocks and businesses straddle boundaries at the same time there is no point in sticking to the traditional site-by-site approach nor in keeping conservation strategies separated within territorial waters or Exclusive Economic Zones (EEZs). There are even more human interactions with the marine ecosystem that merit being addressed at ecoregional (rather than local or global) level: river catchment loads and eutrophication problem areas; climate change adaptation and resilience; and so forth. But when it comes to the fate, effect and elimination of toxic, persistent and bio-accumulative chemicals from land-based diffuse sources and their long-range transboundary transport, the ecoregion approach often has limitations rather than advantages.

In the above context, it is interesting to study the evolution of ocean governance and marine conservation policies in Europe: in the 70s, the need for transboundary international coo-operation was first recognised and enshrined in regional seas agreements such as the Oslo, Paris and Helsinki conventions. In the 80s and 90s, these frameworks gradually moved from a piecemeal to a holistic approach addressing the whole range of human impacts and even attempting to reconcile fisheries and environment - albeit within unnatural geopolitical boundaries. In the current decade, OSPAR, HELCOM and the EU through their European Marine Strategy (EMS) turn to the ecological scales and commit themselves to an ecosystem approach to be implemented within smaller-scale ecoregion-type units.

Biological scales and conservation: Summary Week 1 from a terrestrial perspective

András Báldi, Animal Ecology Research Group, HAS, Hungarian Natural History Museum, Session II Chair.

The aim of the first week of the session on "Biological scales and conservation" was to highlight some key issues, and some general concepts on scale effect and conservation. An apparent issue was the difference between marine and terrestrial habitats. Due to the diversity of the topic, the approaches were also diverse.

Prominent contributors (Paul Opdam, Bill Kunin, Jorgi Bascompte, Michel Baguette) shared their view on how to deal with scale issue, and highlighted the key research questions or problems:

- To find methods and indicators to determine ecosystem networks, which can sustain metapopulations of different species in a human dominated landscape.
- To understand spatial responses of metapopulations to disturbances at a range of scales.
- To identify critical thresholds for metapopulation persistence.
- To link administrative scales (EU, national, region, local) with biological scales.
- Reserve design is depend on both spatial and temporal scales; if large amounts of land are being discussed, subdividing it may be preferred, whereas when less land is available, it might best be kept as a single reserve (at least in the long run). Identifying these critical cut-off scales is a key question for future research.
- Similar cut-off scales should be determined for reserve shape and proximity.
- To identify appropriate scales for the conservation of networks of ecological interactions.
- To identify important species, their habitat requirement, and their metapopulation dynamics with the final aim to design a network of suitable habitats allowing long-term persistence.

Next week we intend to learn the views of experts of different ecosystems and habitats on scale issues.

Biological scales and conservation: Summary Week 1 from a marine perspective

Keith Hiscock, Marine Biological Association of the UK, Plymouth, UK, Session II Chair

From a marine perspective, it has been interesting to see the thrust of contributions from terrestrial colleagues. As hoped, there are useful lessons for marine ecology to learn.

Paul Opdam raised the issues of timescales for conservation – especially predicting effectiveness of conservation measures – something I feel that we have only become reasonably good at in a science-based way in the past couple of years in marine. Perhaps the major difference for marine is that many habitats are little altered by human activities and restoration is rarely an active management process – just 'stop what did the damage'.

Bill Kunin mentions the 'Single Large Or Several Small' reserves question – something that marine ecologists have not generally addressed, perhaps because of the 'bigger the better' criterion in the Habitats Directive that has driven site-based marine conservation for the past ten years or more. Now, maybe, we have a chance to re-visit such questions as we gear-up to achieve the imperative from the World Summit on Sustainable Development in 2002 to establish a network of marine protected areas by 2012.

Peter Duelli and correspondents look to the use of morpho species and parataxonomic units as a quick way of assessing biodiversity. In the marine environment, 'taxonomic distinctiveness' in which counts of the number of major taxonomic groups represented in a sample are made rather than identifying every taxon to species, is gaining popularity as a method of marine biodiversity assessment. (See, for instance, Clarke, K. R., Warwick, R. M. 1998. A taxonomic distinctiveness index and its statistical properties. Journal of Applied Ecology, 35, 523.531.)

Perhaps there was a misunderstanding in Keith Rennolls comment about 'rarity' in and the link made to species-area curves. The point that was made by Jean-Luc Solandt referred to the fact that we marine ecologists just do not have the information needed to identify degree of threat (often synonomised as 'rarity') that is available for percentage loss of hectares of a habitat or percentage decline in numbers of a species that Red List criteria rely on. There are issues surrounding the 'what is rare?' question but it is essential that marine ecologists have pragmatic tools that employ descriptive terms ('rare', 'scarce', 'severe decline' etc.) that politicians and the public can relate to.

So, just what can marine ecologists and conservationists learn from terrestrial experience – or are we really so different?

Marine vs. terrestrial: are we really so different?

Ferdinando Boero, DiSTeBA (Dipartimento di Scienze e Tecnologie Biologiche e Ambientali) Universita' di Lecce, Italy

The EU Habitat directive... did you read the number of habitat types? Did you see how many marine habitats are listed? There are nine marine habitat types (if I remember well) against some 150 terrestrial ones! This is simply ridiculous. Obviously there is not a thorough appreciation of marine biodiversity at an European level. We should work more on this, so as to have a comprehensive list of marine habitat types in Europe and have it accepted in an amended version of the Directive. The Barcelona convention lists habitat types from the Mediterranean, but it is not enough.

As for species, marine ecologists often do not have the taxonomic knowledge to recognize a rare species. For many groups there are very old monographs, and for even a larger number of groups there are no monographs at all. It is maybe useful to find surrogates to species for some purposes, but if species are not covered in biodiversity evaluation, who should cover them? I know it is difficult, I know it is time consuming, I know that the expertise is vanishing. And these are the reasons why people look for surrogates, for proxies. In doing so, while speaking about biodiversity, we end up admitting that species are irrelevant to biodiversity evaluation. Since we try to find ways to avoid their identification. Taxonomic sufficiency, distinctness, you name it, are being developed because we are not able to identify species, besides the more obvious ones. With a bunch of colleagues, I have just published the first monograph on the Hydrozoa of the Mediterranean (you can download it from the web page of Scientia Marina). This means that there was no other before a few months ago. That Fauna is already obsolete, new revisions are being made, new species are being found. If a hydroid species disappears, who has the possibility of appreciating the loss? Is the loss of a tiny hydrozoan less important than the loss of a more conspicuous species? On what ground do we base such assumption? Ecosystems continued to work even after the disappearance of dinosaurs, so who cares? But they continue to work even after the disappearance of the monk seal. What is the idea of the environment that we are producing? Is it some sort of zoo, with nice animals to see? Extracted from their context? The context are the tiny little things.

More: having been educated a catholic; I was taught that God made nature for us to take advantage of. It took me a long time to get rid of this vision. Now I hear people saying that biodiversity is important because of the goods and services that it provides us. This is what the priest was telling me. What about the species that do not provide goods and services? Are they expendable? If they are, we can get rid of the greatest majority of species (after all we do not even know that they exist, so... who cares?).

Where do I want to arrive with this? We need a sound taxonomy of marine habitat types and we have to enforce it at a EU level in the Habitat directive. And we need taxonomic revisions and monographs for all groups. The first thing is at easy reach over the short-medium term, the second one is not easy at all, and it will require a long term investment. But if we never start, we'll never make it. In another forum I complained about this and was given a long list of EU projects on taxonomy. But they all served to provide services to taxonomy, not a single one was aimed at real work, work in which a taxonomist might have used those beautiful services. Isn't this paradoxical? Beautiful services for no one. If we do not consider species (not only the obvious ones) in biodiversity research, where else species will deserve attention? Let's be honest, if the scientific community is convinced that all these tiny species are just a nuisance to productive research and thinks that disentangling their diversity is a useless task, then I am out of this game. If we think that, together with many other things, also this issue is important, (I repeat: ALSO this issue is important, I am not saying that it is the only one to be important) then maybe we should invest some effort in it, besides providing services that nobody will use because there are no EU projects on this topic. I understand that politicians are difficult to convince if one has to explain the importance of gnatostomulids, does this mean that gnatostomulids are irrelevant? There has to be a difference between the green movement and professional biodiversitologists, I think. Of course it is easier to speak about the importance of whales and dolphins than about the importance of bacteria, but at the end bacteria are more important than whales and dolphins. And maybe they are endangered by antibiotics and hormones. Let's save the bacteria!

Re: Marine vs. terrestrial: are we really so different?

Keith Hiscock, Marine Biological Association of the UK, Plymouth, UK, Session II Chair

Ferdinando Boero draws attention to the awfulness of the Annex I marine habitats in the Habitats Directive. They are something that we have had to live with and make the best of in marine conservation and I believe that the Habitats Directive has significantly improved some aspects of marine environmental protection in the UK at least.

The CORINE classification that provided the 'catalogue' from which habitats for protection were selected in 1989-91 has been replaced with the EUNIS (European Nature Information System) classification (http://eunis.eea.eu.int/index.jsp) and UK has taken a lead in developing the marine habitats classification (see: http://www.jncc.gov.uk/marinehabitatsclassification). The classifications are hierarchical (scaled) and there is a lot of effort in the UK being put into identifying which level can best be used to identify marine protected areas and to structure quality assessment for the Water Framework Directive etc. Have a look and see what you think.

Yes, I am concerned that we are not seeing the marine species that might be becoming extinct - the only marine species that I can cite in lectures as having been made extinct in the NE Atlantic by human activities is the great auk. Any more?

As for 'goods and services', the concept includes 'amenity resources', 'recreation and ecotourism', 'refugium value' etc. but I would welcome more discussion about 'goods and services' concepts at different scales for marine conservation.

Marine and terrestrial: all in the same melting pot

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

Dr Boero raises some provocative points in his contribution "Marine vs Terrestrial, are we really so different?"

For the record, Annex 1 of the Habitats Directive classes 8 habitats under "Open sea and tidal areas". These include: Sandbanks which are slightly covered by sea water all the time, Posidonia beds (Posidonion oceanicae), Estuaries, Mudflats and sandflats not covered by seawater at low tide, Coastal lagoons, Large shallow inlets and bays, Reefs, and Submarine structures made by leaking gases. This hardly seems to capture the richness of marine habitats in European waters, and indeed the Council Directive 97/62/EC of 27 October 1997 replaced the designation "open sea and tidal areas" with the more accurate "coastal and halophytic habitats". The Directive then goes on to list 182 non-marine habitats, including for example 5 kinds of sea cliffs and shingle or stony beaches, 18 habitats in fresh water, 19 habitats in temperate forests and 13 in Mediterranean deciduous forests.

This does not mean, however, that "there is not a thorough appreciation of marine biodiversity at an European level" as Dr . Boero would have it. It is more a reflection of the mind-set of European nations at the time that they were drawing up the Directive. The heart of the Habitats Directive is in Article 3, which states that "a coherent European ecological network of special areas of conservation shall be set up under the title Natura 2000". At that time, there was no political agreement on marine protected areas, so there was little point in expending effort on classifying marine habitats. Times are changing, and as more Member States come to agree on the importance of marine protected areas, we may perhaps expect to see a more thorough classification of marine habitats in a future revision of Annex 1.

I cannot go along with Dr. Boero's statement that there is a tendency to behave as though species are irrelevant to biodiversity evaluation because we can no longer identify them. In my view we seek surrogate "indicators" because the alternative (of some exhaustive inventory of taxa) is simply impractical. We "try to find ways to avoid their identification" not because we lack the capacity (which may also be the case) but because the problem of counting things one by one is intractable.

It does not surprise me that there was no previous monograph on Mediterranean hydroid species- we keep hearing that the ratio of unknown to known species is perhaps 10 to 1, or 100 to 1, so new monographs will appear all the time for a long time to come. Dr Boero raises a very important and far-reaching question: "Is the loss of a tiny hydrozoan less important than the loss of a more conspicuous species?". Ecosystems continued to work even after the disappearance of dinosaurs, so who cares?"

I have given this question a lot of thought. Why conserve biodiversity? There are many reasons, but consider this: it is an issue of scale. Our efforts to conserve or destroy biodiversity do not matter much in the longer view. No matter what we do, in 10 million years, you and I will be dust in the wind, and my beliefs (and yours too)will be unutterably irrelevant. Even if we sear terrestrial and freshwater biodiversity from the face of the Earth, life in the deep ocean will probably survive us. From there, given time, life in a myriad of miraculous forms and intricate networks will once again invade emergent land. In the unthinkably distant future, the rain will fall on a wild, savage and biodiverse world, just as it did for hundreds of millions of years before humans arose to decimate the planet. Despite all we can do to obliterate life, in 10 or 20 million years the planet will probably be just about as biologically diverse as it is today, although few of the species, or genera, or families or possibly even orders of organisms will be the same then as they are now. There will probably be flying animals, but they may not be insects or birds or mammals, and perhaps not even their descendants. In that remote and unimaginable future, whether we managed by 2010 to save this or that threatened species of bird will not matter at all.

But our horizons are not fixed 10 million years in the future. Our horizons are at a human scale, and at that scale, it matters what you and I believe and do. Biodiversity matters, not just because of its patient, 3500-million-year intrinsic value, but because it matters to humans. We must do what we can to protect life on our world because humans are concerned, humans are part of biodiversity, humans are deleting untold numbers of species, humans can do something about it, humans affect and are affected by the living world around them. Humans are the engineers of climate change, habitat fragmentation, pollution, over-harvesting, over-exploitation, invasive species, and all the other woes that we inflict on our living planet. And yet we can only survive on this planet because of the other species that share it with us. Biodiversity certainly doesn't need humans, but humans, all of us, up to and including 21st century humans, need biodiversity.

Let me go back to Dr. Boero's contribution, where he makes another highly significant remark concerning the relationship between humans and "nature". Unlike him, I was not raised in the Catholic faith, but I was drenched in Anglicanism, which I suppose is the next best thing. I read in Genesis 1:28 "And God blessed them, and God said unto them, Be fruitful, and multiply, and replenish the earth, and subdue it: and have dominion over the fish of the sea, and over the fowl of the air, and over every living thing that moveth upon the earth". I am viscerally opposed to this vision of god-given human ascendancy, but I agree that this attitude permeates much of the thinking of many industrialised nations. But I don't think that this is why goods and services are an important concept. My view is perhaps cynical: biodiversity will survive anyway, but humans won't. If we want to do something about conserving biodiversity today, for the sake of future generations of humans, we must realise that we depend on biodiversity. Which means? That we depend on goods and services provided by the living world - whatever those services may be.

But, as Dr. Boero asks, "what about the species that do not provide goods and services? Are they expendable?"

Except from a moral or ethical perspective, does extinction matter? Most species that go extinct were previously unknown to science, and therefore by definition of no known use to people. And most of them were also rare, so presumably were of no great ecological significance to the ecosystems in which they lived. From a purely utilitarian perspective, the extinction of rare, unknown species is of interest to nobody but bleeding-heart liberals.

Perhaps unsurprisingly, this again is an issue of scale - in this case, the scale of the extinctions. Few species are hyper-rare. Few species are going extinct. Most species are neither very rare (with populations of only tens or hundreds of individuals) nor very common (populations in the millions or billions). But the populations of almost every known species are decreasing; most species are getting rarer. The rate of extinction will accelerate as the great bulk of species become rarer and rarer. As that happens, it is not only the bleeding-heart liberals who will find that extinction matters.

So in answer to the question "are they expendable?" I would argue, yes, they are, if you just look at species one by one. But its passing is symptomatic of a coming problem. Think of it as one of the first pebbles cascading down ahead of the landslide.

I sympathise with Dr. Boero's view that the "long list of EU projects on taxonomy all served to provide services to taxonomy, not a single one was aimed at real work". If we define "real work" to mean "alpha taxonomy", then this is a valid criticism. It reflects a world view that perhaps we should all try to change.

An example of spatial scale and marshland management

András Báldi, Animal Ecology Research Group, HAS, Hungarian Natural History Museum, Session II Chair

SUMMARY: The suitability of marshlands for marshland nesting passerine birds depends on spatial scale: reedbed edges are preferred by birds on local scale at few meters resolution, while a general lack of preference was found at larger spatial scale at hundred meters resolution. Therefore, observations on these spatial scales yield different guidance for reedbed management.

This week we will have contributions from experts on different habitats. I expect that their view on spatial and temporal scales are rather different, due to the specificity of systems they consider. Instead of a longer introduction, I provide a short contribution based on our marshland spatial pattern studies.

Marshlands are endangered habitats all over Europe. Most of them were drained during history to get agricultural areas, and only the very last remnants exist now. Therefore, their proper management is crucial to maintain this important component of the overall biodiversity of our continent. Is there a scale effect influence management, and if so, how does this affect the potential practical guidelines? I will show that depending on the spatial scale, different management guidelines seem to be the best. Thus, this contribution can be viewed as a specific example of Bill Kunin's last week contribution, where he showed that contradictions in reserve design (e.g. the Single Large Or Several Small debate) is simply a result of considering different spatial scales.

How should marshland heterogeneity be managed? Are reedbed edges and discontinuities, or homogeneous stands preferable? The answer depends on spatial scale. On a local scale, with a spatial resolution of a few meters, all observed passerine bird species preferred edges in a large reedbed in Hungary (Báldi and Kisbenedek 1999). However, on a larger spatial scale, with a resolution of hundred meters, most species had no preference. Therefore, the majority of the species had different preference for habitat discontinuity at different spatial scales, which makes any simply reedbed management practice insufficient and incomplete.

The solution is (1) to understand the mechanism behind the patterns, and (2) to give very specific questions on the target of management. Regarding the first point, fortunately, the mechanisms are suspected: on local scale with a few meter resolution the preference for foraging and nesting site within territories was observed, while on larger spatial scale the distribution of territories was detected. Therefore, larger spatial scale should be the level of management, where the number of territories, i.e. number of birds, can be increased, while at the local scale only within territory movements can be manipulated. Regarding management of spatial heterogeneity of this reedbed, the target object must be specified clearly (e.g., overall diversity of birds, or abundance of an individual species) with the acknowledgement that to manage the reedbed for any given aim will harm other potential aims.

This case study from marshland has limits: it deals only with birds, and only two spatial scales were analysed. Therefore, if other taxa, and other scales, for example the network of marshlands are also included, the picture become extremely complex.

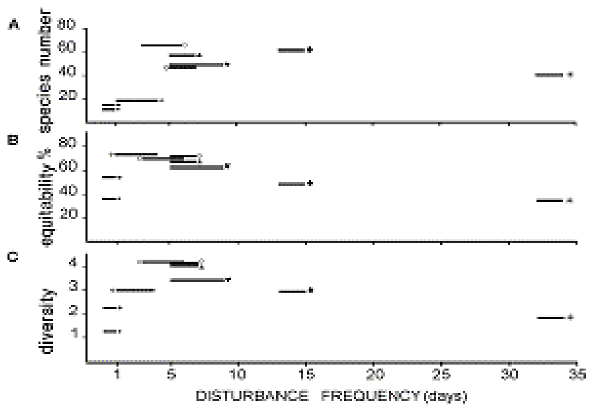
Spatial and temporal scales and algal communities

Judit Padisák, Department of Limnology, University of Veszprém, Hungary

SUMMARY: The topic and importance of scaling has become a major area of research in community ecology in the recent past. Scales are usually understood as spatial and temporal scales. Unlike primary producers of the terrestrial ecosystems, algal communities are usually the smallest and shortest living organisms of aquatic environments, often embedded in a viscous medium (phytoplankton). Short generation times make algal communities ideal test object for studying temporal scales, however and in return, microscopic sizes make handling of spatial scales difficult.

Considerable progress has been made in the past decades in understanding the temporal aspects of the diversity-disturbance relationship of phytoplantkon based largely on Connell's Intermediate Disturbance Hypothesis (IDH)(Padisak et al. 1993, Naselli et al. 2003). As summary, countless studies proved that. On the scale of days to weeks, frequency of disturbance is largely responsible for high diversity of phytoplankton communities and can be summarized as shown on Figure 1.

Figure 1. Dependence of compositional diversity and its two components on frequency of disturbance. In accordance with the IDH diversity maximizes at intermediate frequencies like its components. However, initial slope of increase in equitability is substantially steeper than that of species number suggesting that colonization/extinction processes are slower than population growth that is often exponential so far as resources allow. Note that disturbance frequencies are quantified and, for phytoplankton communities, it can be measured on scales of days (Padisák, J., 1994)



Annual changes of phytoplankton assemblages are well known as seasonal succession, and they involve appearance/disappearance of the populations in a given lake at an annual scale. Because dozens of generations are involved in one annual cycle, the seasonal succession carries more similarities with succession of terrestrial communities than with their seasonal changes. In absence of major man induced driving forces (like eutrophication, acidification, etc.) annual patterns are highly repetitive in a given lake, and often similar in lakes of similar type. Sudden changes in food-web structures normally influence phytoplankton pattern on annual scales, however, if impact is very intense it may have effects expanding for several years (Borics et al. 2000).

Increasing evidence is accumulating on major effects of mesoclimatic cycles (El Niño, NAO) on sudden vegetation changes of the pelagic environment. Most species exhibit interannual oscillations at decadal scale in Lake Ferto/Neusiedlersee (Hungary/Austria) (Padisak 1998), Planktothrix rubescens seems to be recurrent after winters with long-lasting ice cover in Lake Stechlin, as Aulacoseira islandica in the intermediate periods in the same lake (Padisák et al. 2003, Padisák et al. 2004). The El Niño effect enhanced the dominance of the invading (Padisák 1997.) cyanaprokaryote, Cylindrospermopsis raciborskii in South

American reservoirs (Bouvy et al. 2003). Mechanisms, and especially triggers, of such events are completely unexplored despite global warming would need extended basic research.

As diatoms are superior indicators of pH changes even at scales of centuries to millennia, the composition of frustules in lake sediment cores are often used palaeolimnology. Such analyses evidenced an overall, natural acidification of snowmelt lakes in the last late glacial and early postglacial period caused by increasing humic substances originating from the developing terrestrial vegetation (Steinberg 2003). Unfortunately, number of case studies is very limited.

The term “landscape” is not often used in context of subaquatic environments although number of advertisements like “Explore the underwater landscape!” try to attract potential divers, in for example, the famous coral reef area of the Red Sea. Coral reefs are attracting for human eye and on scale of human vision, however, there is no any reason to ignore the approach for less attracting underwater environments. Another example is the recently emerging deep-sea research, especially vicinity of hydrothermal vents, where mapping methods very similar to those used in the land are standard techniques. In continental waters the landscape approach is largely ignored, or, better to say, unrecognized despite number of cases substantiate its usefulness. Weather a lake sediment consists of fine- or coarse grained particles largely determines its invertebrate fauna. The exceptionally large standing crop of signal crayfish in Lake Erken, Sweden, as compared to neighboring lakes, can be attributed to its rocky sediment. In small streams dominance rocky/course/fine grained river-bed largely determines its macroinvertebrate fauna and this feature is widely used in monitoring networks (for example, in the Water Framework Directives). Turbidity of shallow lakes is also depend on sediment properties and might have basic influence on underwater vision of fish and other animals that capture their prey on basis of visual perception. This alone might have cascading influence on complete food webs ending at the primary producer phytoplankton.

Spatial heterogeneity and the consequent habitat diversity plays a key role in maintaining high algal biodiversity. The best example is a small transitional bog, Baláta-tó (Hungary) where close to 400 species of algae were found without diatoms (Borics et al. 1998). Although relationships between habitat diversity and biodiversity of algae has been intuitively well known since probably a century, mechanisms are largely unexplored. One known example is the case of the planktonic, halophil centric diatom, Chaetoceros muelleri in Lake Ferto (Padisák & Dokulil 1994). The species forms relatively large open-water populations during the dry years when salinity of the lake is the highest. In dilution periods it is restricted to the small open water areas enclosed in the extended reed-belt of the lake.

We may conclude that while due to short generation times temporal scales are relatively easy to study in algal assemblages, importance of spatial heterogeneity is only intuitively understood.

Successional scales of ecosystem and conservation

Eduardas Budrys, Institute of Ecology of Vilnius University, Vilnius, Lithuania

SUMMARY: According to Razumovsky (1981), each local habitat is a mosaic of successional stages and a fragment (in both time and space) of a regional successional system. Habitats and species exist within niches that may be measured in successional scales. Therefore, understanding of the successional system is of particular importance for conservation of both habitats and species.

Razumovsky (1981) has stated that an ecosystem at the regional scale is functioning as a successional system, containing endogenous successions of two scales, the ecogenesis and demutation.

Ecogenesis represents development of soil in the local mineral and climatic conditions toward an equilibrium state (pedoclimax), together with the biotic community on it (climax community). It is triggered by landscape-level processes like orogenesis, linear erosion, aeolic deposition, epeirogenic uplift, glaciation, etc. The estimated duration of ecogenesis is several hundreds or even thousands of years, thus it is tangible only using geological methods. Therefore, our knowledge of ecogenesis is much more fragmental than that of the successions of smaller scales; often it is not considered as a succession at all.

Demutation is a recovery of forest vegetation after its removal at the late (wooded) stages of ecogenesis. The natural triggers of demutation are fire, windthrows and grazing of megaherbivores. The estimated duration of demutation is few hundreds of years. This type of succession is often considered as just a forest succession on particular type of soil, and its final stage is then regarded as a climax, actually being a stage of ecogenesis.

In sense of Razumovsky, a habitat represents a mosaic of stages of demutation (demutational complex), or a mosaic of demutations belonging to different stages of ecogenesis (ecogenetic complex). Local set of disturbances may turn them to a virtually stable mosaic of subclimaxes, or to succession-like (however, exogenous) reversal or cyclic changes of vegetation.

Razumovsky did not consider the grass community changes, driven by stochastical processes of plant dispersal after disturbances and more dependent on differences of plant vagility than on their competitiveness, successions. These changes, being less predictable, have nevertheless internal statistical regularities, thus, to my opinion, may be regarded as a kind of successions (“microsuccessions”). They have duration of few years or decades only; therefore they are easier to study than ecogenesis or demutation. Most of criticism against the succession theory arose from the studies of microsuccessional processes. However, this criticism has little stimulated exploration of e.g. typology and differences of mechanisms in distinct types and scales of successions.

Razumovsky has over-estimated the significance of allelopathic interactions and possibly has underestimated the stochastic processes of plant dispersal and the competitive interactions. Therefore, his approach has been criticized in Russia and unnoticed or disregarded in the other countries. However, his synecological views have been accepted and developed, between few others, by palaeontologist V.V. Zherikhin (2003) as a constituent part of the concept of phylocoenogenesis. Zherikhin considered the self-reproducing successional system in sense of Razumovsky a minimal evolutionary unit at the community level. He has pointed out that the successional system in sense of Razumovsky should be regarded like the Ideal Gas: it is absent in the nature but all laws of the gas physics appeal to it.

The currently used habitat and plant community classifications refer to the presence and proportions of plant species. These classifications are equivalent to the early systems of plants based on the number of stamina, or those of animals based on presence/absence of blood. The phytodynamic or phylocoenogenetic classification of habitats, based on their successional “ontogeny” (location in the hyperspace of scales of the successional system) and “phylogeny” (evolutionary formation of that successional system) is not yet elaborated. Thus, our current level of knowledge in the synecology may be compared with that in the pre-Darwinian biology.

The knowledge of the regional successional system is crucial for understanding of local ecosystem structure, functioning, conservation of its elements - habitats and species, and restoration. Therefore, study of the successional scales of ecosystem, the typology and mechanisms of successions must be a priority of the synecological research.

Spatial heterogeneity and management

Alejandro J. Rescia Perazzo, Universidad Complutense de Madrid, Madrid. Spain

The scale is a crucial aspect of ecological heterogeneity. The latter is a function of the former and interpretation of this depends on the level of observation established on studying an ecological system.

Because there is no fundamental scale of investigation, study of the dynamics of scale in a landscape must therefore be carried out taking into account the different observation scales: a given landscape can be heterogeneous on one scale and homogeneous on another.

Studying change in the landscape by means of multi-scale focus -both spatial and temporal- allows us to specify with different levels of detail the characteristics of the processes of change and, at the same time, helps to direct land management at different levels. This is particularly actual in fragmented landscapes. The recognition of scale dependence suggest the use of hierarchical management strategies centred on characteristic scales at which ecological phenomena of interest occur. Analysis of the landscape structure is therefore essential in order to conserve the communities and the biological and cultural diversity.

In fragmented landscapes, human activities on a local scale strongly influence the spatial configuration on a landscape scale. This concerns spatial planning, the main problem of which is not found in selecting a correct observation scale, but in recognizing that the change under study occurs at various scales at the same time. This underlines the importance of analyzing the landscape pattern at different scales with the design of plans and conservation programs in mind.

Taking a birds-eye view to landscape management – the importance of spatial scale and connectivity

Richard Johnson, Department of Environmental Assessment, Swedish University of Agricultural Sciences, Uppsala, Sweden

Ecosystems and their communities are dynamic, reflecting the effects of both natural as well as human-induced stressors. Geographically distinct regions typically have their own distinct biota due to the random processes of colonisation, but also the interplay between biotic and abiotic processes that drive evolutionary change (Ricklefs, 1987). Early studies suggested that local diversity was a product of species interactions, though more recently the role of regional factors and history in regulating the organisation of biotic communities has been recognised (e.g. Cornell, 1999). According to hierarchy theory, physical and biological variables on a small spatial scale are influenced by variables on larger spatial scales (Allen & Starr, 1982; O´Neill et al., 1986). Consequently, although habitat availability is considered to be the template that shapes organisms life-history strategies (e.g. Southwood, 1977; Townsend & Hildrew, 1994), the environmental characteristics of a specific ecosystem (e.g. stream) are not random, but are considered to be controlled by macro-scale geomorphic patterns (Frissell et al., 1986). Building on this premise, a conceptual framework has been developed in which (stream) communities at a site can be seen as a product of a series of "filters" (e.g. continental, regional, basin, reach and habitat) (e.g. Tonn, 1990; Poff, 1997), through which species occurring at a site have had to pass. The importance of ecological scale in ecology is an expanding area, and indeed according to Thompson et al. (2001) the topic of scale is one of the four paramount frontiers in ecology for "understanding how biological and physical processes interact over multiple spatial and temporal scales to shape the earths´ biodiversity".

Few birds (or humans looking out of an airplane window) would contest the observation that much of the landscape of Europe is fragmented by land use practices. Indeed, the European landscape has been altered for centuries, rendering the identification of pristine or even relatively minimally disturbed ecosystems difficult. Although the type and severity of human-generated stressor(s) affecting freshwater ecosystems differ across Europe, the major drivers affecting the biodiversity of terrestrial and aquatic ecosystems are similar (Young et al. in press). For example, regarding aquatic ecosystems the major pan-European stressors can be summarized as overexploitation, nutrient/organic enrichment, acidification and alterations of hydrology. For terrestrial ecosystems, the most common stressors affecting biodiversity are nutrients from agriculture, forestry/afforestation, and habitat fragmental and loss (e.g. loss of grassland due to land abandonment) (see BIOFORUM report I, Young et al., 2003). Unfortunately, there has been a cleft in the way that terrestrial and aquatic ecologists perceive and manage their ecosystems (e.g. Grimm et al. 2003) and this dichotomy has hindered the development of more holistic management approaches. This development is particularly alarming given that present-day problems and management objects (e.g. WFD and CBD work) require a more holistic approach to ecosystem management. For example, although past monitoring and restoration projects have often been concerned individual sites (e.g. lakes or river stretches), the recent ratification of the WFD focus has shifted focus to viewing aquatic ecosystems more holistically (and interconnected), recognising aquatic ecosystems are nested within a terrestrial catchment. This view is clearly manifested in that catchment planning and management (aka River Basin Management Plans) are an intricate part of the WFD. Similarly, a CBD work program has been proposed to establish and maintain by 2010 a comprehensive, cost-effective and ecologically representative global system of networks of protected areas to reduce biological diversity loss at the international, regional, national and sub-national levels (UNEP/CBD/COP/7/4). It has also been recommended that by 2015 all protected areas should be integrated into ecological networks and relevant sectors so as to maintain, and restore where needed, the ecological integrity or connectivity (e.g. land-aquatic and aquatic-aquatic connections, buffer zones and corridors) which is a prerequisite for ecosystem structure and function.

One means of circumventing the problem of “different” ecosystems and to more straightforwardly address the new WFD and CBD initiatives is to recognise that ecosystem structure and function is related to inter-system connectivity (i.e. aquatic - aquatic and terrestrial – aquatic interactions and presumably to a lesser degree aquatic – terrestrial interactions). Although conceptual models exist regarding the interactions between aquatic “ecosystems” nested within their terrestrial landscapes, few studies have attempted to quantify the importance of ecological scale (e.g. Johnson et al., 2004) or connectivity for ecosystem structure and function and subsequently the importance of connectivity for ecosystem resilience to stress (both natural and human-induced). Functional/operational landscape units (conceptually) incorporate present-day knowledge of relations between different ecosystems. In brief, species diversity is often related to habitat diversity (e.g. Harper et al., 1997), and habitat diversity is considered as a function of habitat quality and landscape configuration. Early work showed that heterogeneous landscapes are more species rich than the individual habitat components (e.g. MacArthur, 1972). Building on this premise, both the species pool concept and metapopulation theory were developed where local species richness is regarded as a function of regional species richness (Hanski & Gilpin, 1991; Lawton, 1999). Ecosystem management should build on these principles, and place more focus on the use of functional/operational landscape units to better understand how present-day pressures such as habitat degradation, loss (and simplification) and fragmentation affect the services that these ecosystems provide (e.g. Opham et al., 2003). The use the River Basin approach proposed by the European Water Framework Directive is one (pragmatic) way of managing the integrity of aquatic ecosystems. Agreeing on a (similar) template for how we view the importance of scale and connectivity for terrestrial and aquatic ecosystems would be a major step towards better management of European biodiversity.

To adequately address present-day objectives regarding biodiversity management and conservation initiatives requires that more focus be placed on ecosystem connectivity, viewing systems as functional/operational landscape units as opposed to isolated entities (e.g. individual lakes, stream and wetlands). This approach amalgamates important factors of biodiversity such as the relation between habitat heterogeneity/diversity and species diversity. Moreover, this approach recognises the importance of ecosystem connectivity on ecosystem function and resilience to stress.

Key questions to be addressed:

- How is ecosystem biodiversity and function related to connectivity?
- How does habitat fragmentation or loss affect biodiversity, ecosystem function and ecosystem services?
- How does loss of connectivity affect dispersal?
- Are key species or species traits responsible for key aspects of ecosystem function?

Mixing land uses and farming systems, and lowered input levels critical for wildlife conservation in agricultural landscapes at multiple scales

Juha Helenius, Department of Applied Biology, University of Helsinki, Finland

SUMMARY: Drawing from conservation research at species and ecosystem levels, there is strong evidence for the importance to biological diversity in agricultural environment of maintaining and restoring landscape heterogeneity, diversity of agricultural land uses and farming systems, and lowering management intensity and chemical input levels, all at multiple spatial scales.

In European agriculture, there are several simultaneous trends of change, some towards more environmentally benign practices, some others towards the opposite. Common agricultural policy (CAP) gives much emphasis to the environment, including biodiversity. It forces agri-environmental schemes, which in many Union countries implement some measures to protect biodiversity. However, the major forces directing the development at large is economies of scale, and making benefit from the opportunities to externalize environmental costs in farming. This results in larger farms, more specialized farms, and more concentrated regions of production at the of cost of less favourable areas of agriculture. In the old EU countries environmental regulation has managed to reduce the rates of increase in chemical inputs such as mineral fertilizers and pesticides. However, the new member countries, such as Poland and the Baltic countries, are in the development path of intensification of input use in their agriculture.

These processes result in decreasing trends in biodiversity of agricultural landscapes, which happen at multiple scales and are caused by several simultaneous mechanisms. At the farm scale, specialization results in reduction of number of crop plant species grown, synchronized and uniform farming operations in the fields, less rotation and homogenization of soil management, including uniform practices to control nutrient levels and synchronized pesticide spraying schemes. Specialized crop farms enlarge shape and unify field parcels, and clear non-field habitat elements to get more arable area.

Increases of fertilizer inputs, be these mineral fertilizers or excess slurry or farmyard manure cause leaching of nutrients to non-crop habitats. Together with drifts of herbicide sprays, a lowered number of competitive, weedy plant species is favoured against less competitive species, including species at higher risk of local extinctions. Reduction in diversity of, especially dicot herbs results in reduced resource availability and diversity for herbivorous and predatory species, such as insects and farmland birds.

Mixed farms with livestock increasingly reduce the area available for the animals as pastures, replacing these with feedlot areas to ease management of the larger herds of fewer (usually just one) species of farm animals in fewer farms.

These specialization and intensification processes are obvious at the farm scale, but also at the landscape scale and at regional or even national scale: In any given agricultural landscape the odds are that the same line of specialization is favourable for most of the farmers. Regional and even national specialization trends result from the open market policy underlying CAP, which discourages or even prohibits (by excluding equitable subsidy) regionalized and local policies that would support diversified production.

In a recent Europe wide study, Billeter et al. provide evidence and give further references in how these processes correlate negatively with species richness of wildlife in agricultural environment (R. Billeter et al., Biodiversity, landscape structure and land-use intensity: general relationships for European agro-ecosystems. Manuscript submitted to Science, February 2005. - Billeter at Geobotanical Institute, ETH Zurich, Switzerland). It is alarming that even at global scale, the same economist views work against maintaining agricultural diversity. In the World Trade Organization (WTO), environment and biodiversity are not sufficiently included into the economic equations of agricultural trade arrangements.

Because of dependence on CAP and on the subsidy system, the appropriate level of addressing the problem in the EU is the policy level. At tactical (farm) level, agri-environmental programs encourage ecologically sounder practices, but this is not sufficient. The farmers do not have the economic or political power to correct the larger trend. At strategic (regional agricultural administrative) level, there is insufficient margin to adapt and adjust CAP to meet the local environmental needs, especially in relation to biodiversity concerns. Critically important is how the other operators in the food systems, including processing, transport, market, and citizen's choices, are adjusted to make the necessary space for adjustment in CAP an in farming. Evidence is accumulating (see Pretty et al., Farm costs and food miles: An assessment of the full cost of the UK weekly food basket, Food Policy, in press) in favour of localizing the food systems - and maintaining the agricultural diversity.

Understanding biodiversity: an issue of scale

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

The issue of scale in assessment is well illustrated by "The Blind Men and the Elephant". I know you know the story, but you might not know the poem:

It was six men of Indostan
To learning much inclined,
Who went to see the Elephant
(Though all of them were blind),
That each by observation
Might satisfy his mind

The First approached the Elephant,
And happening to fall
Against his broad and sturdy side,
At once began to bawl:
"God bless me! but the Elephant
Is very like a wall!"

The Second, feeling of the tusk,
Cried, "Ho! what have we here
So very round and smooth and sharp?
To me 'tis mighty clear
This wonder of an Elephant
Is very like a spear!"

The Third approached the animal,
And happening to take
The squirming trunk within his hands,
Thus boldly up and spake:
"I see," quoth he, "the Elephant
Is very like a snake!"

The Fourth reached out an eager hand,
And felt about the knee.
"What most this wondrous beast is like
Is very like a tree!"
" 'Tis clear enough the Elephant
Is mighty plain," quoth he;
The Fifth, who chanced to touch the ear,
Said: "E'en the blindest man
Can tell what this resembles most;
Deny the fact who can
This marvel of an Elephant
Is very like a fan!"

The Sixth no sooner had begun
About the beast to grope,
Than, seizing on the swinging tail
That fell within his scope,
"I see," quoth he, "the Elephant
Is very like a rope!"

And so these men of Indostan
Disputed loud and long,
Each in his own opinion
Exceeding stiff and strong,
Though each was partly in the right,
And all were in the wrong!

Moral:
So oft in theologic wars,
The disputants, I ween,
Rail on in utter ignorance
Of what each other mean,
And prate about an Elephant
Not one of them has seen!

John Godfrey Saxe (1816-1887)

Biological scales and conservation: Summary Week 2

Keith Hiscock, Marine Biological Association of the UK, Plymouth, UK, and **András Báldi**, Animal Ecology Research Group, HAS, Hungarian Natural History Museum, Session II Chairs

There have been several interesting developments in the Session II discussion this week although, from a marine perspective, I (Keith Hiscock) found many of the discussions in other sessions also relevant to biological scales and conservation. In the following, we add a few ideas.

Judit Padisák demonstrated how the short generation time for algal communities makes them ideal test objects for studying temporal scales but that spatial heterogeneity is only intuitively understood. However, it is desirable that the taxonomy of any test object should be certain. Jim Mallet argued that this is not the case, and the most widely used "species richness" as a metric in comparisons of biodiversity in different areas or different taxonomic or functional groups, has a fuzzy nature.

Juha Helenius showed that at tactical (farm) level, agri-environmental programs encourage ecologically sounder practices, but at larger spatial (and administrative) scales local policies that would support diversified production are discouraged.

The importance of habitat heterogeneity and its dependence on spatial scale was stressed by Alejandro Perazzo and András Báldi, complemented by Eduardas Budrys who introduced succession as a key player in habitat heterogeneity.

Martin Sharman listed eight Annex I 'marine' habitats – add 'sea caves' - and pointed out the lack of political agreement on mpa's at the time. I (Keith Hiscock) add now that the Habitats Directive was drafted between the outburst of interest in marine reserves for nature conservation in the early 70's and the renewed interest after the mid-90s. We simply were not ready in the late 1980's to suggest a meaningful suite of threatened marine habitats and species for the Habitats Directive and ended-up with a suite that was mainly of bird habitats (yes, the HD was the 'Non-birds Directive'). Keith Hiscock drew attention to the EUNIS classification that now gives us a much better 'scaled' classification of marine habitats (http://www.eunis.eu.int/index.jsp) than was available at the time the Habitats Directive was being assembled and so are in a strong position for revision.

Rainer Muessner (in the integration scales forum) suggested that the NATURA 2000 areas are probably the last 'boom' of protected areas that will be realised in the near future – I hope that he is being pessimistic as there is much to sort-out in the sea. Current imperatives from the World Summit on Sustainable Development and the OSPAR Commission need to be well thought-through and I hope that we will see improvements for marine conservation.

So, marine protected areas – at what scale? The questions posed by Richard Johnson with regard to 'connectivity' are important in the sea. The sea is supposed to have many less barriers to species spread than land – and yet 'islands' of habitats and species do exist. I (Keith Hiscock) am getting a lot of useful ideas from the various topics in the conference to feed-back into thinking about marine biodiversity scales and protected areas especially.

I (András Báldi) am from a country which has no sea at all, and therefore I have very little knowledge on marine conservation. However, reading the contributions it become clear that there is a huge lack and uncertainty in marine taxonomy, ecology and conservation compared to the terrestrial situation. What I am wondering is that why it is not recognised in the European conservation and ecologist community?

Guidelines for mapping insect biodiversity: a multi-scale approach

Guillem Chust, Departament d'Ecologia, Facultat de Biologia, Universitat de Barcelona; Barcelona, Spain.

SUMMARY: Biodiversity predictive mapping, which is a key tool for the natural-area manager, should focus on exploring multi-scale approaches, statistical modelling such as GAMs, remote sensing capabilities, and spatial modelling of beta-diversity.

Maps of biodiversity are a key tool for the natural-area manager. In regions where fauna is poorly surveyed, or for highly diverse taxa such as insects, biodiversity mapping by sampling over the entire area is not feasible. There is a need for predictive modelling to delimit the areas with valuable biotic components. Here, I present three research axes to improve spatial modelling of biodiversity: 1) to explore all components of the scale of landscape observation; 2) to intensify the use of remote sensing to characterise species' habitats; and 3) to use advanced statistical modelling and rigorous model validation.

Scales of a landscape: Contrary to environmental factors that define niche dimensions, habitat fragmentation affects population dispersal and, hence, is a scale-dependent factor. Individual mobility and cohesion of a local population are constrained by the surrounding landscape, which must be defined in relation to a given spatial scale (distance or area). In practice, the description of species' habitat should be approached taking as much scales as possible; particularly for insects since their "perception" of habitat patches can radically differ from a human-centred perspective. The scale of observation involves both grain and extent. In turn, the grain of landscape perception involves the spatial resolution and the notion of contrast. Few ecologists have dealt with the concept of contrast, which represents the difference across a boundary between adjacent patch types (Wiens et al. 1993). The consideration of these three components of scale has generated spatial models of collembolan and homopteran richness (Chust et al. 2003a,b, Chust et al. 2004).

Remote sensing: The use of satellite imagery for detailing the biophysical characteristics of species' habitat and predicting measures of local diversity is growing (Kerr and Ostrovsky 2003, Turner et al. 2003). However, the multispectral, multitemporal and multiscale capabilities are still poorly used for biodiversity mapping purposes. The most part of works in this area are restricted to use 1) a unique spatial resolution instead of combining different sensors, 2) binary land cover classifications (e.g. habitat – matrix) instead of exploring the real continuum such as that captured with vegetation indices (e.g. NDVI), 3) uni-temporal scenes instead of covering the phenological changes of vegetation, 4) optical imagery alone, while radar imagery is capable to extract arquitectural forest parameters and to avoid cloud interferences, 5) a limited number of methods to process images, neglecting, for instance, image segmentation and fuzzy classifications that allow a rich and detailed characterisation of landscapes.

Statistical modelling with GAMs: A variety of statistical methods for predictive habitat distribution and diversity patterns is growing (Guisan and Zimmermann 2000), e.g. multiple regression, ordination methods, neural networks, Bayesian models. Among them, Generalised additive models (GAM) (Hastie and Tibshirani 1990) join interesting characteristics for ecologists: to fit non-linear models, to extend the application of classical regression into other statistical distributions (e.g. binomial, Poisson, Gamma), and to estimate response curves with a non-parametric smoothing function that automatically identify appropriate transformations. Models should be validated with external data or resampling techniques, e.g. Jack-knife procedure (Guisan and Zimmermann 2000). Much conservation discussion has focused on areas of exceptional local diversity, as measured by high values of species richness, rarity, or of endemism (Williams et al. 1996; Myers et al. 2000), ignoring the overlap in species composition across sites. Beta-diversity is arguably more important in conservation planning, as suggested by efforts to optimise species protection with mathematical models (Cabeza and Moilanen 2001). There is a need for implementing spatial predictive models of beta-diversity to delimit priority conservation areas.

Practical application of scales for marine conservation

Keith Hiscock, Marine Biological Association of the UK, Plymouth, UK, Session II Chair

Discussions in this e-conference have explored areas of 'scale' that could be very important in guiding actions to protect marine biodiversity. So, let's get practical and let's influence.

On 24 March, I contribute to an 'expert workshop on marine protected areas' chaired by the UK Environment and Fisheries Minister. I will take with me ideas and my conclusions from this e-conference – including material yet to come from you by midday gmt on Wednesday!

Let me 'float' some ideas and ask you (even provoke you) to give everyone your views.

1. How much sea area do you think is needed for a 'representative network' of marine protected areas – 30% is often suggested. Well, is 30% just a 'magic number' (in which case, we might as well stick to '42' from the Hitchhikers Guide) or does 30% have some meaning in terms of ecosystem function? Are there any similar assessments on land that marine planners might learn from? My view is that a representative series of MPAs is bound to include at least 10% of inshore (say within 3 nautical miles of the coast) areas just to include a full range of types.

2. I will also ask again if any of you out there can tell me what constitutes a 'network' of protected areas. Does the term imply that some minimum distance is needed between examples of a habitat or populations of a species for them to interact and maintain genetic diversity or recruit? I do not think that the concept of something joined-up is needed in the sea. But, by all means, put me right.

3. On the question of scale, is there any reason to apply strict protection to more than the area covered/populated by a threatened or rare habitat/species? I believe that a larger area is often needed in enclosed waters to support the ecosystem processes that habitat or species relies on where sediment supply, freshwater input, turbidity and possibly larval recruitment regimes etc. are locally driven and contained. Therefore, whole lagoons, estuaries, sea lochs and enclosed bays should be the 'scale' chosen for protection.It is less easy to identify distinct bounded areas in the open sea and, here, I advocate strict (i.e. non-extractive) protection of the area occupied by the habitats or species of conservation interest supported by a much better 'duty-of-care' system for all of the marine environment than we have at the moment. I believe this because of the interconnectedness of the marine environment – larval recruitment either occurs from very local or potentially very distant sources (depending on life history characteristics of the species/component species of biotopes). Duty-of-care measures include technical measures for fishing gear and discharge consents that ensure minimal contaminant levels. Your views?

4. 'Marine Landscapes' are no more than re-packaged 'physiographic units' (the 'old' unit used in Britain to compare like-with-like at a large scale). Or are they? Does anyone have a different view to mine or can I be comfortable and politically correct in now talking about 'marine landscapes' and just re-cycle site selection procedures based on physiographic units?

5. I like the phrase used by James Mallet of using 'bite-sized' pieces of cake. He was referring to taxonomic levels but I believe that we will make more progress with marine conservation if we suggest protected areas that are understandable to people and that are not too large to be policed. So, pragmatism is important. What is a 'bite-sized' scale for size in marine ecosystems?

6. Marine conservation has a problem raised by several correspondents – our knowledge of what is where and how much of it there is is very incomplete. We cannot spot potentially rare, rich or productive habitats and rare species from the top of a hill, the window of a train or a helicopter [and remote acoustic techniques, whatever their supporters say, only really identify sticking-up bits of seabed at the moment]. Therefore, we look for surrogates, especially physiographic types, to identify potential MPAs. So, selecting examples – and preferably the best – of those different physiographic types in different biogeographical areas would be the way of coping with lack of precise knowledge about what is where. Can we

hope that such a coarse net will catch a full range of our marine habitats and species and, as we get more knowledge, refine the MPA series by adding sites and de-notifying sites?

7. Don't forget scales of time in this part of the e-conference. It may be that restoration is a possibility for some damaged habitats – although how long it will take we often do not know. We will watch Strangford Lough with interest as – in the lifetime of the Marine Nature Reserve and SAC there, the rich horse mussel beds have been destroyed by fishing and restoration is a desperate imperative to avoid infaction proceedings. It would be good to have some more correspondence about time scales.

Marine Protected Areas - scale depends on life histories and behaviours

Jean-Luc Solandt, Marine Conservation Society, Ross-on-Wye, UK

Regarding marine protected areas in time and space - the issue of scale is vitally important with MPA development.

In order to be practical, one must be clear as to identifying the key beneficiaries of the MPA itself prior to design and designation.

Considering the design of MPAs for UK whitefish stocks, planners must take into account spawning grounds, recruitment grounds and feeding grounds of often widely distributed species, and consider the minimal area needed to preserve and develop a viable population. I would think that the grounds needed to conserve cod stocks on an ecosystem scale (such as the north sea) would be a considerable portion of the sea area given (a) the wide ranging nature of the species, (b) the widely different geographical areas needed to carry out different stages of its life history, and (c) the current status of the stock given the recommendations from ICES. Whether or not this approaches or exceeds the 30% benchmark is up for fisheries scientists (ICES?) to recommend.

However, setting up practical MPAs for protection of inshore biodiversity (benthic attached species such as corals, algae, invertebrates) needs much smaller areas than that which is needed for migratory/mobile species protection. The realistic need for surrogate areas, and surrogate species and habitats (I would be more inclined to use habitats) can be very helpful in assisting in a process which is data deficient. Again, one has to look for the species/habitats that are threatened, investigate what is known of their life history patterns, particularly those governing reproduction and recruitment, otherwise, the area may be too small or even isolated from surrounding areas of adequate habitat for spill-over. For example, it was assumed before the 1990s that coral reef fish wouldn't necessarily recruit to areas near to where they were spawned, but much of the larval reef fish behavioural studies of the 1990s showed considerable site fidelity of recruits between generations - is this the same for temperate marine species?

How one develops an MPA network for migratory species such as whales, dolphins and basking sharks is more ambiguous. Perhaps we need to more carefully assess where the species are seen and when (which is currently done using surface sighting schemes), which is especially difficult with basking sharks, because obviously they aren't always at surface waters (usually only found in surface waters at fronts in the diel period). We have SACs in inshore waters for cetaceans in the UK, but perhaps we need to consider large areas of sea to be closed off to migratory species where and when they are known to be in areas which will coincide with anthropogenic impacts such as inshore set net and mobile fisheries, and recreational boat use.

Marine conservation using new technology

Robert Kenward, CEH Fellow & Technology Transfer Consultant, UK

In terms of the minimal levels of socio-economic organization required for marine conservation, local community may apply for shoreline and small boats, national territorial waters for boats that must return to port when storms are likely and international level for larger vessels. Conservation will require protected areas (of varying size as noted by Jean-Luc Solandt) at first stage. Policing of shoreline to territorial waters can be done at local-to-national levels. At international level, agreement would be required (i) for larger vessels to fit GPS recorders transmitting via satellite to a data centre and (ii) on penalties for transgressing reserves. This will minimize (remove?) cost of policing at sea. Permitted access routes may be required across large reserves. Reduced cost of GPS technology may eventually enable more sophisticated schemes to regulate licensed sustainable use outside protected areas (e.g. based on permitted duration in zones).

Qualitative biodiversity assessment for conservation purposes

Nikolay Sobolev, Biodiversity Conservation Center, Moscow, Russian Federation.

SUMMARY: Ecosystem-scale qualitative assessment of biodiversity may be based on the presence of several rare aboriginal species. Proposed method may be modified to the large-scale qualitative assessment of biodiversity naturalness by map analysis and remote sensing. The research in Russia shows the mentioned method useful for ecological network planning.

Regarding crucial contribution of ecosystem self-regulation to the global ecological stability, we need a method to assess biodiversity on the scale of a natural community as the most dynamic ecosystem component.

Potential (fundamental) ecological niches are somewhat wider than realised ones in a community of co-adapted aboriginal species. Due to this, such species are able to replace one another when their populations fluctuate in size. A community of co-adapted aboriginal species smoothes over disturbances and so stabilises environment. When we assess biodiversity as a factor of ecosystem ability to self-regulation, we must not consider total biodiversity richness, but deal at first with co-adapted aboriginal species (so called, Natural, or even Native, Biodiversity). Alien and synanthropic species should be taken away or considered as disturbing factor.

In order to ascertain ecosystem stability we must assess the presence of co-adapted species being in perpetual soft concurrence among them within a majority of various functional ecosystem blocks (as for example, sinusia, guilds, consortia, trophic levels). We can find in each block some species having environmental requirements higher than these ones of other species belonging to the same block. Because of this they are vulnerable to human impact and as usually disappear from the community before other species with similar but wider ecological positions. So we consider the presence of several rare aboriginal species filling in ecological niches through all range of biotic and abiotic conditions within ecosystem as the Ecosystem Qualitative Criterion of the Biodiversity Naturalness (Sobolev, 1992; Sobolev and oth., 1995). Such a criterion doesn't depend on the ecosystem origin, so it may be applied to the assessment of restored "natural" communities too.

Sometimes environmental conditions favourable for one or several similar rare species may accidentally spring up on the transformed areas where impacts of changed environmental characteristics compensate one another. So, the presence of only one or several similar rare species would not be enough to consider the state of the corresponding species community as close to natural. On the other hand, several rare aboriginal species belonging to the same large functional block (for example, a trophic level) of a disturbed natural community indicate that this particular functional block has no need to be restored. It should be carefully kept during restoration measures in order to further become a base of the natural community rebirth.

The size of habitat (or habitat complex) that species population needs is an essential species characteristic to be taken into account when assessing diversity of rare species on the community / ecosystem scale. For the purpose of such assessment we note several generalised size classes of areas that viable species population may need. The list of such Territorial Size Classes (TSC) with several but not comprehensive examples is the following:

1 - microbiotope / spatial mosaic patch within ecosystem (fungi, some herbs and invertebrates);

2 - group of spatial mosaic patches within ecosystem (shrubs, amphibians and reptiles, several dragonflies and butterflies);

3 - biotope / biocenose as usually identified by physiognomic characteristics (trees, small mammals and birds);

4 - group of similar biotopes (large herbivorous mammals, middle size birds and carnivorous mammals);

5 - natural tracts consisting of many various biotopes (large mammals and middle size raptor birds);

6 - natural tracts and its complexes of the eco-regional level (biggest mammals and raptor birds).

The species with TSC = 4, 5, or 6 are presented in natural communities of biocenose scale by local populations or even by only several individuals being a part of the viable metapopulations.

Our investigations in the Central Russian plain showed that territory inhabited by rare aboriginal species of TSC N obligatory includes habitats of rare aboriginal TSC (N-1) species. So we determined the index named "Level of Natural Biodiversity" (LNB) as a major TSC of the rare aboriginal species occurring in the investigated territory. The presence of rare species with TSC 6 or 5 is a General Qualitative Criterion (GQC) of the Biodiversity Naturalness. Such a criterion has been developed only for qualitative assessment of self-regulating potential of the investigated territory. For example, it may be useful for making a true choice between supporting and restoring strategies of the regional ecological network development.

The correlation between LNB and landscape characteristics allows large-scale qualitative assessment of biodiversity naturalness by map analysis and remote sensing. In the Central Russian plain GQC meeting natural areas should be of at least 12 th. hectares if being linked by semi-natural corridors. In the same region the smallest GQC meeting natural area topographically separated from other ones is of 39 th. hectares (Sobolev, 1998). These critical parameters seems to increase to the North and to decrease to the South.

An extensive range of GQC meeting natural territories is situated on the north and northeastern regions of European Russia, the Northern Ural, the north and central Siberia, and the Far East. It is known as the Great Euro-Asian Natural Backbone (Sobolev, Rousseau, 1998). Investigations of the Global Forest Watch Russia based on the remote sensing methodology of High Conservation Value Forest revelation up-dated in BCC showed heterogeneity of the Great Euro-Asian Natural Backbone and urgent necessity of the especial attention to it (Atlas..., 2002). Large-scale ecological corridors linking the biggest forest tracts and another forests in Europe should be established in order to ensure the Pan-European ecological integrity.

Biodiversity assessment in selected regions of the Russian plain showed several low LNB indexes as 3 or 4 for the belt of Broad-Lived Forests. More detailed observations show suboptimal state of the majority of broad-lived forest remnants. On the other hand, steppe vegetation wide spreads to the North only by riverbanks. We expect this to be a consequence of the landscape fragmentation and incoherent changes in climate and soil conditions. In result oak forests are replaced not by zone steppes but by birch boscages having a low LNB and unable to resist dispersion of invasive alien species. North-South and West-East ecological linkages should be improved to stop biodiversity loss in ecosystems of the Broad-Lived Forest Zone.

Considering spatial scales for amphibian conservation

Luz Boyero, School of Tropical Biology, James Cook University, Townsville, Australia & **Jaime Bosch**, Museo Nacional de Ciencias Naturales, CSIC, Madrid, Spain.

Issues of scale are a primary focus of ecological research (Wiens 1989). Apparent patterns of variation in biological populations and communities change with the spatial scale of observation, and the patterns themselves are produced by processes acting at multiple scales, not necessarily the same scales at which patterns are observed (Levin 1992). Knowing the scales of variation of communities allows the selection of the relevant scales for particular studies (Boyero 2003).

Understanding patterns in taxon richness at a variety of spatial scales is critical to prevent losses of biodiversity (Vinson & Hawkins 1998). Moreover, the choice of the appropriate scales for biodiversity assessments has important implications for practical conservation (Noss 1992), as it can help minimize costs and maximize efficiency in the management of natural populations.

Amphibian populations typically show a patchy distribution, related to their dependency on aquatic habitats, and often present a metapopulation structure (Alford & Richards 1999). Therefore, processes operating at medium scales determine local species assemblage composition and population size, so only studies that consider different spatial scales, from local to regional, will allow proper understanding of their distribution. Conversely, different studies at different spatial scales are likely to result in different conclusions, based on stochastic events, and thus will provide management strategies which are probably wrong.

An example of the multiscale approach is Bosch et al. (2004), who examined the patterns of spatial variation in an amphibian assemblage in a protected montane area in Central Spain. They suggested that amphibian conservation in the area should be focused on: 1) ensuring the preservation of maximum species richness in the two watersheds that compose the area (which are affected by different conservation problems); and 2) preserving pond types with characteristics that favour the presence of a maximum number of amphibian species and individuals, rather than preserving the maximum variability of pond characteristics.

Traditional management initiatives based on studies at small spatial scales have been focused on effective habitat "patches" – that is, breeding sites. However, it is becoming increasingly recognized that terrestrial habitats surrounding aquatic patches are extremely relevant to population health, since metapopulation dynamics requires recruitment processes among breeding sites. Unfortunately, most management programs have only taken into account the general rule that the wider a biological corridor, the more it facilitates amphibian movements. Nevertheless, the limited movement abilities of most amphibian species mean that most of their life cycle occurs within the corridor, requiring a series of generations to reach the next favourable patch (Beier & Lowe 1992). Therefore, successful amphibian corridors have supplemented suitable breeding habitats and such provision is a necessary inclusion at intermediate spatial scales in management programs. Unfortunately, most current conservation approaches consider the biology of a limited suite of taxa (mostly birds), ignoring for the potential role for amphibian conservation of very small water bodies, which often lack a conservation status. Therefore, conservation decisions today often involve the sacrifice of suitable amphibians habitats, even crowded breeding ponds. For this reason, studies at multiple spatial scales are needed not only to gain a broad understanding of the distribution of amphibian populations, but also to develop good conservation strategies. Moreover, they are likely to be especially helpful in designing cost-effective long-term strategies for the conservation of amphibian populations.

Scaling: One-parametric diversity index families and other issues

Béla Tóthmérész, Ecological Institute, Debrecen university, Debrecen, Hungary

SUMMARY: Many indices for measuring species diversity have been proposed. In this contribution, the importance of one-parametric diversity index families is stressed, which makes the Shannon index, the Simpson diversity and the Berger-Parker index of dominance special cases of a more general index. The general index includes a parameter, alpha, that can be interpreted from ecological point of view as a scale parameter. The importance of the commonness-and-rarity scaling is demonstrated by an example.

During the 60s A. Rényi studied the possibilities to develop a generalized entropy (diversity) measure, which includes as a special case the Shannon diversity; he presented his result at the 4th Berkeley Symposium on Mathematical Statistics and Probability. This is a one-parametric diversity index family, which offers a scale-dependent characterization of the diversity. The one-parametric diversity indices may be portrayed graphically by plotting diversities against a (scale) parameter. This curve is the diversity profile of the assemblage (Patil and Taillie 1979). Members of a one-parametric diversity index family have varying sensitivities to the rare and abundant species as the scale parameter changes. Besides the Shannon diversity, the Simpson diversity and the Berger-Parke index of dominance are also special cases of the Rényi diversity index family. There exists a large family of one-parametric diversity functions (see Tóthmérész 1995).

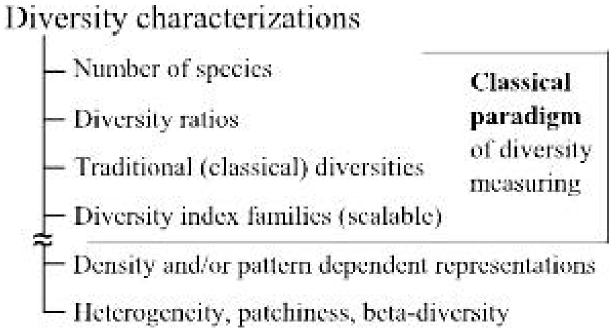
One may wish the index to be sensitive to dominant species but relatively indifferent to rare ones. This is possible with the one-parametric index families, since changing the scale parameter modifies the sensitivity of the diversity index. Evidently, the species richness is extremely sensitive to the rare species: detecting even only one individual of a species increases the number of species by 1. Just the opposite is the sensitivity of the Berger-Parker index of dominance: its value depends only on the dominance (relative frequency) of the most frequent species. These two traditional (classical) diversity indices are the starting and the end point of the scales of the Rényi diversity index family (details see in Tóthmérész (1998)). These methods are also discussed in the classical monograph on ecological methods (Southwood and Henderson 2000); they can be calculated e.g. in R using Oksanen (2004) package or other packages (Tóthmérész 2005).

The importance of the commonness-and-rarity scaling is demonstrated by the extremely simple example below. If you are ignorant of the scaling interpretation you may argue that anything is admissible using diversity. But this is not the case! The FAIR interpretation is as follows: C1 assemblage is more diverse in the frequent/dominant species, while C1 is more diverse in the rare species. The dominance structure of an assemblage may change considerably without changing the number of species. The re-arrangement of the structure may be explored using the scalable one-parametric diversity index families (Tóthmérész and Magura 2005).

Example. Two assemblages, C1 and C2 compared. C1 = (40, 30, 30) , and C2 = (60, 20, 10, 10) . The trichtomy of diversity for the assemblages C1 and C2 is demonstrated by the table below, which may suggest the false interpretation that anything is admissible using diversity.

	C1 C2
Number of species	3 < 4
Shannon diversity	1.0889 = 1.0889
Quadratic diversity	0.66 > 0.58

It would be extremely important to keep in mind the message of the above example during the interpretation of diversity studies. The scalable interpretation provided by the one-parametric diversity index families is a crucial step during the evolution of diversity measuring methods (for details see Tóthmérész 1998). I would like to stress just one more aspect. Each of the methods of the USUAL techniques of measuring diversity (in fact alpha diversity) is based on the following (usually implicit) assumption: (1) the studied assemblage is infinitely large, (2) the assemblage is Completely Spatially Random (CSR); i.e. (2a) the individuals occur randomly and (2a) the occurrence is independent of each other. These are rather strict assumptions, because they means the complete ignorance of the spatial patterns of the studied ecological (biodiversity) problem. Therefore, alike many other during the discussion, I would like to stress those techniques which are also take into account the spatial pattern and the hierarchy of the pattern during the study of biodiversity.



And the research questions are?

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

This conference has been running for some time now, and to me at least it has not always been clear what the key research issues are. Can you suggest one or more one-sentence questions that identify what must be known on the scaling issue if we are to slow or halt the loss of biodiversity?

Re: And the research questions are?

András Báldi, Animal Ecology Research Group, HAS, Hungarian Natural History Museum, Session II Chair

This session got several important contributions, and there are many proposed research problems - in reality there are too many. As a personal view, it is important that there is still a need to identify appropriate indicators of biodiversity change, e.g. in networks of natural ecosystems. This issue has been known, however, for a long time. A new and important aspect is that nature is non-linear, which means for us that as natural habitats are declining, we should expect sudden changes, catastrophes in biodiversity, while the habitat conversions are continuous. So an important question is to identify threshold values, and whether (1) local population catastrophes have drastic effects on larger spatial scales ("transfer of catastrophe across spatial scales"), and (2) what is the effect of time scale, that is what is the time delay in species extinction after habitat destruction (scale dependence of the extinction debt)?

Re: Re: And the research questions are?

Alejandro Rescia Perazzo, Departamento Ecología, Facultad Biología, UCM, Spain

I believe that an important but not unique question to maintain or to decrease the rate of loss of biodiversity may be: What type of diversity (i.ie., species diversity, ecosystem diversity, landscape diversity) must we manage to conserve biodiversity more efficiently? Furthermore, is functional diversity really an optimal measure?

Re: Re: Re: And the research questions are?

Felix Gugerli, Swiss Federal Research Institute WSL, Birmensdorf, Switzerland

.... and who cares about genetic diversity, the base line for population and species survival, in particular considering the on-going environmental changes to which organisms are exposed?

Political and economic scales in relation to biodiversity

Introduction to Session III: Political and economic scales in relation to biodiversity-

Sybille van den Hove and **Thomas Koetz**, Institute of Environmental Sciences and Technologies, Autonomous University of Barcelona and **Ekko van Ierland**, Wageningen University, Session III Chairs

The main direct and indirect causes of the present biodiversity crisis are anthropogenic, with human activities affecting marine and terrestrial biodiversity through a variety of interactions. Drivers of biodiversity loss include economic, political and social activities that function at different temporal and spatial scales. Such activities are characterized by the fact that they result from decisions taken at a particular economic, political or social level and occur at a certain moment in time, have possibly been functioning in the past for longer or shorter periods and might continue in the future for shorter or longer periods. The impacts of these activities also have their temporal and spatial scales. At the very local level the first direct impacts may occur, but through physical, chemical or ecological processes, the impacts may affect ecosystems and social systems at much larger scales, possibly at the global level for spatial aspects, and far away in the future for temporal scales.

Regarding responses, both the temporal and spatial scales of economic and political processes are extremely important. Biodiversity-related policies need to be based on a profound understanding of the spatial interactions in ecosystems, but also of human activities (what will happen where and what are the spatial aspects of competing economic claims on the use of biodiversity). In addition we need to understand spatial aspects of policymaking: which jurisdictions apply at the various spatial levels (ranging from local, to regional to global), how can they be coordinated in international policy making? For the temporal scale, we need to investigate how policymaking considers different time horizons (levels) (very short term, short term, medium term, long term) when dealing with issues such as the sustainable use of natural resources. Key issues are how decision makers can balance the short term pressure of direct economic gains, employment, or re-election, versus the long term needs of sustainable development and protective measures for maintaining healthy ecosystems.

The purpose of this session of the e-conference is to identify research needs with regard to Political and Economic scales in relation to biodiversity in order to ensure that research contributes to halting the loss of biodiversity. Results of the discussion, in particular recommendations for future research, will then feed in directly to the European Platform for Biodiversity Research Strategy (EPBRS) and BioPlatform meeting in Hungary on “Landscape Scale Biodiversity Assessment, the Problem of Scaling”, 31 March - 4 April 2005.

The session will be split into three main themes which will be addressed successively during the next two and a half weeks: (1) economic scales in relation to biodiversity (week 1); (2) political and policy scales in relation to biodiversity (week 2); and (3) integration of ecological, political and economic scales (week 3).

During this first week, we would like to discuss issues of scales in relation to (i) economic theory and methods, (ii) biodiversity valuation, (iii) integrated ecological-economic modelling and (iv) equity.

We look forward to your contributions on the items listed above as well as any other research issues and needs which you consider as important in relation to the topic of the session, and wish us all a fruitful and lively discussion.

Public-private cooperation. An experience in Sintra-Cascais (Portugal)

Jan Jansen, Radboud University Nijmegen, Institute for Bio Sciences, Nijmegen, The Netherlands

These days of withdrawing authorities there is an urge for viable private initiative in nature management. The challenge is to find sufficient common ground between the biodiversity sector and the corporate sectors entering into mutually rewarding partnerships. Good opportunities may lie at the interface of biodiversity and economic land-use sectors such as agriculture (green services) and the construction business. Is there a legal basis for contracts between private and public parties in nature management? Are there fiscal advantages and possibilities to be eligible for subsidies (Portugal or European Union)?

In Portugal we made an attempt to build a bridge between biodiversity interests and economic interests at the local scale. The initiative was based on common ground for operational partnership between nature managers (Parque Natural de Sintra-Cascais) and owners of the site. On one hand lack of financial means causes the management authorities to withdraw from nature areas, resulting in loss of biodiversity. On the other hand economic profit causes enterprises to invest in construction buildings along the coastline, often resulting in dramatic loss of biodiversity. The effects can be witnessed at large scale along the European coasts, including the Sintra-Cascais area. In fact this mechanism is a major reason why some areas became protected sites. In the past decades there were quite some conflicts between nature conservationists and real estate developers. Now, instead of opposing, the ecology sector and economy sector might find opportunities for partnerships between businesses and nature conservationists!

Withdrawal (= ‘doing nothing’) might be beneficial for natural habitats that undergo only natural forces like wind, solar radiation, salt spray, etc. However, there is an insidious process of invasive species undermining the strategy of ‘doing nothing’. Without active management certain habitats invaded by some aggressive species cannot sustain. Take for example sand dune areas. It is stressed that without active management, habitat quality of most Portuguese sand dune areas will diminish largely since they are often invaded by species like Acacia and Carpobrotus. At the local scale, dunes in Sintra-Cascais area include two major communities. The ‘grey’ dunes may be covered with the Armerio welwitschii-Crucianelletum maritimae. The ‘green’ dunes are stabilized dunes and partly covered by the Osyrio quadripartitae-Juniperetum turbinatae. The dunes are increasingly invaded by alien species, mainly Acacia longifolia and Carpobrotus edulis. They have been introduced to fix the dunes and are actually replacing a number of important, often endemic, plant species and thus causing a major ecological disaster. Plant species occurring in the study area, mentioned in Annex I of the Habitats Directive (HD) are: Coincya cintrana, Ionopsidium acaule, Limonium spp., Silene longicilia, and Verbascum litigiosum. In addition there are 10 Annex I biotopes.

It is expected that the nature value of the area will diminish largely if there is no management to restore the original situation. In general authorities cannot guarantee important nature values without the help of owners and citizens, since the public budget is too low to permit a reasonable management of all the valuable areas.

The idea of the project was to come to an agreement between the owners of a particular locality and the local management authorities to manage the area in such a way that both parties may have profit without too much destruction of the original character of the site. The owners want to invest in accommodation facilities. A part from their profit may be used to manage the area sufficiently according to the conditions of the authorities. In return the authorities may allow the owners to build under certain restrictions. Possible justification allowing constructing under restricted conditions comes from the effects of not allowing constructing. Not allowing constructing means continuation of abandonment and consequently final loss of nature values through replacement by aliens. In addition active management of the owners may restore the original vegetation already destroyed by the invasive species and conserve the existing high quality vegetation. It was recommended to follow an integral approach tackling both construction and nature management.

The government stopped the project. Probably because there was not a reliable juridical-political basis for it. Perhaps there was also a social problem. People often distrust real estate companies, thinking that nature is always sacrificed for economic interests. More public communication on this issue is needed. There is an increasing number of companies that understand that pushing too far they might kill the goose that lay the golden eggs. Would not a large scale economic-political-juridical framework, say national or even EU-broad, facilitate cooperation at the local scale that can deliver cost-effective 'made-to-measure' solutions to specific local variants of larger scale ecological problems?

Does anyone of the participants have good experience with similar public-private cooperation in nature management?

Economic value of biodiversity and drinking water

Loreto Rossi, Department of Genetics and Molecular Biology -ECOLOGY AREA, Rome, Italy

The economic value of biodiversity is generally difficult to assess. Important contributions have been written by Robert Costanza and others. However, a good of nature is, at present, easily to value financially (many million dollars per year); this good is drinking water. Often, drinking water is captured by rivers and lakes to be distributed without any heavy treatment (apart from chlorination) because the self-purification mechanisms work to maintain water poor in nutrients and other chemicals. Self-purification operates through many kinds of organisms and ecological processes. The high diversity (and role) of decomposer organisms must be studied to estimate their contribution to this environmental good.

Biodiversity valuation in the mountain context

Nakul Chettri, International Centre for Integrated Mountain Development, Kathmandu, Nepal

Mountains are among the most fragile and complex ecosystems of the world. They cover about 24% of the land surface of our planet, with diverse regions stretching from the Equator almost to both the poles. They are the center of major global biological resources and home to 12% of the global human population. Over a billion people depend on the mountains for goods and services such as water, food, forest products, and recreation. Additional billions of people benefit from other mountain services including the provision for energy and minerals, biodiversity-based goods and many environmental services. However, in the global developmental perspectives the mountains are the most challenging area with little or practically no development. It is evident that about 80% of the mountain people are depended on land-based activities for their subsistence living and the mountain lands are characterized by poor soil fertility, inaccessibility, fragility and heterogeneity in its use. The mountain specificities such as inaccessibility, fragility, marginality, socio-cultural diversity and lack of opportunities are causing serous degradation in the mountain environment. The resources in the mountain are declining mainly due to limited options, increased degree of desperation among the people to thrive and reduced level of flexibility going for alternatives. Such prevailing conditions are manifested by land degradation, declining crop yields, increasing food insecurity and gaps in demand and supply of biomass leading to environmental degradation. Thus, the communities living in these fragile and rich ecosystems are the poorest of the poor and marginalized. Therefore, there is a strong gap on understanding socio-economic and socio-culture aspect of biodiversity conservation for sustainable development in the mountains.

Strengthening ecological coherence and resilience is necessary for both biodiversity conservation and sustainable development in the mountains. The most pressing question in the present conservation paradigm is "for whom mountain people should conserve these biological resources at the cost of their livelihoods?" There have been ample discussions on the compensation mechanism for the environmental services and on the needs of highland-lowland linkages. However, we are yet to device concrete and satisfactory methodology do address these issue of compensation. Some efforts have been put in valuation of biodiversity and the environmental services at local as well as at the global levels. But there are limitations on the methodologies. In the valuation process contingent valuation, willingness to pay and cost benefit analysis were the few methods that were used so far. However, these methods have been able to address only the gross values of ecosystems, which were translated either in economic, or service values. The environmental services provided by micro-organisms in soil fertility and the aesthetic values ingrained in the culture and traditions of the mountain communities are difficult to assess in terms of money. Moreover, valuation of watershed services as an integrated system and more pressingly the services provided by mountains in climate change and carbon trades are still a distant dream of environmental economists. With these limitations, it is necessary to ask us that will valuation approach be able to address the conservations.

Technical progress in ecology and economy

Jurgen Tack, Belgian Biodiversity Platform, Instituut voor Natuurbehoud, Brussels, Belgium

Technological progress can reconcile economic growth and biodiversity conservation. However technical progress in ecology and economy is out of balance. The author gives two reasons for this: budget and scale.

The major challenges to biodiversity conservation are different between developed and third world countries. Where human population growth is the major challenge in third world countries, increasing per capita consumption is the major challenge in the developed nations. The synthesis of those two trends is economic growth: an increase in the production and consumption of goods and services. The main causes of biodiversity loss are at the same time the drivers behind economic growth. Governments of, in particular, developed countries always have argued (and are still arguing) that technological progress will reconcile economic growth and biodiversity conservation.

Technological progress refers to invention and innovation, two items of major importance for the knowledge driven economy where Europe wants to take the pole position. Everybody expects innovation (including innovative research) to result in technological progress. However, this is not always the case. In an economic world technological progress only occurs when more is produced with a given amount of resource input. This means technological progress is linked to rising productive efficiency. This could lead in the future to progress in biodiversity conservation.

Ecology and economy are closely related. The economic sectors have a trophic structure just like nature itself. In natural ecosystems you find producers (plants), consumers (herbivores, omnivores and carnivores) and service providers (detritivores, ecosystem services,...). In the human economy you find a similar structure: agriculture and extraction form the productive base, the consumers are the manufacturing sectors, and the service providers are banking, insurance and other providers. Technological progress is responsible for the dynamic of the human economy (from crop harvesting implements to computer technology). But did we not argue that technological progress will reconcile economic growth and biodiversity conservation? With all this technological progress, why does biodiversity continue to decline?

I see two reasons: budget and scale.

Technological progress is one of the most important qualities of the genus Homo for the past 4 million years. The species name of the present hominids (sapiens) refers in the first place to inventions and innovations. During the last centuries there was a shift from inventions and innovation made by individuals to an institutionalised process (institutes, universities, industrial laboratories, networks of excellence,...). Today scientists and engineers are responsible for most of the inventions and innovation. In our human economy they must be paid for doing this job. During the last decades programmes were developed to support research in different fields, generally called research and development (R&D). If you have a look at the R&D figures within the EU you see immediately the enormous gap between the R&D input of industry and the research funding of regional, national and international government bodies. More research money is spent to study 'salt' than to study biodiversity. So technical progress is stimulated particularly in the economic context. Urgent action should be taken to increase innovative research in the field of environmental problems, particularly biodiversity related research, to balance technological progress between ecology and economy.

Scale is a second problem to tackle. While the human economy created an ecosystem with world wide rules, laws, links and interactions we are still not capable of describing in an adequate way even one tiny local ecosystem. Comparing ecosystems is almost impossible and thinking of global laws for all existing ecosystems is a priori answering an almost impossible question. Out of the world wide biodiversity research budget most money is still spent on species-related questions. However to halt the loss of biodiversity we urgently need innovative research on a much larger scale, just like economists are doing research. To do this we can learn a lot from economic research. However, I do not think ecological science should become part of the economic science or copy it, but should be in balance with it and interact with it. Not biodiversity valuation but biodiversity validation will halt the loss of biodiversity.

Re: Technical progress in ecology and economy

Jan Dick, Centre for Ecology and Hydrology, Edinburgh, Scotland, UK

I have just read with great interest the contribution of Dr. Jurgen Tack, Belgian Biodiversity Platform, entitled Technical progress in ecology and economy.

He argues that a primary threat to global biodiversity is 'economic growth: an increase in the production and consumption of goods and services': yet appears not to favour an approach to value biodiversity and its services directly i.e. create a tradable 'ecosystem economy' where all levels of society pay either directly or indirectly to maintain or improve an ecosystem if they use either a product or service of that ecosystem.

I am at a lost to fully understand Dr. Tack's final remarks: Not biodiversity valuation but biodiversity validation will halt the loss of biodiversity.

Not biodiversity valuation but biodiversity validation will halt the loss of biodiversity

Jurgen Tack, Belgian Biodiversity Platform, Instituut voor Natuurbehoud, Brussels, Belgium

The difference I make between valuation and validation is:
* Valuation: the process to estimate or to determine the market value of a thing, in this specific case biodiversity
* Validation: the process to make something at once relevant and meaningful

In reality valuation will put a price on each aspect of biodiversity or on the ecosystem services it provides. I argue we are not capable of doing this with the present knowledge we have in ecology and economics. The ecology of species and ecosystems is so difficult we are not capable (and will not be capable in the near future) to reverse the loss of biodiversity because we do not even understand how the present environmental degradation impacts biodiversity in all its aspects. As long as we are not able to influence the processes behind this loss we will not be able to value biodiversity or aspects of biodiversity.

Valuation of biodiversity probably requires a whole new ecological and economics language which is not mathematical (not in the way we know mathematics today).

However, to validate biodiversity we have tools available (e.g. communication); but still 95% of the people do not know, do not understand the importance of biodiversity, the role biodiversity plays in ecosystem services, or worse do not even know what biodiversity is.

Valuation as part of Validation

Jan Dick, Centre for Ecology and Hydrology, Edinburgh, Scotland, UK

I must thank Jurgen Tack for his clear definition. I agree with his argument that we are not in a position to value ecosystems in monetary terms at the moment. His ideas are echoed to some extent by Nakul Chettri in terms of mountain landscapes - will valuation address the problems of conservation?

Clearly money is not the only criteria which directs anthropogenic causes of biodiversity loss and money is not the only factor which will halt biodiversity loss at the local, national or global scale - but it is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change.

I would argue that it is vital to embrace the valuation of landscapes as one (and I stress only one) mechanism in the process of biodiversity validation as defined by Jurgen i.e. process to make biodiversity at once relevant and meaningful. I would further argue that a holistic approach must be adopted where the whole ecosystem or landscape is valued taking account of all the very many socio-ecological elements, rather than the narrow focussed approach which was common during the last century (e.g. agricultural polices single focus on productivity which was then changed to environmental services rather than widened to encompass the many facets and uses of agricultural landscapes).

What is the scientific community offering as an alternative mechanism to monetary valuation?

Monetary evaluation is not scale -independent

Felix Rauschmayer, UFZ Leipzig-Halle, Germany

Jan Dick stated that money "is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change."

Which scales does she refer to? Certainly not the time scale, as Clive Spash made it clear that monetary valuation is dependent on the time scale chosen and usually myopic. Certainly not the social scale, as services and goods produced within a household rarely are valuated with money, and the monetisation increases with the economic scale addressed: non-monetised services might be common with the local butcher, but not in the globalised cash and carry market. The spatial scale is usually correlated with the social scale (quite loosely in industrialised countries, more strongly for the largest part of the world).

Valuing whole ecosystems is a methodologically impossible endeavour, as the discussion after the Costanza articles showed. Monetising ecosystem services is possible, it might be politically useful (as in the example of Hein and van Ierland - could you give the reference please?), but what does it show? These exercises show that people pay money (at least: claim to do so) for maintaining or improving certain ecosystem services. They cannot show the value of ecosystems (as we don't know the whole value), and they reduce the decision context to the context we are used to in the economy. There are several reasons for not wanting this context in decision making on nature conservation (the myopics of monetary evaluation is one of them).

Consequently, the answer to the question, Jan Dick gives at the end of her last communication: "What is the scientific community offering as an alternative mechanism to monetary valuation?" depends on the context which the society chooses appropriate in the specific context. And this context is likely to include answers to the questions on how to address the irreversibility of ecosystem destruction, of the regional, national, perhaps global effects of the local action, etc. by using advocates specific scenario techniques. Knowledge on the effects of possible actions on ecosystem functions and services, and on the induced change in well-being of humans and non-humans should be a part of these decision processes which mostly will be participatory.

You may find an evaluation of different participatory and multicriteria processes in environmental conflicts (not specific to biodiversity, though), in Rauschmayer and Wittmer 2004. These processes are not as theoretically elegant as monetary evaluation, and there is no best approach in all contexts, but they are part of a tool-box with tools for different circumstances.

Re: Monetary evaluation is not scale-independent

Jan Dick, Centre for Ecology and Hydrology, Edinburgh, Scotland, UK

The scale I refer to is relevant to the 'mechanism of money' not the unit value which I completely agree varies temporally and spatially. The mechanism of money i.e. exchange of some tradable unit for some product or service is I would still argue "a scale independent, internationally recognised mechanism for directing and managing anthropogenic change."- or am I missing something?

Money, money, money

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

I very much liked Dr Rauschmayer's comments "Monetary evaluation is not scale-independent". I had had the same reaction as Felix to reading Dr Dick's remark about the scale independence of money, but Felix expressed my reaction much more elegantly than I could have.

I also liked his remark about valuing whole ecosystems. In general I find the idea of placing value on biodiversity somewhat ridiculous - in the sense that it makes me want to laugh, rather nervously. I admit I am not an economist, and I am uncomfortable with the idea that we can usefully give everything a sensible non-ambiguous value.

Let's try a thought experiment. You are standing in the middle of a frozen lake whose depth you do not know. The ice is not thick, but for the moment it is bearing your weight, even though you think you feel it creaking. One of your close relatives is watching you from the shore, as is the director of your life insurance company, a cameraman from the local TV company, and a neighbour whom you have been blackmailing. How much is the ice under your feet worth to the collectivity of stakeholders? To you the worth is infinite - if you go through, you will probably drown. To the cameraman, having the ice fail could mean such good footage for the evening news that he is already dreaming of promotion.

I suggest that this is exactly analogous to the fisherman out on the ocean in his trawler, with an equivalent range of stakeholders waiting for him on shore. Only in this case it is not the ice that is creaking, but the ecosystem that supplies the fish on which his livelihood depends. How much is that ecosystem worth? I think that the answer is always going to be: it depends. It depends on which stakeholder you talk to, and it depends on the scale, in time, in space, in the size of the fishing fleet and the size (and conservation status and trend - another scale-related set of issues) of the population on which the fisherman preys.

How Much is that Ecosystem in the Window?

Clive Spash, Socio-Economic Research Programme (SERP), The Macaulay Institute and Department of Geography & Environment, University of Aberdeen, UK

Is valuation in terms of an exchange price for biodiversity necessary for its preservation and conservation? Is money an objective neutral universal mechanism for changing human behaviour? I would like to try to address some of what I think may be underlying the disagreements over the role and meaning of the "mechanism of money".

The concept of money is very contextual across time and space. Consider some instances of things used as money (i.e. units to aid exchange) these include cigarettes (common in prisons), shells, large stones which could not be moved, small bits of metal, little bits of paper and now electronic signals. Economists regard successful money as having certain characteristics, such as limited and controlled supply, ease of transfer, low or no value in itself, social acceptance and trust. Although, many historically common forms of money have also violated economic assumptions of "good" characteristics.

The nature of money has changed rather dramatically over time and space. There have been human societies based for long periods on systems without money (as we understand it) e.g. the barter economy. The old colloquialism "money is the root of all evil" has a foundation in remembrance of times and places where money was absent from such a dominant role in human society even as a means of exchange. Even today we run many of our most valued affairs without money as the means of management or measure of value. This is not to deny the essential usefulness of monetary systems. Money takes on central importance with scale of transactions and as the complexity of transactions grows across time and space. For example, the Romans used money extensively because they had an empire to run, but without that empire the role and use of money changed. People use local exchange and barter at small scales. Even large social systems can be run without money or market e.g. feudalism, central planning, command and control.

Money is an instrument of political economy reflecting power and control, and it does change with scale. Does a money system with 1, 2 or 10 people have the same meaning as one with 1 million, 2 million or 10 million people? Presumably those starting local exchange trading systems (LETS) believe small scale monetary systems are fundamentally different from those of the global market. How about a system with one billion people? Is the Euro fundamentally different from the currencies it replaced? If not then why bother?

Modern banking and forms of money are distinctly different from the bits of metal which had value in themselves. The form and nature of money has changed fundamentally even in the last sixty years e.g. the move away from the gold standard by which all paper could be converted to gold on demand. The scale of transactions, international politics and power have a lot to do with the changing nature and character of money. There is no mistake in the fact that the International Monetary Fund (IMF) and the World Bank were both located in/controlled by the USA.

Money changes with the scale of transactions and has had a fundamental influence on human perception of "value". This later point is important because of the assumption that using monetary valuation of biodiversity can only help. There is something contradictory in calculating a trade price for something you do not wish to trade. The problem with a focus on monetary valuation is the failure to recognise its limits, content and meaning. For example, you all earn an annual income which is the trade price for your labour, but few would presumably claim the value of their life is merely annual income times life expectancy! Yet this is what is being suggested for life support systems.

So, to return to the challenge posed by conference contributors, what then are the alternatives? Well other social and political phenomenon can be substituted into the sentence discussed as justifying "money mechanisms". The following examples make as much and perhaps more sense: Revolution is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change; Religion is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change; War is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change; Law is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change; Justice is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change; Morality is a scale independent, internationally recognised mechanism for directing and managing anthropogenic change.

Humans use many alternatives for (mis)managing their affairs and many are far from desirable or require bounding and control by others. What can scientists do in terms of offering alternatives? Well scientists can offer a lot more than one perspective on the value of biodiversity and the ways in which to approach its management. They can also warn of the pitfalls in plumping for certain approaches as opposed to others.

The biggest challenge is to get the institutional and social systems which will improve a situation which is identified as being bad and getting worse. This means identifying how we got here in the first place. The problem with a narrow valuation approach is the way in which it can perpetuate the dominance of social systems and institutions which many regard as a cause of the problem rather than a means to its solution

Re: How Much is that Ecosystem in the Window?

Silva Marzetti, DSE, University of Bologna, Italy

The preservation of coastal areas is an essential task of public authorities. The natural and near-natural characteristics of these areas at tourist sites attract numerous visitors and make major contributions to local economies. From the recreational point of view, investing in a natural area is successful and a sustainable coastal planning requires defence projects to be selected also in order to preserve biodiversity. This is the result of a comparison between the recreational value of a natural beach area and the recreational value of a nearby developed beach area in the Italian coastal site of Lido di Dante near Ravenna. (Here for developed beach area I mean a sandy beach strip behind which dunes and pinewood are destroyed in order to build tourist facilities; while for preserved or natural beach area I mean a beach strip behind which dunes and pinewood are conserved.)

Within the EU DELOS (2000-03) framework, in this site a survey by questionnaire was carried out in 2002 in order to provide data useful for the decision-making process about the protection of the natural beach. It consists of the application of the Contingent Valuation Method for assessing the non-marketable recreational use (such as sunbathing, walking and swimming) of the Lido di Dante beach areas in the present state and in hypothetical scenarios of erosion and artificial defence. The daily use value in Euros of the Lido di Dante natural beach is higher than that of the developed and semi-developed beaches where biodiversity is to a great extent sacrificed to tourism growth. In particular, the recreational value is considerably smaller in the situation of erosion with respect to the present state use value. According to the majority of respondents, the natural beach should be defended from erosion, confirming the great value that visitors generally assign to 'sun and sea' recreational activities.

Why money?

Felix Rauschmayer, UFZ Leipzig-Halle, Germany

I am not astonished by the results of your study, but why are these results an argument for making monetary evaluations (not of biodiversity, but of recreational activities)? You could have reached similar results by making a non-monetary survey, I suppose.

Using money gives the impression that everything is for sale and can be traded (off). Monetary values are supposed to make a strong impression on decision makers (this is why so many of our natural science colleagues use the studies from Costanza et al. on the ecosystem value, Pimental et al. on costs of biological invasions etc). What are the social and moral effects of using monetary values of something which can be sold (cutting the wood = the habitat of one species), but which should not be sold. Environmental Economists often claim that monetary values of nature are convincing authorities or the public to preserve nature - is this true? Does anyone know of studies on this? Philosophers (and among them Ecological Economists) claim that using monetary values may lead to crowding out of moral arguments for nature preservation. Is this so? Are there studies on this?

I know that I only address the practical, instrumental side of the issue: Is monetary valuation of nature useful or not? I did not address the normative or substantial side of the issue: Is it right or not? On this latter question, a lot of arguments have been exchanged.

Coming back to your study, Silva: What did it change that you measured the preference for a “natural beach” via the monetisation of recreational activities? What have been the real impacts and what could they have been with another type of assessment?

Re: Why money?

Silva Marzetti, DSE, University of Bologna, Italy

As regards using money or not, Why not? Money is an useful tool for estimating values, and the recreational use of a natural beach is a value that can be ascribed to it. What is the total value of a natural resource? Environmental economists make reference to the total economic value of a natural resource, which is: use value + existence value + option value + bequest value. All these components can be estimated in monetary terms because they make reference to the individual preferences. In addition, the Primary value is recognised but it cannot be valuated in monetary terms.

From the practical point of view, it seems to me that your doubt is: Is the cost-benefit analysis (CBA) useful in policy-making? My reply is yes.

According to the IUCN (1998, p.3), ‘a protected area is ... managed through legal or other effective means’ such as public funds. Nevertheless, public funds are limited (the crucial thing). In other terms, investing in natural areas competes with other alternative public investments (such as hospitals, schools, and so on), and other alternative uses of the same area. In particular, according to my experience, a lot of ordinary people (who pay taxes) believe that it is right not to conserve, because they prefer present economic benefits (for example, they prefer not to preserve dunes and to build a tourist village). Therefore the fact that an investment in protected areas may provide significant sustainable economic benefits has to be proved (the word ‘economic’ here is intended in a wide sense because it includes also non-market benefits, such as bequest and existence values).

The cost-benefit analysis (CBA) has the task of showing whether an investment project will have a net social benefit, or will increase the social welfare. It is also designed to show which of the competing projects has the highest net benefit and should be implemented. The CBA requires all the benefits and costs ascribed to a project to be expressed in monetary terms. Nevertheless, not all the benefits and costs of a protected area have a market price. In particular, if benefits such as existence value, bequest value and non-market use value are not evaluated (with specific economic methods such as the contingent valuation method) they cannot be considered in the CBA; the economic value of the area is underestimated and the project may not win the selection. It seems to me that this is a very good reason for applying the CBA, even if the total value of a natural resource is underestimated because the primary value cannot be evaluated and so included in the computation.

From the philosophical point of view, I cannot reply because I am not a philosopher.

As regards my specific research, the visitors of the natural area enjoy the sun and sea activities because biodiversity is the major characteristic of the area. They do not visit the developed beach. So, because biodiversity makes their enjoyment higher than that obtained by visitors in the developed beach, It seems to me that this difference can be considered one of the biodiversity values of the area.

What do you exactly mean when you write ‘another type of assessment’?

Terrestrial conservation using new technologies

Robert Kenward, CEH Fellow & Technology Transfer Consultant, UK

Protection of species and reserves has been invaluable for preserving biodiversity against rapid development, but has not prevented huge biodiversity loss. Loss will continue locally as land-use changes, but much work now shows restoration to be practical and there is huge scope for it. The EU’s 6th Environmental Action Programme 6 puts “restoring and developing the functioning of natural systems” in Article 1, before the “decoupling” and “2010” objectives. CBD Article 10 (Sustainable Use of Components of Biological Diversity), requires that Parties “(d) Support local populations to develop and implement remedial action in degraded areas where biological diversity has been reduced”.

Of 19 substantive articles in CBD, 13 mention sustainable use, compared with 2 articles (one for definition) on protection (i.e. Protected Areas). E-Conference Participants might also like to look at the new CBD Principles and Guidelines for Sustainable Use (www.biodiv.org), which address governance, adaptive management, scale (especially local) and other issues. These Principles and Guidelines reflect the approaches favoured by Erling Berge, Nakul Chettri, Rainer Muessner, Jouni Paavola, Frank Waetzold and others. Conservation through sustainable use gives a second pillar for conservation, and a challenge: how much can we now restore outside reserves?

I hesitate to mention money, but it is at least a convenient measure of pressure on land-use outside reserves (and can calibrate to other social measures like voluntary time). Moreover, as well as land-use values from production and ecological services, much is paid for recreational use of wild resources (at least US$81 billion in USA in 2001). A challenge for socio-economics is how to tap all potential land-use values (social and economic) to conserve biodiversity in just ways that attract social support.

We lack a great deal of knowledge on subjects like monitoring, indicators, scale effects. However, the summarising of Jose Garcia del Barrio is helpful. Area: minimal socio-political units (municipalities) are convenient for integrating socio-economic capabilities, but nothing should preclude finer scales (farm, garden) later. Biota: coverage at municipality level and as disaggregated as possible (Allan Watt) will need volunteers, who probably can only do richness (although detection frequency in repeat sampling may also give indices of density); genetic diversity of wild species must be further researched before application techniques are developed.

As 2010 is approaching and quite a lot of research has been done on restoring biodiversity, perhaps CBD’s Principles and Guidelines can help a start from “fire-watching” towards “fire-fighting”. Thus CBD now promotes adaptive management as a way forward where data are imprecise, with monitoring and research to refine that management. Combination of three new technologies provides a possible approach:

1) Develop extensive monitoring, capable of providing as many indices (indicators) and in as much detail as possible, making maximum use of voluntary contributions to reduce cost, on (a) biodiversity (b) land-use and (c) land-use socio-economics; use the internet to coordinate and GPS for scalable vector mapping with aggregation in GIS;
2) Internet collation of research on actions that restore biodiversity at minimal cost;
3) Use of CAP, pay-to-use and voluntary resources to motivate restoration;
4) Review of existing EU legislation to remove perverse socio-economic incentives;
5) Integrate (1)-(4) (i.e. biodiversity data, land-use data, useful rules and useful procedures) to start delivering decision support by internet to local level, to guide the myriad decisions on land-use that affect landscapes and their biodiversity;

6) Use data-association from (1), modelling from (2) and experiments funded by (3) to refine cost-effective biodiversity restoration and inform policy in adaptive management cycles.

As Jan Jansen notes, integration needs to be at EU level. Central guidance at European-to-Provincial scale could use wetware (humans). European to Municipality (and even to farm-garden scale) is practical with automated IT. Revision of Directives on Birds, and on Habitats (through a Biodiversity Directive?) might be needed to address concerns of Susan Baker and Horst Korn, perhaps giving greater flexibility of protection measures (Cascade Protection, Zoning) to optimize conservation through protection where species/habitats are rare and other uses where they are common.

Pan-European technology integration for conservation is ambitious and would require Commission commitment, but could (i) benefit research (standardized comparisons, large-scale experiments, rapid plug-in of results), (ii) underpin a European knowledge-economy in the environment and (iii) improve social cohesion by education, central-local communication and local cooperation initiatives. Perhaps such an approach to restoring biodiversity outside reserves would be a good alternative to the black-boxing (eco-apartheid) about which Chimere Diaw warns.

Bridging the scales between ecological, economic and political systems

Martijn van der Heide, Agricultural Economics Research Institute, The Hague, the Netherlands

Biodiversity and ecosystem services are the pillars on which humanity builds civilizations. The multifaceted concept of biodiversity is intimately linked to political and economic scales. Or, to put it differently, the way to approach biodiversity is amenable to political, economic, cultural and discursive forces, and their mutual constitution in the context of conservation debates and practices. Political and economic elites are keen to benefit from the widespread environmental anxiety and curiosity. Moreover, policy-makers influence the management of ecosystems by directing goals and agendas in ways that require environmental management decisions to be based on much more than ecological knowledge. Causes of biodiversity loss are multiple and can be divided into proximate causes and underlying causes. The proximate causes, such as over-exploitation of species and land-use changes, are partly within the domain of natural sciences and partly within the domain of social sciences. The underlying causes, such as pressure of human population growth and the structure of property rights, are largely within the domain of the social sciences. Unfortunately, underlying causes are not clearly identifiable and, therefore, subject to debate.

Proper management of biodiversity requires policy measures at all levels, ranging from local to global. In fact, 'think globally, act locally' is a true measure of how biodiversity conservation must take place. However, in economic terms, people living in or near a protected ecosystem often capture little benefit from preservation or sustainable resource use. The benefits of biodiversity protection increase with the scale from local to regional to national to global. In contrast, the economic costs incurred as a result of biodiversity protection prescriptions follow an opposite trend. The heaviest burden tends to be borne by people situated in rural areas, in the vicinity of protected areas.

As biodiversity and ecosystem services are crucial for the livelihoods of many (poor) people globally, there is a need to manage, rather than just conserve, biodiversity to promote economic growth and improve livelihoods of the poor. In other words, biodiversity policy actions should form a win-win strategy. However, a sustainable exploitation of biodiversity is hampered by the fact that many natural assets, such as species and ecosystems, are characterized by the absence of fully defined property rights. Many of these assets are considered to be public goods, or possess some features associated with such goods. Because it is impossible, or at least very costly, to deny access to a natural asset, markets fail to allocate biodiversity resources with public goods characteristics efficiently. This may be understood by noting that prices do not then signal the true scarcity of the asset. In response to these markets failures, the assignment of property rights as a form of biodiversity policy has often been suggested. It is important to realize, however, that government intervention may not always be a magic cure-all in correcting markets failures. With many competing social and economic objectives to be satisfied (such as employment, agriculture and economic growth), and many political issues to deal with (funding, incentives, willingness to participate, co-operative governance, institutional capacity etcetera), there are nearly always intervention failures, inefficient or uncoordinated regulations, policies and institutions that exacerbate adverse impacts on biodiversity. Decentralization and self-organization may then be reasonable strategies of public action.

Issues of scale in integrated ecological-economic modelling for biodiversity conservation

Frank Wätzold, UFZ Leipzig-Halle, Germany

SUMMARY: Problems related to spatial scales that arise in integrated ecological-economic research (with a particular focus on modelling) are discussed and a possible solution proposed.

Given that many biodiversity management problems have an economic and ecological dimension there is a strong case for integrated ecological and economic research in general and, given that both disciplines frequently apply models there is also a strong case for ecological-economic modelling. However, the issue of "space" seems to be one of the most prominent areas where misunderstandings and problems for research that integrates knowledge from ecology and economics are likely to happen.

One important difference between the two disciplines is probably that the issue of space is comparatively more important in ecology than in economics. However, if economists explicitly take into account space the spatial scales considered by the two disciplines may be different (Holub 1999). For ecologists, the spatial scales depend on the species they are looking at and range from cm2 to global scales. For economists, space only matters if economic parameters (e.g. costs, prices, benefits, institutions) important for the question under review spatially differ. However, they may differ on other scales than the relevant ecological parameters (e.g. economic parameters would hardly differ on a scale of cm2) which makes integration difficult.

An analysis of differences and similarities of ecological and economic models by Drechsler et al. (2005a) supports the anecdotal evidence reported above. Drechsler et al. analysed 60 models (half of them taken from ecological and half of them taken from economic journals) and found that one important aspect where models from the two disciplines differ is the consideration of space. First, only 25% of the economic models analysed explicitly considered space whereas nearly half of the ecological models did. Furthermore, a significant share of the ecological models took into account space with explicit reference to landscape co-ordinates (e.g. distance between neighbouring patches) whereas the economic models only took into account space in an abstract manner (e.g. by assuming two abstract regions).

Given the differences in spatial issues in ecological and economic research in general and modelling in particular, how can knowledge from the two disciplines be usefully integrated?

A good approach is probably not to discuss the problem of space in an abstract manner but rather start from a particular conservation problem. Let us consider as an example the aim of developing cost-effective compensation payments for conservation measures for an endangered butterfly species (Maculinea teleius) protected by the EU Habitats Directive. The appropriate conservation measure is mowing meadows at a particular date for which a certain conservation budget is available for a particular region to compensate farmers for conservation costs. This problem structure determines the spatial scale for both the economic and the ecological research. First, the costs and the ecological effects of butterfly friendly mowing have to be determined for each meadow in the region, and, second, costs as well as ecological effects have to be aggregated on the regional level (see Drechsler et al. 2005b for details).

Issues of scale in the valuation of biodiversity

Clive Spash, Geography & Environment, University of Aberdeen, and Socio-Economic Research Programme Macaulay Institute, UK

SUMMARY: Scale issues in economic valuation arise across time and space. The basis of standard economic valuation is preference utilitarianism, which raises a set of scale valuation issues that are distinct from other ways in which value can be expressed. I focus here on the economic approach.

The time issue is the most commonly acknowledged and leads to discounting (a solution for some and a problem for others). The basic conundrum is that a small value over a long time becomes very large (at the extreme infinite) which can swamp a large value now that lasts a short time. So the measure of species value is then longevity. If the concern is ecosystems or their functions and all are expected to last forever (or at least ideally so) then all are of equal value. The economic answer is to use market interest rates to reduce future values asymptotically towards zero. An alternative would be zero discounting with a set cut-off date. Neither is a particularly satisfactory way in which to value the diversity of life on the planet. Discounting also ignores the rights of future generations (for more on this see Spash 2002).

Problems with respect to space or distance have been given little attention. In economics these arise mostly when discussing income inequities leading to valuation problems. An example is the value of a species such as say a tiger to locals versus those in Europe or the North America; or a real example which causes much consternation is the value of human life appearing much higher in industrially developed economies. Basically once the standard economic approach is adopted the power of money is the judge of value. There are other aspects to geographical scale, which work two ways: one is the interest of those globally in locally unimportant biodiversity and the other is locally important biodiversity that nobody else cares about. These issues can result in such things as a biodiversity poor park in a city appearing to have a far greater economic value than a pristine old growth forest hundreds of miles from any sizable population. A personal experience was being asked to ignore locals with respect to marine biodiversity valuation because tourists had higher income and were expected to pay more (in fact the locals had a higher mean willingness to pay despite the income difference).

A psychological problem for economic valuation based upon preferences is that individuals tend to focus on the immediate and in terms of species those higher in the food chain. The focus of economic valuation studies has then, unsurprisingly, been upon key species, and so far has rarely address species diversity, and hardly ever ecosystems and never genetic diversity. A survey on the conservation value people place on the Panda is more likely to get responses as opposed to one on the value of bamboo; even though they are part of the same ecosystem. Once soil micro biodiversity is considered the preferences of the general public seem to have little relevance. Of course they may have little relevance to most environmental values (for more in this vein see O'Neill 1993). However, the interest in monetary valuation does mean they will.

Research needs to address alternative methods for expressing the values people hold with respect to the diversity of life and reasons for its preservation.

General issues of scales within complex systems in relation to economic and political perspectives on biodiversity

Thomas Koetz, Institute for Environmental Sciences and Technology, Universitat Autonoma de Barcelona, Spain

SUMMARY: Fundamental aspects related to the issue of scales are connected to questions concerning (i) How scale, extent, and resolution affect the identification of patterns; (ii) How diverse levels on a scale affect the explanation of social phenomena; (iii) issues of generalisation and (iv) optimisation of processes at a specific levels. When dealing with complex systems a key to these questions is the recognition that any specific object of study has a double nature: as a whole including objects of lower scales and as a part of an object at a higher scale.

When wanting to talk about scales and levels in a multi-/interdisciplinary arena it is necessary to be clear on the use of the key terms applied in order to achieve more transparency in arguments. Gibson et al. (2000) argue that while there exist a high awareness and relatively well-defined concepts of scales in natural sciences, in social sciences often less precise and varying conceptions of scales have been applied.

In this sense, these authors have suggested some definitions for key terms (see Box) and stressed the need "for social scientists to identify more clearly the effects of diverse levels on multiple scales in their own analyses, to comprehend how other social scientists employ diverse kinds of levels and scales, and to begin a dialogue with natural scientists about how different conceptions of scales and levels are related." (Gibson et al. 2000, p 218) This electronic conference offers a great opportunity to engage in such a discourse and learning process.

Gibson et al. further suggested four areas of theoretical questions related to issues of scales, which they see as 'fundamental to the task of explanation in all science': " (1) How scale, extent, and resolution affect the identification of patterns; (2) How diverse levels on a scale affect the explanation of social phenomena; (3) how theoretical proposition derived at one level on a spatial, temporal, or quantitative [and analytical] scale may be generalized to another level …; (4) How processes can be optimized at particular points or regions on a scale." (Gibson et al. 2000, p 221)

While (1) is pointing to the problem that choosing one perspective of a given system shows some patterns while hiding others implies the existence of multiple nonequivalent but equally legitimate observations (Giampietro 2004), (2) picks up issues of 'up- and downward causation', referring to key variable(s) being used in an explanation that occurs at a lower/higher level than the object of explanation (Gibson et al. 2000). Point (3) refers to issues of up- and down scaling, meaning the application of findings from lower to higher levels, and vice versa, while (4) raises important questions that have to be dealt with on a day to day basis especially in economics and studies of multilevel political systems as the EU.

Acknowledging that the systems we have to deal with in biodiversity conservation issues (biological as well as human systems) are self-organising, adaptive, and organised in nested hierarchies, we are confronted with a set of aspects that make the issues addressed by Gibson et al. far from trivial. One very central characteristic of such systems is the emergence of new properties when moving from one level to the next.

As a consequence, when focusing on a specific object of study within a certain space-time window it is necessary to bear in mind that (1) this object as a whole is made of smaller parts (existing on lower levels – smaller space- and time spans) that determine its structural stability as a whole; and (2) that this object itself is part of a higher-level structure (larger space- and time spans) setting functional constraints. "Hence, no description of the dynamics of a focal level can escape the issues of structural constraints (what/how, explanations of structure and operation going on at lower levels) and functional constraints (why/how, explanations of finalized functions and purposes, going on at or in relation to the higher level)." Giampietro (2004, p 39), Further research is needed to explore structural issues related to the application of economic and political theory on biodiversity issues on the one hand, and functional issues of the embedding economic and political systems and of potential alternatives.

A selection of definitions of key terms suggested by Gibson et al. (2000, p 218):

- Scales: the spatial, temporal, quantitative, or analytical dimensions used to measure and study any phenomenon;
- Levels: the unit of analysis that are located at the same position on a scale. Many conceptual scales contain levels that are ordered hierarchically, but not all levels are linked to each other in a hierarchical system;
- Hierarchy: a conceptually or causally linked system of grouping objects or processes along an analytical scale;
- Extent: the size of the spatial, temporal, quantitative, or analytical dimensions of a scale;
- Resolution: the precision used in measurement.

Political and economic scales in relation to biodiversity: Summary Week 1

Sybille van den Hove and **Thomas Koetz**, Institute of Environmental Sciences and Technologies, Autonomous University of Barcelona and **Ekko van Ierland**, Wageningen University, Session III Chairs

The objective of this first week of Session III was to discuss issues of scales in relation to (i) economic theory and methods, (ii) biodiversity valuation, (iii) integrated ecological-economic modelling and (iv) equity. In particular the aim is to identify research priorities on these topics and other topics related to economic scales and biodiversity.

Some key research questions or problems that were identified and discussed were:

· Further research is needed to explore structural issues related to the application of economic and political theory on biodiversity issues on the one hand (what/how, explanations of structure and operation going on at lower levels), and functional issues of the embedding economic and political systems and of potential alternatives (why/how, explanations of finalized functions and purposes, going on at or in relation to the higher level). [T. Koetz]

· There is a strong gap on understanding socio-economic and socio-cultural aspects of biodiversity conservation for sustainable development in specific ecosystems, such as e.g. the mountains [N. Chettri].

· Government intervention are not always a magic cure-all in correcting markets failures. With many competing social and economic objectives to be satisfied, and many political issues to deal with, there are nearly always failures in government intervention, inefficient or uncoordinated regulations, policies and institutions that exacerbate adverse impacts on biodiversity. These should be further explored, as well as the potential of decentralization and self-organization. [M. van den Heide]

· Research is needed to analyse public-private partnerships for biodiversity conservation and management in order to bridge between biodiversity interests and economic interests at the local scale (case studies and research on underlying juridical, political and social issues). [J. Jansen]

· Innovative research is needed in the field of environmental problems, particularly biodiversity related research, to balance technological progress between ecology and economy. [J. Tack]

· The issue of "space" seems to be one of the most prominent areas where misunderstandings and problems for research that integrates knowledge from ecology and economics are likely to happen. In particular, economic parameters may differ on other scales than the relevant ecological parameters. To integrate knowledge from the two disciplines, approaches should not to discuss the problem of space in an abstract manner but rather start from a particular conservation problem, whose structure will determine the spatial scale for both the economic and the ecological research. [F. Wätzold]

· The valuation problem is a key challenge. While some confusion remains on which terminology to use, a distinction can be made between economic monetary valuation and a broader understanding of valuation (e.g., "validation" was defined by Jurgen Tack as the process to make something at once relevant and meaningful). Limitations to the existing (economic monetary) valuation methodologies were stressed (e.g. accounting for environmental services from organisms and ecosystems or monetary valuation of aesthetic values [N. Chettri];) in particular difficulties in relation to temporal and spatial scales (for the temporal scale: discounting, myopic effects, focus on the immediate, and intergenerational inequity; and for the spatial scale: lack of attention to space or distance, the "power of money" as the judge of value, differences between local and global valuations of the same object, focus on species higher in the food chain) [C. Spash]. Valuation of biodiversity in a broader sense requires a better understanding of the processes behind the loss of biodiversity and a whole new ecological and economic language which is not mathematical (not in the way we know mathematics today). [J. Tack] Research needs to address alternative methods for expressing the values people hold with respect to the diversity of life and reasons for its preservation. [C. Spash]. Such methods and their results depend on context [F. Rauschmayer], in particular on which stakeholders are involved, and on space and time scales [M. Sharman]. Methods are needed that allow for the valuation of whole ecosystems or landscapes, taking account of all the very many socio-ecological elements [J. Dick].

· The diversity and role of some organisms and ecological processes which provide important services (such as e.g. the role of decomposer organisms in selfpurification of water) must be studied to estimate their contribution to environmental goods and services. [L. Rossi]

· As biodiversity and ecosystem services are crucial for the livelihoods of many (poor) people globally, there is a need to manage, rather than just conserve, biodiversity. Proper management of biodiversity requires policy measures at all levels, ranging from local to global. However, in economic terms, people living in or near a protected ecosystem often capture little benefit from preservation or sustainable resource use. The benefits of biodiversity protection increase with the scale from local to regional to national to global. In contrast, the economic costs incurred as a result of biodiversity protection prescriptions follow an opposite trend. The heaviest burden tends to be borne by people situated in rural areas, in the vicinity of protected areas. [M. van den Heide] More research is needed into the equity aspects of biodiversity conservation, restoration and management.

Our warmest thanks to the keynote contributors and all participants for their input in the first week. Next week, we will address political and policy scales in relation to biodiversity. There will be keynote contributions on (i) Multi-level biodiversity governance of the European Union; (ii) Justice, institutions and scales for biodiversity governance; (iii) Dealing with political scales in biodiversity governance in practice; (iv) Scales in conservation theories and strategies: a Southern perspective; (v) Issues of scales in biodiversity governance in practice: biodiversity governance and indigenous people.

We are looking forward to your contributions, to lively discussions and to suggestions for priority research topics to halt biodiversity loss. And we encourage young researchers to also take the floor!

The challenge of multi-level biodiversity governance in the EU

Susan Baker, Cardiff University, UK

International Governance: The ratification of the United Nations Convention on Biological Diversity (CBD) coincides with the emergence of several controversial interfaces between biodiversity and the spheres of politics and commerce. The CBD reflects the UNCED understanding of sustainable development, affirming the primacy of social and economic development, while coupling that development with biodiversity protection. Its complexity and scope, relative lack of public visibility, political ramifications and the under-developed nature of its key tools represent major challenges to implementation. Nevertheless, the CBD is significant as it is redefining biodiversity in social, political and economic terms.

Implementing the CBD: Stress on EU Multi-Levels Governance: The EC signed the CBD in 1992. Until 1998 policy remained ad hoc, partly due to lack of enthusiasm about the Convention. The current drive to address biodiversity issues is driven by (1) an institutional logic and (2) a functional logic.

Institutional Logic: The regular meetings of the CoPs to the CBD provide a reporting and implementation dynamic that has propelled EU-level action, particularly within the Commission. The 1998 Biodiversity Strategy and the sector specific Biodiversity Action Plans are evidence of this. Thus, the claim of regime theory that institutional co-operation between states, over time and on a regular basis, influences national behaviour, applies well to the evolving EU engagement. While this institutional logic drives collective EU-level action, it exposes the ambiguity between the boundary of Commission and Member State jurisdiction, that is, how the principle of subsidiarity is to be interpreted and applied. Here, Member States have been reluctant to concede competence to the EU. The Strategy relies upon the Birds and Habitat Directives, which promote in situ conservation through the Natura 2000 programme. The Strategy is weakened by poor implementation within Member States. This has the potential to undermine both the ability of the Community as a whole to meet its CBD obligations and its influence on the developing global, rule making regimes of environmental governance. This strengthens Commission resolve to deliver on its obligations. Functional logic: The incompatibility between the boundaries formed by political and administrative structures and those formed by ecological criteria, such as the composition of land cover and of ecosystems, means that ecological criteria are increasingly being called upon as a basis for delineating units of conservation within Natura 2000. The use of 'bio geographical' and 'ecological' regions force cross-border and inter-regional co-operation. The Commission acts as the main conduit for this co-operation. Hence, the functional logic of biodiversity management, a logic expressed through scale and space, deepens the EU's involvement. Biodiversity management has become entangled in webs of collaboration and policy co-operation, at the subnational, regional and inter-regional levels. EU biodiversity policy is driven by an institutional and functional logic; when viewed within the context of the longer-standing EU role in pollution management, this is resulting in an integrated environmental policy space at the European level.

Further Research: The three fold dynamic, between the international, European and national levels, is giving rise to a growing structural stress between the different levels of governance involved in biodiversity management. On the one hand, collective action is driven forward by functional and institutional logics. On the other hand, there are the constraints placed on the Community by its own structures of governance. These include, at the international level, the cumbersome system of 'concurrent competence' and domestically, its fragmented policy making system. This brings the Commission policy responsibility without the power to deliver on policy outcomes through control over implementation. Understanding the dynamics of EU biodiversity policy means unify the research insights gained from the study of the EU as an international institution and actor with the knowledge gleaned from the study of the EU's internal system of policy governance.

Justice, Institutions and Scales for Biodiversity Governance

Jouni Paavola, CSERGE, University of East Anglia, UK

SUMMARY: This contribution examines linkages between institutions, scales and justice in the governance of biodiversity. Environmental governance is intimately intertwined with social justice. One reason for this is that environmental governance is about the resolution of environmental conflicts by the establishment, modification or reaffirmation of governance institutions (Adger et al, 2003; Paterson et al, 2003). Governance institutions are the "rules of the game" (North, 1990; Ostrom, 1990) with regard to environmental resources such as biodiversity: they settle benefit sharing by defining resource rights and their limits, establish burden sharing for provisioning of environmental resources, and provide for monitoring, enforcement, conflict resolution and participation in environmental decision-making. This is the function of legal institutions such as the Habitats Directive and the Birds Directive and pertinent national legislation of Member States in the governance of biodiversity in Europe.

Several reasons (Berkes 2002; Young, 2002a, 2002b) demand the involvement of multiple scales or levels of decision making and jurisdiction in environmental governance. One reason is the physical attributes of environmental resources –resources such as habitats of migratory birds are so large that they cross the boundaries of primary jurisdictions and require the involvement of higher-level jurisdictions and the formation of "nested" structures (Ostrom, 1990). The involvement of higher level jurisdictions in negotiating and implementing benefit and burden sharing arrangements may also be helpful when the costs and benefits of environmental resources or their governance are spread across primary jurisdictions. Sometimes the involvement of higher or lower level jurisdictions can also make governance and provisioning of the resource less expensive. For example, weak capacity of central states in the developing world often makes local involvement in governance of biodiversity necessary on the grounds of effectiveness.

Justice is important in environmental governance because all governance solutions have justice consequences and because decisions to adopt governance solutions need to be legitimate among involved and affected parties (Adger et al., 2003). Justice also influences the effectiveness of environmental governance solutions which rests on voluntary compliance and legitimacy. Justice and legitimacy have two key dimensions (Paavola 2004, 2005). The legitimacy of environmental decisions rests in part on procedural justice, which encompasses issues such as recognition, participation and distribution of power. But it also rests in part on distributive justice – the fairness of the incidence of beneficial and adverse consequences among the involved parties. Procedural and distributive justice are intertwined because lack of participation in decision-making often translates to adverse distributive outcomes.

The involvement of multiple scales translate to greater number and heterogeneity of involved parties, which can make it more difficult to attain legitimate solutions to governance problems (Paavola, 2005). Multi-level solutions can also make procedural justice more difficult to attain because those who are directly affected by governance solutions at the local level often have little voice in decisions made at the higher levels. Local protests over the designation of Natura 2000 sites in Member States are examples of responses to this problem (Hiedanpää, 2002; Paavola, 2004). But the involvement of multiple scales also offers new possibilities for resolving distributive and procedural justice issues in environmental governance. Benefits and burdens of maintaining and enhancing biodiversity can be redistributed more widely among member states and between different groups. Best practice fair procedures could in turn enhance impartiality and legitimacy of environmental decisions both in the European Community and in the Member States. Yet governance solutions are seldom systematically studied from a social justice viewpoint to draw applicable lessons.

Dealing with political scales in biodiversity governance in practice: the example of the Biodiversity Convention in German

Horst Korn , Federal Agency for Nature Conservation, Isle of Vilm, Germany

SUMMARY: There is a mismatch in several important areas between the international obligations of the Country as a Party to the CBD and the different levels of government (province, county, community) that are responsible for the implementation of the Convention.

The Federal Republic of Germany is a party to the Convention on Biological Diversity (CBD) and is therefore responsible for its implementation. The CBD has a very wide mandate, covering 7 thematic and 17 cross-cutting issues as well as the Biosafety Protocol. Many of the obligations to be fulfilled are not, or not entirely, the responsibility of Germany as a party. The European Union, which is itself a party, is responsible for community issues, e.g. trade related issues, agriculture and fishery. More difficult is the implementation on lower scales like the provinces, counties and communities. Germany as a federal state has 16 Provinces (called Laender) and our constitution grants them the sole responsibility for political areas like nature conservation or education. Any international commitment in these areas by the federal state cannot be implemented against the will of the provinces! A good example of that dilemma is the problem that Germany had with the implementation of the EU-Habitats Directive, due to the lack of designation of sites by the provinces. Any protected area in Germany (except for the Exclusive Economic Zone in the Marine Environment) has to be designated by them (even National Parks!) and often they have different priorities or interests (mostly economic) than the federal government. But even when the majority of the Laender are willing to implement, Germany as a country may not be able to fulfil its international obligations in some areas until the last province has decided to do so.

Another big topic within the CBD that touches on almost every work programme of the convention is the implementation of the Ecosystem Approach as a strategy for the integrated management of land, water and living resources. It calls for wide stakeholder participation in decision making. Again, when it comes down to the designation of sites or other management issues, the federal state has no responsibility and no say on these issues. Here the planning authority lies often not even with the province but on the level of the municipality or the county. The only options that the federal government has are to fund research on these topics or to develop guidelines and guides of best practise for others to use as they wish. So far the political will is lacking, especially in the provinces, to make any change to the system that would mean for them to loose political power to the federal government. But due to the present system, the international engagement of Germany is weakened in some areas, such as nature conservation.

As a first step it would be good to have, on the European Scale, an overview of possible mismatches between international obligations of a country and its internal structures for implementation. That information could then be used to suggest improvements of the system, taking into account the different political structures of a country.

Re: Dealing with political scales in biodiversity governance in practice

Jan Jansen, Radboud University Nijmegen, Institute for Bio Sciences, Nijmegen, The Netherlands

The province of Noord-Brabant is very active to increase the quality of natural areas in our landscape. The province is ready to meet the challenges that have to be faced and aims to resolve the dilemmas in a way that will preserve and enhance its ecological capital for the future. The province of Noord-Brabant was the first region in Europe to embrace Countdown

2010 and the international and European targets to stop the loss of biodiversity by 2010. The province aims to implement its strategic goals in its current policies and activities.

An important programme is intended to realise an uninterrupted network of existing and yet to be developed nature and woodland areas in the province of Noord-Brabant, the Ecological Framework (EHS). In 2012, the EHS is supposed to consist of well over 140,000 hectares of nature and woodland area and well over 1,500 km of Ecological Corridors. Regional planning spatially protects the EHS. Another programme is aimed at maintaining and reinforcing the ecological and landscape quality of the countryside outside the EHS. The programme primarily targets the areas with current or potential nature and landscape values. In total this concerns approximately 125,000 hectares of countryside. The guiding principle is the restoration of the ecological cohesion in relation to the EHS, cultural-historical and landscape values, and the functioning of water and soil systems. A third programme pertains to nature and landscape in the urban areas.

Now the province has a large team to do the inventories of plants species, vegetation and avifauna in the countryside. This work is repeated every 11 years. Besides inventories, the team also works on monitoring the effectiveness of the provincial policy regarding nature values in Noord-Brabant.

The inventory work contributes very well to our knowledge of the nature quality of the countryside. The monitoring network provides insight in biodiversity trends. Every 4 years a report on this is published (e.g. Van de Staaij & Van der Linden 2004). However, despite all these efforts, our exact knowledge of the EHS is rather incomplete. The method used has never been tested scientifically. Most of the biodiversity is in the EHS. How can we know that we have stopped loss of biodiversity if we don't know the situation now? In order to know the situation we probably also have to do inventories of the nature areas and corridors (EHS).

The question rises who is going to do that? The government? Or is it the province or perhaps the municipalities? And how are they going to this? Is there one method, the same as in another municipality, province, government? Wouldn't it be great if we had one reliable method to do the work? Would the EU not be the most appropriate level to design such a reliable method? And wouldn't it be practical to work on the level of provinces? That level is less complex, including only few landscapes with related biotopes and land-use systems.

Now there are obligations of the national authorities for the Natura 2000 network. The Habitat and Bird Directives express the need to promote biodiversity by maintaining or restoring certain habitats and species at 'favourable conservation status' within the context of Natura 2000 sites, while taking into account economic, social, cultural and regional requirements, as a means to achieve sustainable development. Who is going to examine this and who is going to pay for it? The owners of the site, the authorities? If the provinces would do the examination (read monitoring) then the national authorities should give them the financial tools.

Indeed I agree with Horst Korn from the Federal Agency for Nature Conservation in Germany, that as a first step it would be good to have, on the European Scale, an overview of possible mismatches between international obligations of a country and its internal structures for implementation. That information could then be used to suggest improvements of the system, taking into account the different political structures of a country.

Political and economic scales in relation to biodiversity: Summary Week 2

Sybille van den Hove and **Thomas Koetz**, Institute of Environmental Sciences and Technologies, Autonomous University of Barcelona and **Ekko van Ierland**, Wageningen University, Session III Chairs

The objective of the second week of Session III was to discuss political and policy scales in relation to biodiversity. Keynote contributions addressed the challenge of multi-level biodiversity governance in the EU from three different perspectives: (i) from a political science perspective focusing on the current drivers of EU biodiversity policies in terms of (a) an institutional logic and (b) a functional logic; (ii) from the practical point of view based on the experience in a Member State (Germany); and (iii) focusing on issues of social justice examining linkages between institutions, scales and justice in the governance of biodiversity.

In her contribution Susan Baker focused on the consequences of EU biodiversity governance being embedded in the Convention on Biological Diversity (CBD). She argues that the three fold dynamic, between the international, European and national levels, is giving rise to a growing structural stress between the different levels of governance involved in biodiversity management. On the one hand, collective action is driven forward by functional and institutional logics. On the other hand, there are the constraints placed on the Community by its own structures of governance. Understanding the dynamics of EU biodiversity policy means unifying the research insights gained from the study of the EU as an international institution and actor with the knowledge gleaned from the study of the EU's internal system of policy governance.

Reasons why implementation has been so difficult in the case of Germany are addressed in the contribution of Horst Korn, stressing the mismatch in several important areas between the international obligations of the Country as a Party to the CBD and the different levels of government (province, county, community) that are responsible for the implementation of the Convention. He concludes that as a first step it would be good to have, on the European Scale, an overview of possible mismatches between international obligations of a country and its internal structures for implementation. That information could then be used to suggest improvements of the system, taking into account the different political structures of a country.

Also addressing problems of implementation, Jouni Paavola highlights the need to pay more attention to issues of social justice that arise in multi-level governance, in particular as means to influence the effectiveness of environmental governance solutions which rests on voluntary compliance and legitimacy. He argues that the involvement of multiple scales in biodiversity governance offers new possibilities for resolving both distributive and procedural justice issues. Yet governance solutions are seldom systematically studied from a social justice viewpoint to draw applicable lessons.

Our warmest thanks to the keynote contributors for their input in this second week. We were a bit disappointed by the lack of discussions and reactions from participants and hope that there will be more in the last three days of the e-conference during which comments on the contributions from week 2 are still welcome. Next week (until Wednesday), we will address the topic of Integration of ecological, political and economic scales with contributions on (i) Multi-Scale Integrated Analysis of complex environmental issues, and (ii) Interdisciplinary methods and the integration of ecological, political and economic scales. We are looking forward to your contributions and to suggestions for priority research topics to halt biodiversity loss.

Scales in conservation theories: Another clash of civilizations?

Mariteuw Chimère Diaw, Center for International Forestry Research (CIFOR)

SUMMARY: Conservationists should reposition humans -people- at the heart of the project to reorganize our long-term relations with other species and the environment; they should invest more energy, research, and resources in the reinvention of North-South solidarities in global economic and environmental forums and in locally constructed multiple use landscapes focused on human development and pluralist governance of biodiversity.

Leading conservation biologists make the case that, in this age of global extinction alert, protected areas do not meet the spatial and temporal scales of biodiversity conservation requirements anymore. They propose a new radical attempt to form development-free ecosystems spanning entire regions and continents. The key ingredients of this agenda include: (i) rejection of "the myth" of sustainability; rejection of development, which "for biodiversity… is really de-development or denaturation" (Soulé and Terbogh, 1999). (ii) investing the mapping arena (with the related manipulations of scale) to control the global land use agenda (ii) expanding the areas under 'strict protection' through "top-down" impositions, backed by the state and the military, including internationally financed "nature keeping" forces

These profound ideological stances explain the importance of maps for control of the land use agenda: "maps stimulate desires – for territory, for natural resources, for real estate development, even for conservation…If developers are the only people mapping the land's future use, then they control the land use agenda… If maps are the agenda, then conservationists must enter the mapping arena. They must begin producing attractive alternatives – maps that also promise a social good: the benefits of wilderness and nature protection" (Soulé and Terbogh, 1999). The main form that has recently taken the production of maps in the global conservation scene is the development of "cartographically-enabled regional land use planning approaches under the rubrics of ecoregional conservation, hotspots, transboundary protected areas, and others" (Brossius, 2004). Taking advantage of the new availability of powerful GIS-remote sensing imageries and algorithms, large conservation NGOs have moved to produce "compelling, visually exuberant images of regions and locations targeted for conservation" (Brossius, ibid). This has been very effective in attracting strategic donor support to a global land use agenda that tends to exclude local people (Mac Chapin, 2004). For, the new maps do not just enable the visualization of biodiversity at larger, multiple spatial and temporal scales; they also deliberately exclude social and political interfaces as prime discriminators of these cartographic representations of the world. In that process, "assumptions about human communities become coded (as threats) or elided… this, in turn, produces capillary processes of power by which visualizations are transferred from map to ground" (Brossius, 2003).

We indeed see the need for a change of spatial and temporal scales; past bureaucratic planning of both development and conservation has largely resulted in the fragmentation of tropical forest landscapes into myriads private concessions and protected areas weakly connected to communal lands and economies.

The black boxing of the social and the political in modelling and mapping decisions is a self-made sociopolitical trap and a recipe for surprises. The story of the Meso-american Biological Corridor (MBC), which transformed from a purely bioecological project in mid-80s to include rural development by 1997 under the pressure of indigenous groups followed by their governments, is telling. Today, some its initiators pester against the hijacking of biodiversity by development while others relish the opportunity of weaving a larger web where people and wildlife can coexist (Kaiser, 2001)

This, in a nutshell, sums up the issues of scales, spatial and temporal, as well as political, socioeconomic and ethical, at which human development and biodiversity governance will be addressed together or will fail. Conservationists should reposition humans –people- at the heart of the project to reorganize our long-term relations with other species and the environment. This means reinventing new forms of North-South solidarities in global economic and environmental forums, as well as in political ecoregions and sociocultural landscapes. The construction of socially-oriented multiple use landscapes at local and regional levels constitute today the key challenge for research and action in development and biodiversity governance.

Multi-Scale Integrated Analysis of complex environmental issues

Mario Giampietro, Istituto Nazionale Ricerca Alimenti e Nutrizione (INRAN), Rome, Italy

I would like to start this short discussion from the title of a book edited by Dasgupta and Mäler (2004): "The economics of non-convex ecosystems". This seems to be the state of the art of politically correct Environmental Economics arriving to justify even the application of Precautionary Principle (!). The bizarre expression "non-convex ecosystems" is used to admit that life does not behave as predicted by the equations (in economic jargon that ecosystems express a non-linear behaviour). Such an expression represents also a desperate attempt to maintain the claim that standard analytical tools of economics can handle the challenge posed by self-modifying systems [a class introduced by Kampis, 1991].

The provocative question of Schrödinger (1967) "what is life?" wanted to point at a major epistemological challenge introduced by living systems. The identity of the converter defines the quality of the energy input, from the point of view of the user. That is, for living systems there is no substantive definition of energy or resource or cost or benefit. Hay is energy for a mule but not for a car, electricity is energy for a refrigerator but not for a human being. Oil was an entertaining burning water at the time of Marco Polo, but is a key resource today justifying wars. The problem with life is that it is a property expressed by complex adaptive systems operating across different scales. This is why the only convincing formal definition of life proposed so far is that life cannot be simulated using formal systems of inference (Rosen, 2000). Getting into the field of theoretical ecology things do not get clearer. There is no consensus on how to define biodiversity in formal terms or on what should be considered the unit at which evolution occurs. The familiar concepts of organisms, ecosystems, species, populations, landscapes refer all to types, which are by definition out of scale (Allen and Hoekstra, 1992). It is only when all these types are properly scaled into a holistic picture of their actual interaction across scales that it becomes possible to get into the business of looking for indicators of stress or performance (what a cost or a benefit means in the ecological realm). A genetic mutation that is lethal for the survival of individual organisms represents the only hope of survival for the species to which the individual belong. Defining what is a cost and what is a benefit in a nested hierarchical system is a very treacherous endeavour.

In a world of living beings the choice of who is the observer and the narrator becomes crucial. In Buddhism suffering is defined as a discrepancy between a given set of expectations and a given perception of the reality. In science defining costs and benefits implies formalizing, within a given system of accounting, a series of semantic concepts associated with costs (suffering) and benefit (enjoyment of life). Buddha warned us that there are different paths for dealing with suffering: (i) you can eliminate it by removing the perceived problem. This implies taking successful action within the original perception/representation of reality; (ii) you can change your expectations about the reality (changing your goals within the same narrative); (iii) you can meditate in order to be able to verify and update your actual perception of the reality (checking the relevance of your narratives). In this way you become a different person, a different observer/agent that suddenly discovers to live in a different reality.

Very few scientists working in the business of formalizing costs and benefits within a given system of accounting (either in economic or biophysical terms) seem to be aware that their formalizations (numerical assessment) entail a pre-analytical selection of a narrative (and a scale). In turn any narrative entails/requires an observer/agent that must have an identity associated with values, goals and taboos at any given point in space and time. Even if hard scientists seem not to be aware of the positive effect of meditation, human systems do change their identity in time across scales, defining new observers, narrators, and carriers of values, goals and taboos. The same phenomenon is expressed by ecological systems. Complexity implies the existence of: (i) multiple relevant time differentials for the perception and representation of the process of becoming on multiple scales; (ii) non-equivalent observers/agents adopting logically independent perceptions and narratives about costs and benefits.

Strategically important research that should be undertaken: (1) developing a new epistemology for handling the perception and representation of the process of becoming on different scales. An epistemology which acknowledges that the observer/narrator is a part of the self-modifying system. This requires participatory integrated assessment, since when dealing with sustainability substantive definitions of what is a cost and what is a benefit simply do not exist. They do not exist when considering different actors within socio-economic systems, let alone when including also the interaction with ecological systems; (2) Avoiding to collapse the descriptive with the normative when dealing with sustainability issues. Sustainability requires dealing with different types of costs and benefits perceived and represented in logically independent ways on different scales by non-equivalent observers/agents. This implies that it is the quality of the process of evaluation (who decides whose perspectives count and how) that will determine the final output. Therefore, scientists should abandon the dream of reducing, from their desk, different typologies of costs and benefits into a single system of accounting (e.g. the creative attempts of doing Cost-Benefit Analysis applied to sustainability issues in Environmental Economics). Introducing fancy expressions such as “non-convex ecosystems” will not do it.

Interdisciplinary methods and the integration of ecological, political and economic scales: when the language of math becomes a unitless scaler and some related consequences

Katharine Farrell, Institute of Governance, Public Policy and Social Research, Queen's University of Belfast, Northern Ireland

SUMMARY: Informed biodiversity policy requires multi-dimensional, spatio-temporally complex ecological political economy data that cannot be compiled through reliance on mathematical analytics.

It is hoped the current contribution can build on preceding discussions. I assume that biological diversity plays an important role in ecosystem and species level resilience (Gunderson et al. 2002, Levin 2000) and address the question: how can meaningful ecological political economic data on dynamics impacting biodiversity be compiled?

Focusing at the level 'species', looking specifically at homo sapiens (humans), let us consider the human activity 'making decisions concerning biodiversity policy', which includes a sub-activity 'interdisciplinary ecological political economic analysis'. In particular, let us consider how interdisciplinary co-operation between analysts within the academe can support integration of information across ecological, political and economic scales while preserving relevant discipline specific data.

In order to discuss relationships we can employ hierarchy theory (Allen & Starr 1982), viewing the functional roles of three activities, ecological, political and economic analyses, as components of a higher order composite activity ecological political economy analysis (EPEA). Each component activity has a similar relationship with the composite activity but a distinct nomenclature, ontology and epistemological perspective on the analytical object 'biodiversity'. Communication between the component activities takes place at the EPEA level (Figure 1) and will necessarily employ a fourth distinct nomenclature, ontological and epistemological structure with regard to biodiversity. It might seem obvious, even efficient, to rely on mathematics to combine data from the three component frames. Copious discipline specific mathematical analyses on relevant topics are readily available and operationally they can be easily combined. This is a very bad idea (e.g. Sousa & Domingos 2004)

EPEA data relevant to the activity 'making decisions concerning biodiversity policy' must report on dynamics operating at a highly complex interface. Assume total system boundaries are fixed at political boundaries. Predicting whether a given EU policy will produce improved biodiversity conditions for migratory seabird populations requires knowledge, not only from the three constituent disciplines but also from the EPEA level, where local political obstacles to implementation in a key 'stop over state', due to specific economic and/or ecological issues, may be critically relevant. The EPEA analytic must process data (including feedback relationships) where impacts are iteratively transposed across all three areas (Figure 2).

Scale difference can be nested but it can also be crosscutting. Multi-dimensional spatio-temporal inter-scale crossovers and feedback are central to EPEA (Giampetro 2004, Giampetro & Mayumi 2000a, 2000b, 2001). Concerned with scaling across multiple dimensions, Costanza proposes that the fractal like character of life orientated and therefore systematically stochastic phenomena (in this case a shoreline) can nonetheless be mathematically "described using a unitless dimension that summarizes how it changes with resolution…allow[ing] convenient scaling of predictability measurements taken at one resolution to others" (Constanza 2003:660). Prigogine (1997) observes a similar strategy in Hawking's (1998) concept of 'imaginary time', used to defend his reliance upon formulae that assume time reversibility. We may class imaginary time, Costanza's unitless dimension and reliance upon math for EPEA as 'unitless scaler' approaches. Bumpy EPEA scale differences cane be 'smoothed' through the abandonment of discipline specific reference structures but the results are meaningless. Because mathematical equations have no meaning, unless the symbols they employ are associated with a specific ontology, approaches that rely on unitless scalers cannot provide meaningful inter-scale data (Georgescu-Roegen 1971, Cao 2004).

Gaps in our knowledge: We do not know how common is such reliance upon unitless scalers in the EPEA domain.Because unitless scalers smooth away the very relationships that we need to look at in integrated assessments, non-mathematical integrative methodologies are urgently required. These methodologies will need to address inter-disciplinary collaboration as a social collective cognition activity and we lack data on this point. The above argument also leads to a perspective where comprehending the structure of EPEA problem (including biodiversity policy formulation), articulating aspiration and purpose and choosing between options are all complex activities that cannot be reduced to political, epistemological or intentional spheres. On this basis, inter-disciplinary scientific methodology concerned with EPEA research and subsequent recommendations relating to these choices is also high political philosophy, concerned with fundamental relationships between decision, power, choice and freedom.

Strategically important research that should be undertaken: Costanza and Hawking are not fringe theorists, their positions are mainstream and the problems arising from their use of unitless scalers is not widely acknowledged. Research into the prevalence of unitless scalers and the ontological and epistemological consequences of this practice is urgently required. The role of time and time as a complex and scale dependant factor, particularly with regard to 1) how a given anthropogenic impact may affect other species in unique ways, as a consequence of their species specific time context and 2) how the human time context may be related to economic and political impacts arising from ecosystem level dynamics associated with system stability, resilience and even disruption and restructuring into new systems. Existing knowledge on human cognition, philosophy of the mind, organisational management and group behaviour should be applied to this topic. The role of political philosophy in developing these methodologies should be strengthened and EPEA research teams should include political philosophers.

Figure 1: The EPEA / Biodiversity Complex

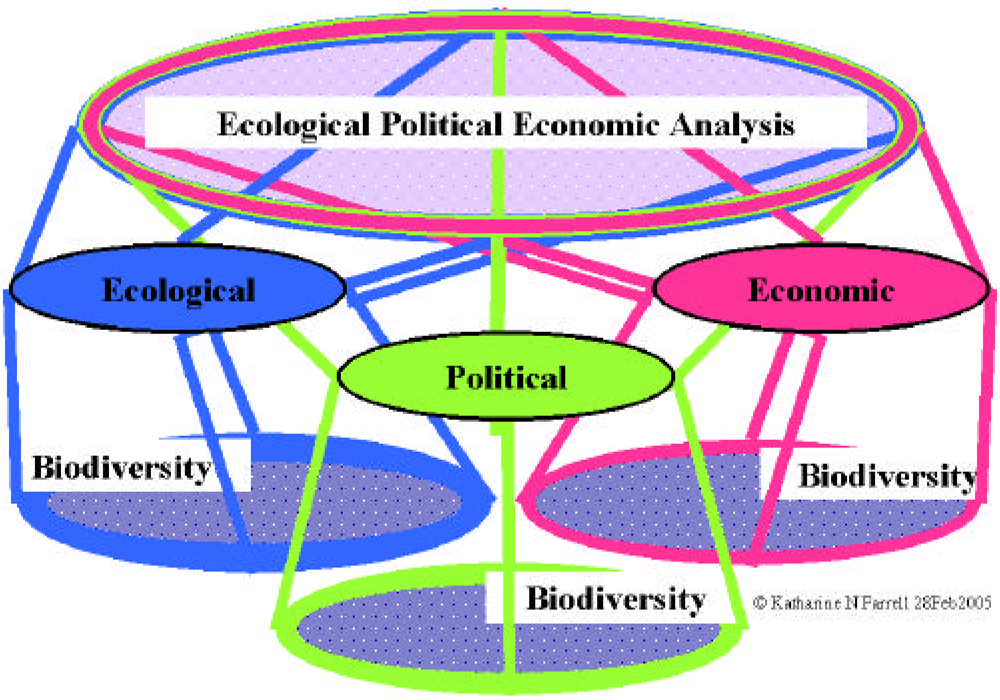
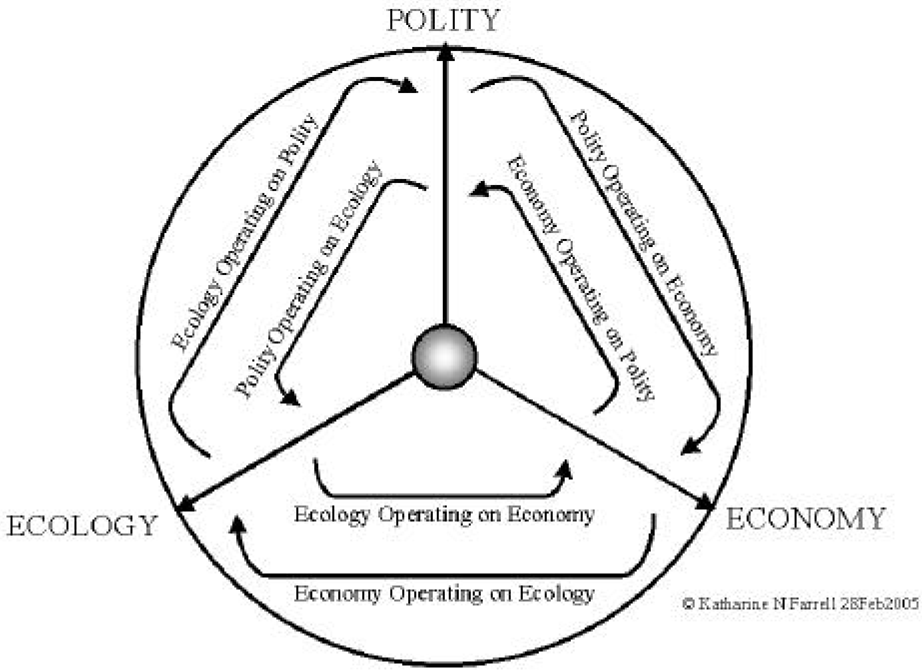


Figure 2: a freeze-frame view of the EPEA data complex



Political and economic scales in relation to biodiversity: Summary Week 3

Sybille van den Hove and **Thomas Koetz**, Institute of Environmental Sciences and Technologies, Autonomous University of Barcelona and **Ekko van Ierland**, Wageningen University, Session III Chairs

This third week of the e-conference aimed to discuss the integration of ecological, political and economic scales with contributions from (i) Mario Giampietro on ‘Multi-scale integrated analysis of complex environmental issues’ and (ii) Kate Farrell reflecting on ‘Interdisciplinary methods and the integration of ecological, political and economic scales’. The week started with a contribution by Chimere Diaw on ‘Scales in conservation theories: Another clash of civilizations?’ related more to the context of last week’s discussion – political and policy scales in relation to biodiversity. However we were delighted to see that the discussion picked up again resulting in further interesting comments on both economic as well as political aspects of our discussion. I will start with these comments.

Silva Marzetti argues in favour of monetary valuation of biodiversity at a local scale in form of cost-benefit analysis a useful tool for estimating values. Felix Rauschmayer, responding to this comment, questions the practicability of monetary valuation of biodiversity and the effects of it. He emphasises the need for studies to clarify whether ‘monetary values of nature are convincing authorities or the public to preserve nature’ or if ‘using monetary values may lead to crowding out of moral arguments for nature preservation’

Picking up on the ongoing, more normative, discussion on this issue Clive Spash addresses some of what may be underlying the disagreements over the role and meaning of the “mechanism of money”. He makes the case that there exist other social and political phenomena than money used by humans to (mis)manage their affairs. He claims that scientists can (and should) offer a lot more than one (the monetary) perspective on the value of biodiversity and the ways in which to approach its management. With regard to further research he stresses the need to identify characteristics of institutional and social systems that take a more multidimensional stance on values.

Jan Jansen’s comment supports the contribution of Horst Korn on institutional mismatches hindering effective biodiversity policy implementation raising open questions such as what polity-level in the EU multi-level governance is responsible for the definition of a reliable method for monitoring of biodiversity, its realisation, and the policy analysis of conservation efforts (such as the Natura 2000 network).

Robert Kenward argues in favour of restoration of biodiversity as in conservation through sustainable use as a second pillar for conservation, and a challenge. In line with the CBD he promotes adaptive management as a way forward where data are imprecise, with monitoring and research to refine management and offers a possible approach to enhance restoration (a combination of three new technologies). He suggests a revision of Directives on Birds, and on Habitats (through a Biodiversity Directive?) perhaps giving greater flexibility of protection measures. He further argues that Pan-European technology integration for conservation could (i) benefit research (standardized comparisons, large-scale experiments, rapid plug-in of results), (ii) underpin a European knowledge-economy in the environment and (iii) improve social cohesion by education, central-local communication and local cooperation initiatives.

Chimere Diaw focuses on new radical attempts of leading conservation biologists (e.g. Soulé) to form development-free ecosystems spanning entire regions and continents. He argues that new maps and remote sensing imageries do not just enable the visualization of biodiversity at larger, multiple spatial and temporal scales; they also deliberately exclude social and political interfaces as prime discriminators of these cartographic representations of the world and function thus as tools to support such strategies. Seeing the black boxing of the social and the political in modelling and mapping decisions as a self-made socio-political trap and a recipe for surprises, he argues in favour of the reposition of the people at the heart of the broad range conservation strategies. Accordingly he stresses the construction of socially-oriented multiple use landscapes at local and regional levels as to be the key challenge for research and action in development and biodiversity governance.

Addressing a multi-scale integrated analysis of complex environmental issues Mario Giampietro claims that complexity implies the existence of: (i) non-equivalent observers/agents adopting logically independent perceptions and narratives about costs and benefits or any complex ecological issue; (ii) multiple relevant time differentials for the perception and representation of the process of becoming on multiple scales. Therefore he calls for research on participatory integrated assessments (1) required for developing a new epistemology, which acknowledges that the observer/narrator is a part of the self-modifying system, (2) that focus on the quality of the process of evaluation (who decides whose perspectives count and how) avoiding collapsing the descriptive with the normative when dealing with sustainability issues – leading to the abandonment of reductionism.

Finally, Kate Farrell addresses the question of how meaningful ecological, political, and economic data on dynamics impacting biodiversity can be compiled. She makes the case against mathematical approaches that are often proposed by scientists to overcome differences of scales when integrating ecological, political and economic data. She argues that it is in these disciplinary differences (in nomenclature, ontology, and epistemological perspective) that important information relevant for biodiversity policy-making can be found. Instead of reducing this information to meaningless mathematical equations she emphasises the need to develop a fourth distinct interdisciplinary nomenclature, ontological and epistemological structure with regard to biodiversity that will articulate into non-mathematical integrative methodologies. In order to develop such methodologies she emphasises (1) the role of time as a complex and scale dependant factor, (2) the importance of knowledge on human cognition, philosophy of the mind, organisational management and group behaviour, and in particular (3) the role of political philosophy. Further, she stresses the need for research into the prevalence of mathematical analytical approaches to overcome scale differences and the ontological and epistemological consequences of this practice.

Integrating ecological and social scales

Introduction to Session IV: Integrating ecological and social scales

Ekko van Ierland, **Rob Dellink**, **Arjan Ruijs** and **Hans-Peter Weikard**, Environmental Economics and Natural Resources Group, Wageningen University, and **György Pataki** and **András Lányi**, Hungarian Academy of Sciences, Session IV Chairs.

Managing biodiversity, both marine and terrestrial, can only be successful if both ecological and social processes are considered simultaneously. Many competing claims occur for natural resources affecting biodiversity. Which of these claims can or should be satisfied depends on the vulnerability and resilience of the ecosystem and on the social and/or economic importance of them. Acceptance or rejection of such claims only on the basis of social or of ecological grounds will not result in efficient natural resources use.

A difficulty with biodiversity conservation is, however, related to the differences in geographical and temporal scales associated with natural and social sciences, different terminologies used, different aspects analysed and differences in the factors affecting the processes analysed. The ecological and social scales of species may sometimes overlap, but often there are substantial differences between them. For example, the state and amount of plankton available at different locations in the oceans is of main importance for marine biologists and ecologists as this is on the basis of whole system of marine living resources. Water temperature, ph, water flow, quality of river water flowing into the seas, etc. are of extreme importance for the number and types of marine living resources, their dispersion and therefore for the marine biodiversity in general. Social scientists concentrate more on the issues like the incentives of fishermen to continue fishing, the existence value of the blue whale, the historical changes of the fisheries industries, the economic viability of windmill parks in the sea, etc. Aspects to be analysed include for example, welfare, preference structures of individuals, economic efficiency, international cooperation, etc.

In scientific research, integrating gamma and beta sciences has proven to be both very desirable and difficult. Truly interdisciplinary scientists are rare, and working in multidisciplinary research teams is a challenge, to say the least. One of the main problems is the difference in terminology used. Apart from communication problems, there are also difficulties with respect to the content of the research. The relevant aspects and perspectives on a scientific problem vary largely between the gamma and beta sciences. Such differences in relevant aspects can largely be overcome by choosing the appropriate level of detail, i.e. by identifying the appropriate scale of analysis.

Also, the time scales differ. Changing social habits and processes is possible but takes a lot of time. Pollution effects on biodiversity may often be slow, but can be immediate once carrying capacities are reached and may also be irreversible. The time scale with which an ecosystem and economy adapt to changing circumstances are only indirectly related, and hence the temporal scales may diverge.

Furthermore, the spatial scales can be quite different. Where ecologists may look at both the micro-level of bacteria, insects and plankton as the macro-level of species distribution and dispersion, social scientists deal with people, regions, nations and international cooperation. One of the main reasons of the tragedy of the commons is related to the difficulty or impossibility to properly assign property rights. For migratory species that constantly cross international borders, assigning such property rights so necessary for sustainable management is impossible. Geographical scales also play an important role in the management of species and ecosystems: certain species exist only in some places and hence are only relevant for local ecosystems, while their social impact may be global. One can think of the existence value of a blue whale, a species that is considered to be valuable to people throughout the world, regardless of whether they live close to the habitat of the species or whether they will ever visit them. This brings us to the issue of scales and dimensions of value.

Biodiversity loss concerns different dimensions of value. First, the value of species must be clearly distinguished from the value of diversity. The value of diversity can be decomposed into components of instrumental and intrinsic value:

- Each species has a particular value: some are sources of food, some offer transport services, others can be used as a source for renewable energy or raw materials for various purposes, and we enjoy the beauty of flowers and butterflies. Biodiversity loss involves a loss of species or, in case of within-species diversity loss; it puts the survival of a species at risk. In any case, one component of the value of biodiversity is species value.
- The instrumental value of biodiversity cannot be attributed to particular species. Ecosystem services like the provision of fresh water or the fertility of soils rely essentially on the interaction of different species. Since there is evidence that species diversity on average supports the stability of an ecosystem, diversity as such plays a crucial role in providing these services. However, we must carefully consider to what unit the concept of diversity should be applied. Genetic diversity seems to be relevant for an assessment of within-species diversity which is a decisive factor for species survival and the potential for development in the evolutionary process. A useful concept of between-species diversity can hardly rely on genetic information. Ecosystem services, for example, are provided by chains of interacting species. What is important is the functional role of different species in that interaction rather than their genetic make-up. A diverse sample of species does not make a functioning ecosystem. Only for lack of knowledge of the complex interactions in an ecosystem we may focus on maintaining species diversity in order to maintain ecosystem services.
- Maintaining diversity also provides an insurance of species value. It is reasonable to rely on a mixed portfolio of species to insure the availability of food, raw materials and ecosystem services in an uncertain environment. This value of insurance is the option value of biodiversity. Moreover, not only the availability of particular sources of supply is uncertain but also future demand. The future demand for certain substances, e.g. for pharmaceuticals, can hardly be predicted, since it is generated by the results of future research and our medical and bio-chemical knowledge. The possibility of learning gives rise to a quasi-option value of biodiversity. Like ecosystem services, the option value and the quasi-option value are instrumental diversity values.
- The intrinsic value of diversity captures our preference for living in a more diverse and stimulating environment, even if a less diverse environment provides all essential services and sufficient insurance. According to Weitzman (1993, 158), "it seems that increasingly many people believe that there is some inherent value in preserving diversity, even though they cannot exactly define what it is." This observation needs some further comment. That increasingly many people attach value to diversity may reflect an increasing awareness of the importance of ecosystem services. Moreover, genetic diversity as such presumably does not mean much to most people. An intrinsic value of diversity is more plausible when it comes to observable features of the different species. We enjoy the diversity of shapes, colours, sounds, and ways of life of the different species. However the diversity of the observable features does not necessarily correspond to genetic diversity. Genetically similar species may be morphologically quite different and vice versa. The claim that there is an intrinsic value of diversity is too vague and needs further specification.

To conclude, as scales, terminologies and aspects analysed are so different, successful integration of social and ecological sciences is a necessity but requires cooperation between disciplines that is hard to realise, good communication, the proper scales and strong motivation.

The restoration of place

András Lányi, Institute of Sociology, Eötvös Loránd University, Budapest

We are living in a placeless world, claims Edward Relph, an urban geographer from. His statement is intended to be descriptive, rather than normative. The concept of placelessness only refers to the gradual disappearance of the significance of locality in our life. Local characteristics and bounds are increasingly sacrificed all over the world by:

- Mass production and standardisation, mass communication, transportation and copying that eliminate the rich diversity of local technologies, habits and styles by making everything reproducible and accessible to anyone without a concern for local conditions;
- Transforming the link between human beings and their environment into a contingent one, since all of us can almost live anywhere, we are living in the haze of a ceaseless mobility, and our decisive experience are formed by the virtual reality of the techno-telemedia;
- The spirit of business that sees the environment and nature as a pool, or stock, of actual or possible resources and services to be extracted if profitable;
- The devaluation of face-to-face, direct connections and relationships based on continuity in space that results in our preference for forming contingent communities with partners whom we might meet in contingent places, or do not meet at all, over forming communities with our neighbours;
- Modern architecture, the perfectionism of engineering calculations, as well as a post-modern eclectic that might disguise the dreary functionalism of the operation of the environment-machine, but cannot provide it with any substance, since both modern and post-modern have the same feature of impersonality and homelessness;
- The sacrifice of public spaces to the demands of motorisation and land speculation, against which urban planning attempts to protect urban places by the standardised regulation of places without respecting the exceptionality and originality of each and every place.

Writing in Hungarian (as was the case with this essay) makes it easier to discuss place in the sense that in Hungarian “place-less” (hely-telen) is, at the same time, wrong (helytelen). In contrast, good is right (helyes) and is endorsed (helyesel). I know no other language in which locality and morality have the same name. Are we going too far if we claim that all languages express particular, therefore diverse, historical experience?

It seems to me that we have to recognise the moral significance of creating and using places. The morality of pla ce helps us to understand that good and bad, right and wrong might only be judged by knowing the concrete, particular local situations and conditions. A similar meaning might be inferred from the rightly famous and wonderful moral claim made by Aldo Leopold: “A thing is right if it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise.” Human beings’ original experience relates to differentiating among diverse places, scales and directions (be they attractive or threatening) and not the homogeneity and uniform distribution in space. Space, as distance and closeness, border and connection, is brought into being by and amidst of localities possessing diverse meanings. As we are looking for the places of things and actions, we are posing questions to cultural patterns and forced to judge competing practices. However, the diversity of places and, relatedly, the diversity of cultural patterns and practices cannot be disconnected from their respective biotic context; the diversities of our places is interwoven with the diversities resides and evolves in nature.

Re: The restoration of place

Gyorgyi Bela, Institute of Environmental and Landscape Management, St. Istvan University, Godollo, Hungary

Other aspects of the "place morality" could be found in Zygmunt Bauman writings. Baumann (1998) claims that elimination of temporal/spatial distance by new information or transport technologies intensify the polarization on all levels of society.

A highly mobile elite is free from constraints of local space and they are able to distance themselves from any locality, they can refuse to participate in local discourses. For those who remained, "locked-in", who are not mobile, the significance of locality is not disappeared, yet; rather they mostly bear the adverse consequences of changes in the local environment.

Although biodiversity strategies in several nations as well as the Convention on Biodiversity (CBD) appreciate locality through underlining the importance of, e.g., local knowledge, local ecosystem health, local people participation in biodiversity conservation, but in public discussions at the national level the notion of locality is being devalued, it is being equated with terms like "old-fashioned" or "incapable" or "immobile".

Re: The restoration of place

Norbert Kohlheb, Szent Istvan University, Institute for Environmental and Landscape Management

The writer's fears and skepticism about the consequence of a placeless word are fully acceptable. I intend to use this virtual "place" for adding some remarks on sustainability aspects.

As far as I am concerned, the sketched trends of increasing time- and placelessness, as well as the similar use of places, contingent mobility and devaluation of face to face communication might cause some serious damages that hinder us from establishing foundations for sustainable development.

Three aspects could be mentioned:

On the one hand: We lose knowledge about how to use "places" in a sustainable way by using different places in the same way. In this regard we neglect ancient wisdom about sustainable use of "places" equipped with differing "resources".

On the other hand: By using different "places on the same way, we miss the possibility to test different "ways of life" regarding their sustainability. With this we close a loop of alienation and loss of knowledge about our adjacent surrounding.

And third aspect: The use of places determines our way of thinking about what is wrong and what is right. A highly standardized and widely accepted lifestyle will undermine our reflectivity to the changes in the environment. i.e. in the worst case we will accept severe damages to the environment and get away with these damages as normal and unavoidable.

Economic equality and the scale of analysis

Gregory Mikkelson, McGill University, Canada

SUMMARY: I hypothesize that economic equality should facilitate the protection of biodiversity, but that the strength of this effect will depend on the scale of analysis.

Boyd (1988) argued that while different moral goods conflict in some cases, they mostly reinforce one another. He called this phenomenon the "homeostatic unity" of the good. In this contribution, I apply this idea to the human economy and the global ecosystem of which it is a part. I hypothesize that achieving greater economic equality would facilitate efforts to protect biodiversity. Among other benefits, this "upward" effect of part (economy) on whole (ecosystem) would help to sustain "downward" effects of biodiversity on human welfare, e.g., through diversity's contribution to stability and "ecosystem services" (cf. Mikkelson 2004).

Current forms of economic growth have produced tremendous inequalities in the distribution of resources, both among human beings and between humans and other species. Humans now appropriate approximately 40% of the biomass produced on land by photosynthesis, that was formerly consumed by other organisms (Vitousek et al. 1997). We have granted other species full respite from human economic activity on only about 5% of the Earth's land surface. And expensive, extensive road-building projects are driving human colonization of the rest (Myers and Kent 2001). As a result, we now threaten to kill off the majority of other species within the next 1,000 years (Rosenzweig 2003).

Basic principles of intra- and inter-specific justice entail that the above-mentioned inequalities would only be defensible if they were necessary to fulfill vital human needs, especially those of the poor (Rawls 1971, Naess 1988). But several lines of argument and evidence indicate that re-distributing wealth within and among human societies would fulfill those needs more effectively than economic growth does (Mill 1848, Sen 1993, Kawachi and Kennedy 2002). And a "stationary" or "steady-state" economy would go some way toward reducing the negative impacts of humans on biodiversity.

Besides reducing the pressure to grow the economy, I hypothesize that re-distributing wealth would advance the cause of protecting biodiversity in another way. We might expect demand for nature preservation to vary with individual wealth just as several measures of health do: increasing, but diminishing, returns. In other words, the amount that an individual is willing to pay for nature protection no doubt increases with his or her wealth. But if each additional dollar or euro of wealth results in a smaller and smaller increase in the amount the individual is willing to pay, then, other things being equal, re-distribution should increase the demand from the formerly-poor, more than it decreases the demand from the formerly-rich. As a result, the aggregate demand for nature protection should increase.

The empirical challenge will be to marshal evidence for and/or against the hypothesized relationship between economic equality and nature preservation, and mechanisms for such a relationship. Furthermore, since the relationship between economic equality and health depend on the scale at which inequality is measured (e.g., cities vs. states or provinces vs. entire countries; Ross 2004), we should not be surprised if the strength, and perhaps even the direction, of the relationship between economic equality and demand for biodiversity protection also depends on scale.

A metapopulation approach to farmer seed systems: Methodology for agricultural biodiversity conservation policy

Eric Van Dusen, UC Berkeley,

SUMMARY: A metapopulation concept from population biology is applied to farmer seed systems in an attempt to link social and economic factors to ecological processes.

This work is being developed in order to provide methodology and empirical approaches for understanding farmer seed systems in developing countries. A meta-population is composed of a number of nodes of smaller populations inhabiting ecological niches connected by migration and colonization. A farmer seed system is composed of a series of farmers and the different ways in which genetic material is exchanged and moves between farmers, as well as the ways that genetic material is selected and shaped by each farmer's behaviour.

A starting point for this approach is to improve the understanding of farmer seed systems, utilizing information on farmer management of seeds to move from the scale of an individual farmer to the level of a local population, composed of a region of farmers interacting through seed exchange. This population approach to seed system is a needed policy tool in three areas: 1) the in situ and on farm conservation of agricultural biodiversity, 2) modelling the possible impacts of the release of GM crops in areas of high genetic diversity, and 3) improving access to genetic materials where institutional and environmental efforts have limited success in the past. In all of these areas there are policy needs for answers to questions on how social and economic factors impact the genetic material in farmers fields, and how market and institutional forces shape outcomes at both local and regional scales.

The empirical application, a case study from rural Mexico, explores social, economic, and institutional factors that can influence the outcomes of the seed system. Statistical and econometric examples of how this approach can be applied are provided in order to illustrate the different possible components of a seed system model. Statistical approaches are presented for geneflow, effective population size, seed source and seed replacement. Regression results are presented for the age of seed lots held by farmers with respect to agro-ecological, social and economic variables. Age of household head, area planted, overall system diversity, and migration are found to be weakly linked to the age of seeds planted by farmers. These are compared to results for the diversity of the farmers fields to link different types of population effects.

The next methodological step is to take parameters derived from household level data, and to construct a simulation model to get at inter-temporal impacts of different conservation scenarios. The application to crop metapopulation would not be limited to how to conserve, but would similarly strive to provide the ability to forecast population and genetic outcomes for changes in the environment, in this case the economic and social variables affecting farmer behaviour. Finally, the model will hopefully be able to incorporate some stochastic components, to be able to add the dimension of risk to policy information based on costs and benefits.

Ways of defining an appropriate measurement for ecosystem services

Andrea Gonczlik, ELTE University, Budapest, Hungary

SUMMARY: The aim of this work is to provide alternatives to build up connections between ecosystems and the quality of the services they provide.

"Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life. They maintain biodiversity and the production of ecosystem goods". (Daily, 1997). These processes can only be called ecosystem services if they are proved to be useful for us humans or if they add to our lives in a positive way. Ecosystem services can affect us on the level of individuals, community and even as a civilization. However, in numerous cases these processes are the same as the metabolism of certain species, more often they are carried out by whole communities or through the connection of populations. Therefore, many services strongly depend on the biodiversity of ecosystems, while others can be provided irrespective of how many species there are in the community. Services are rather along a continuum of diversity dependence which may spread in a wide range.

Furthermore, in connection with services it is very rare that the contributions of certain species can simply be added up. (Chapin et al, 2000.) The interaction among species of a community can change material and energy flow directly or indirectly, making a contribution to certain services. Within a community, biodiversity does not necessarily mean better functioning of services. On the other hand, being more diverse a community has the ability to provide better quality and wider-scale services.

The links between the diversity of ecosystems and their functioning still remains controversial. It has become clear that species richness does not always show positive correlation with the quality of services. The diversity of populations is a better indicator of ecosystem functioning, although it is still not a proper measurement of the quality of ecosystem services. The concept of functional diversity provides us the possibility to make a connection between ecosystem services and the composition of ecosystems. "Functional diversity is the value and range of functional traits of the organisms present in a given ecosystem. Functional traits are the characteristics of an organism that are considered relevant to its response to the environment and its effects on ecosystem functioning." (Tilman 2001 cit. Díaz & Cabido, 2001). Service-providing unit (SPU) is an even more specific tool in our hand to describe the linkage between ecosystems and the quality of services that they provide (Luck et al, 2003.)

After having identified services, we should concentrate on defining what groups of organisms provide them. Owing to the complexity of ecosystem services and their linkages, it is not predictable what effects composition change in ecosystems can cause in connection with their functioning.

Re: Ways of defining an appropriate measurement for ecosystem services

Sarah Goslee, USDA-ARS, USA

I'm pleased to see you make the points that:
a) ecosystem function/services are not strictly related to species richness; and
b) ecosystem functions are not necessarily additive.

I'm working in managed grasslands of the Northeastern United States (primarily dairy pastures), and our challenge is to understand how species with particular functional attributes combine to create a stable, productive, stress-resistant pastures. We have built up a body of knowledge on how important forage species function individually, and are working from there to develop an understanding of community function. Most importantly, I wish to understand well enough to predict particular functions for a given set of species. What functions are additive? multiplicative? subtractive? and under what circumstances?

This is a challenge that is just beginning to be addressed, and ties into measurement of functional rather than species diversity, and into community assembly as well.

Spatial scales, stakeholders and the valuation of ecosystem services

Lars Hein (Environmental Systems Analysis Group) and **Ekko van Ierland** (environmental economics and natural resources group), Wageningen University, the Netherlands

SUMMARY: Starting in the late 1960s, there has been a growing interest in the analysis and valuation of the multiple benefits provided by ecosystems. Nevertheless, to date, relatively little elaboration of the scales of ecosystem services has taken place, and how this affects the interests of stakeholders in ecosystem management (Tacconi 2000; Turner et al., 2000; Millennium Ecosystem Assessment, 2003; Turner et al., 2003).

In this paper we analyze the spatial scales at which ecosystem services are supplied, and the implications of these scales for the values attached to ecosystem services by different stakeholders. We provide a framework for the consideration of scales in ecosystem services valuation, and we provide a case study for the De Wieden wetland in the Netherlands. De Weiden is a lowland peat ecosystem, comprising around 5500 ha. The case study comprises an assessment of the scales at which four services supplied by the wetland (fisheries, reed cutting, recreation, and nature conservation) are supplied to stakeholders. The De Wieden case study is based upon fieldwork and interviews with all major stakeholders of the area, conducted in the period January - September 2003.

The scales distinguished in the framework are presented in figure 1. For the De Wieden wetland, the total value generated by the four selected ecosystem services amounts to 830 euro/ha/year, about double the value generated by surrounding agricultural land. Furthermore, it appears that reed cutting and fisheries are only important at the municipal scale, recreation is most relevant at the municipal and provincial scale, and nature conservation is important in particular at the national and international level. Our analysis shows that stakeholders at different spatial scales can have very different interests in ecosystem services, and we argue that it is highly important to consider the scales of ecosystem services when valuation of services is applied to support the formulation or implementation of ecosystem management plans.

Integrating Ecological and Social Scales: Summary Week 1

György Pataki and **András Lányi**, Hungarian Academy of Sciences and **Ekko van Ierland**, Wageningen University, The Netherlands, Session IV Chairs.

Session IV has enjoyed very diverse contributions, including topics ranging from the morality of place, the relationship between social inequalities and biodiversity loss, through linking the metapopulation concept of population biology to farmers seed system, to the possible connections between biodiversity and ecosystem services, and scales in ecosystem services valuation.

The following key points were raised by the contributors:

- The relation between cultural and biotic context in spatial scales:

An environmental ethic stems from the morality of place; locality and morality are intertwined. Without attaching moral significance of creating and using places, one cannot understand how the diversity of places and, relatedly, the diversity of cultural patterns and practices relate to their respective biotic context. Biodiversity loss is closely connected to the spreading phenomenon of placelessness.

- The role of economic equity and social equality in nature conservation.

Preserving biodiversity at different scales can be furthered through policies that reduce the many types of social inequalities characterising most of our present societies. It needs to be pointed out what are the specific mechanisms responsible for such a relationship between biodiversity and social equality. Particular attention might be paid to how this relationship depends on the scale at which inequality is measured.

- Applying the concept of metapopulation in modelling on-farm conservation of agricultural biodiversity:
- The link between ecosystems characteristics and the services provided:

Ecosystem services are important for us, as individuals, for our social communities, as well as for whole civilizations. Though some of the ecosystem services are independent of biodiversity, many of them are dependent upon the biodiversity of ecosystems. The most promising ways to connect biodiversity to ecosystem services are through the concepts of functional diversity and service providing units (SPUs). Thus, research may focus on identifying ecosystem services and defining what groups of organisms provide them.

- Spatial scales, stakeholders and the valuation of ecosystem services:

The importance of scale in the economic valuation of ecosystem services has not been given the prominence it deserves. Ecosystem services provided at different ecological scales (ranging from the biosphere through landscape to plant) affect the particular interests of different stakeholders engaged in ecosystem management. Consequently, exploring the values attached to ecosystem services should take into account that human-ecosystem interactions reach out to different ecological and social-institutional scales.

Next week, keynote contributions will address further methodological issues, such as the possibilities of network analysis to enhance our awareness of issues of scale; social psychological models related to nature conservation; the role of traditional ecological knowledge and community management of natural resources. We hope a more lively discussion will take place next week around the contribution of social sciences to biodiversity issues.

Diversity: scale, hierarchy and function

Ferenc Jordán, Institute of Ecology and Botany, Hungarian Academy of Sciences, Vácrátót, Hungary

In order to efficiently managing diversity, obviously, it is useful to understand the multiple faces of this flagword. Biodiversity is not a very deep and useful concept if only the number of coexisting species is understood (species richness). However, it is a very important and basic feature of biological systems if carefully defined and used. For example, if the behaviour of an ecological community is to be understood, we have to assess its diversity from at least three interrelated viewpoints: (1) the organisational scale at the community is described, i.e. the aggregation of entities into larger functional units, (2) the hierarchy of diversity by means of genetic, phenotypic and species-level diversity, e.g. a monoculture still can perform well if genetically very polimorphic (of course, not typical in agriculture), and (3) functional diversity as considering the roles species play, i.e. identifying keystone vs redundant species. The network analysis of an interaction structure may be an elementary tool for unifying these viewpoints.

While constructing a helpful network that helps to understand the behaviour of the ecological system, these features of diversity are to be considered. Then, network analysis can help in evaluating the role and the nature of diversity within the studied community. A rarity rank of the coexisting species is useful for administrative purposes and easy decisions but has no ecological relevance (of course much more to taxonomists). A functional red list might include abundant species as well, as soon as the importance of species is not evaluated one by one but within a network context for each. Interesting parallels appear in recent sociological analyses: recent techniques of social network analysis have shown that the key players of human groups are not the obvious „stars”. Instead, even with only a couple of links a person can well be highly important, for example, if connecting large cliques. Striking similarities are supposed to exist between the two kinds of systems. In the architecture of ecological communities, similar techniques can help identifying the so-called topological keystone species. Depending on whether we are to understand our society or only a small human group (e.g. a classroom), and whether we are to understand the biosphere or a host-parasitoid community, the parallels can be stronger or weaker. But, basically, studying social communities can be useful for understanding ecological ones and vice versa. The behaviour of both species and persons are constrained by the community they live in.

Re: Diversity: scale, hierarchy and function

Balint Balazs, Institute of Environmental and Landscape Management, St. Istvan University, Godollo, Hungary

Network analysis (NA) offers useful analytical tools to understand and explain the behaviour of social or ecological communities. The units of the analysis are actors in a community (being species or persons) that are constrained or empowered by the network they live in. The network comprises of actors, units or entities and their community relations thus the network analysis starts where the (direct and indirect) relations of units or entities in a certain (ecological or social) community cause some distinctive behaviour, process of that community. NA aims at identifying those interactions of entities or group of entities that explain something of the community characteristics, attributes, processes and behaviours, being social or ecological communities.

NA methodology can help in understanding and assessing the role and nature of diversity within a studied social or ecological community by considering the following questions:

- What is the unit of analysis, what are the nodes or actors in the network? (In an ecological network these can be species, organisations, ecological events, phenomena, functions or positions in an organisation or in a community)
- What interactions and relations of a community are to be understood and interpreted? (In social science the typical relations are exchange, kinship, communicative, affective, instrumental, power relations. What would be the interspecific themes of an ecological NA?)
- How do the graph properties enable us to understand behaviour of a social system or an ecosystem?
- What are the boundaries of the community? Who will tell who is a member of the network? (In case of ecological network it is easier, because the researcher will identify the community; social scientists are not so lucky: the actors of the social community themselves will claim the right to say who is part of the network.)
- In the light of the relational patterns in a network (density, intensity of relations) what are the central or cohesive (keystone) roles and what are marginal (redundant) roles?
- Based on the structural similarities (or similar set of relations), what positions can be defined in a community?
- The network is only interesting if certain community actions, behaviours, processes (as dependent variables) can be explained by these network features. How can NA help to contextualise the behaviour of a community?
- How do the organisational, hierarchical, and functional features of a network interrelate? (What is the analytical difference between the functional, organizational, hierarchical perspectives of an ecosystem?)

On the problematic 'political will' to conserve

Erling Berge, Norwegian University of Science and Technology, Trondheim, Norway

The discussion of conservation measures seems to revolve around protected areas and how many and big they have to be. Some have doubts that there ever will be many and large enough protected areas. They are right to worry. In democratic polities there are strict limits on the kinds of instruments politicians are allowed to use in exercising political will. Today it is not conceivable to declare an area for protected and then shut out or requiring non-development from the local people like it was done for example in the Serengeti National Park. Conservation of biodiversity will become increasingly difficult if protection is the only instrument known to deliver better conditions for ecosystems.

If biodiversity needs better conditions they have to be created around where humans live and work. To do this, politicians need to be able to judge the relative impact on biodiversity from several reasonable courses of action. Do we know enough to predict the true impact of a particular decision net of all confounding factors (that is all factors not part of the political decision)? If the decision to create a National Park in an area increases the number of tourists by 20% while the traditional use of the area by farmers dwindle to nothing, what is the effect on the ecosystem net of confounding factors? Will it be an improvement for our ecosphere if we were able to increase the proportion of cars running on electricity to 10%? What is better for biodiversity: building compact high-rise cities or land extensive suburban type cities? Both at small and large scales such questions require many and consistent decisions on many levels ranging from the person deciding to buy a car or landowner deciding to build to governments shaping taxes and regulations to encourage desired behaviour. To manage nature politicians need to know the net effects on ecosystems of their decisions. They have to be able to consistently choose the ecologically better of two options, not by guesswork but based on scientific knowledge. Today this knowledge does not exist.

Politics has traditionally been developed on a trial and error basis. Only during the last 50-100 years has policy development based on scientific knowledge been tried, and, admittedly, on balance, not with any remarkable success. But there is no turning back. Social and environmental change occurs at a high and maybe increasing frequency. The environmental problems have to find their solution more quickly than the ordinary trial and error approach can promise. But to my knowledge there is no large research program directed at the problem of determining the "subtle" effects of humans on ecosystem (Russell 1993).

In the world of social science a quantitative investigation of the impact of politically manipulated variables on ecosystems will be a mega project rivalling in cost anything known to this writer. But is it worth doing? Will it matter? Before going into this kind of mega project we should step back a bit and think hard on the question of whether a quantitative approach really will help us determine net causal connections. And if cannot, can it still be worth doing? Causality is a difficult concept. In order to establish causal connections the basic requirement is that other things be equal, and that we are able to include all relevant variables. This amounts to a complete list of all initial and boundary conditions of the system we study. If this requirement is met, we are assured that our estimates, within the sampling error, are true estimates of the impact of causal forces. We are confident that the ecosystem will respond to equal quantities of impacts in the same way, most times. Hence, we are able to predict outcomes with a known uncertainty and able to advice on changes in policy.

However, there are reasonable arguments that the assumption of a complete listing of all initial and boundary conditions relevant for a study of system change is untenable not only for human societies, but also for biospheres. Kauffman (2000) argues this rather convincingly. He first conjectures that it probably is theoretically impossible to state the initial and boundary conditions for a biosphere. However, he argues, even if we grant that there perhaps is a theoretical possibility, it probably is practically impossible within the lifetime of the universe to enumerate all initial and boundary conditions relevant for the evolution of a biosphere. And moreover, the practical problems are of such a nature that it also is impossible to establish the distributions required for a statistical study of the possible outcomes.

The conjecture sounds familiar for a sociologist. The debates about research methods and goals for research in social science revolve around the question of predictability. What we are most interested in, social change and innovation are inherently unpredictable. On the other hand, most of human activity is rather routine and repetitive. People do react in predictable ways to changes in their physical and social environment. Hence local and short term predictions are feasible. But will shaping of local and short term behaviour be sufficient for conservation purposes?

One reason for the practical impossibility of stating initial and boundary conditions completely is that for every stimuli and every level of stimuli there are vastly more possible responses than what in reality can happen (in any finite lifetime of the universe), and therefore be observed. With no practical way of listing the state space of our problem analytically, life will always have a potential for surprising us. Life is inherently unpredictable. Yet, it is not chaotic. In hindsight, we do see the paths taken and the causes forcing the development.

If we have to abandon the ambition to generalize, we will at least be able to establish empirical connections with some validity in the short run and for the areas studied. How good we are at selecting relevant variables will determine how good our predictions are, and for how long they will be valid. As long as the goal of protecting biodiversity is clear, this will be a vastly better guide to policy than ordinary trial and error. However, it also means that results are not guarantied. We need to supplement any policy intervention with a learning program. Every change in policy should be viewed as an experiment from which we can learn. Today only occasionally there are linked relevant policy variables to observations of biodiversity. Data to start the slow process of accumulating knowledge about ecosystem responses to policy decisions do not accumulate.

It was noted above that the choice of policy instruments or policy variables are strictly circumscribed in democratic polities. The debate around differences between democratic polities and other kinds of governance systems may not be conclusive, but there are some strong indications that some kinds of policy instruments are impossible to use regularly in democratic societies. Among these are all policies that rely on physical coercion of large groups of people, or large-scale takings of established property. In general, it is conjectured that systematic violations of human rights will generate political and social backlashes that negate any possibly laudable goal one wants to further by such policies. This means we have to put aside all grand revolutionary solutions. In democracies choice of policy instruments need a foundation in the values and opinions of the public in order to have a beneficial impact. What politicians actually can use are the subtle tools of ordinary democratic political action, the piecemeal and small scale measures. They need advice on how to fashion taxes and subsidies, and on how to formulate the marginal changes in legislation that directs behavior towards sustainable use. Politicians need evidence of how changes in priorities of land use planning can further sustainable resource usage. They need a map of which marginal shifts in values should be reinforced, and not least, they need knowledge about the internal consistency of various policy measures. Political rhetoric and public information campaigns have their place in biodiversity policy. But they work only if they are integrated with institutional changes and are fashioned to reinforce these.

The conclusion from this line of argument is that in order to further the goal of sustainable use of biodiversity we need to know the direction and size of the impact from a change in a policy variable and how the impact changes from context to context. Before we can know this, the systematic collection of data has to commence. From there on case studies will accumulate. Cumulative analysis of quantitative case studies may be the only feasible way of approaching the problem of democratic biodiversity policy. There is a long way to go, but ultimately political will comes as much from the ability to do something as from wanting the thing done. Until a better ability to give knowledge-based advice exists we cannot lament the lack of political will.

World problems in our world views: Can individuals do anything?

Mózes Szekely, Eötvös Loránd University of Sciences, Institute of Psychology, Hungary.

SUMMARY: The main questions facing humanity are extensively known in Hungarian society, there are strong-minded and consonant beliefs in their significance, and the majority agree that the individuals can contribute to the reduction of these problems.

Most of us have a good knowledge about the great challenges of humanity. We are well aware of the destructions of wars and epidemics throughout history, we are informed about the agony of famine and overpopulation from the news, we can find some exemplars of contradictory coexistence of poverty and thriftless consumption in our neighbourhood, and we run across notices and warnings about environmental pollution and abuse day by day. Contrary to Kofi Annan's words – meeting people's needs – to call for the participation of the many actors including scientific associations, it is really difficult to trace any research engaged in all of these global issues at the same time. In the case of environment, as one of the most investigated field, Stuart Oskamp the eminent social psychologist called on: "In view of the seriousness and imminence of the environmental threats, many – ideally, all (sic!) social scientists should be focusing more of their work on avoiding or surmounting them" (Oskamp, 1995).

In 2000 we conducted a survey interviewing 1000 persons aged 18 and above, a representative sample of Hungary's adult population regarding age, sex, residency and education. Our results show that at the dawn of the new millennium people have time to think about worldwide questions beside completing their daily routines. More than 95 percent of the sample can mention at least one global problem – two-thirds of them know more. For the open-ended questions we got unexpectedly various answers with fascinatingly diverse subjects. With this colourful palette of concepts we can sketch a detailed map of world-problems in the public opinion, and disclose the relations between them.

In the public mind there is a multifaceted, but homogeneous system of concepts outlined around the expression of 'global problems' (see Figure 1). First, the arrangement of the general terms of serious phenomena vary by different aspects: certain things are important at world scale, others for our country specifically, and there are others taking the lead when it comes to the problems to be solved in the next century. Second, the complex events both close to us and far from us are categorized or classified similarly, and moreover, the ranking of importance of nearly all socio-demographical groups are alike.

The results challenge the popular view that the individuals have the feeling of absolute inability to do anything about the world's problems (see Figure 2). Two third of the participants have a different opinion, and mention concrete single actions, with which the individuals can support the solutions of questions. The evidence gathered from the representative sample shows, that Hungarian adults have strong commitment to main global problems. Nine out of ten are willing to contribute to the resolutions, and more than two (out of ten) would engage in this at a daily level on low payment.

Besides people in general, the researchers also have to keep the big problems of humanity in mind. Handbooks and textbooks of social sciences have to say plenty about the global and challenging phenomena. We can cite Cooperrider and Pasmore (1991), who called for "a science whose primary task is not to »capture« empirical reality, but to study that which has not yet occurred, that is creating, alternative conceptions of the future". As the American sociologist, Joe Feagin (2001) shaped: "more sociologists should engage in the study of alternative social future", and they "need to think deeply and imaginatively about sustainable social futures and to aid in building better human societies".

The proposal of Kofi Annan in the Millennium Summit of United Nations has the same conclusions as our survey: the main problems are poverty, wars and environment. "Two are founding aims of the United Nations whose achievement eludes us still: freedom from want, and freedom from fear. No one dreamed at the time the Charter was written, that the third – leaving to successor generations an environmentally sustainable future – would emerge as one of the most daunting challenges of all.” (Annan, 2000)

Figure 1. The five most important world problems by a representative sample of Hungary’s adult population/frequencies in open-ended questions (N=984) and average ranks in closed list of problems (N=989; the best possible value is 1.00); the significantly different categories (by Cochran’s Q test and Friedman’s chi-squared test; p<0.001) are represented with different colours or markers/

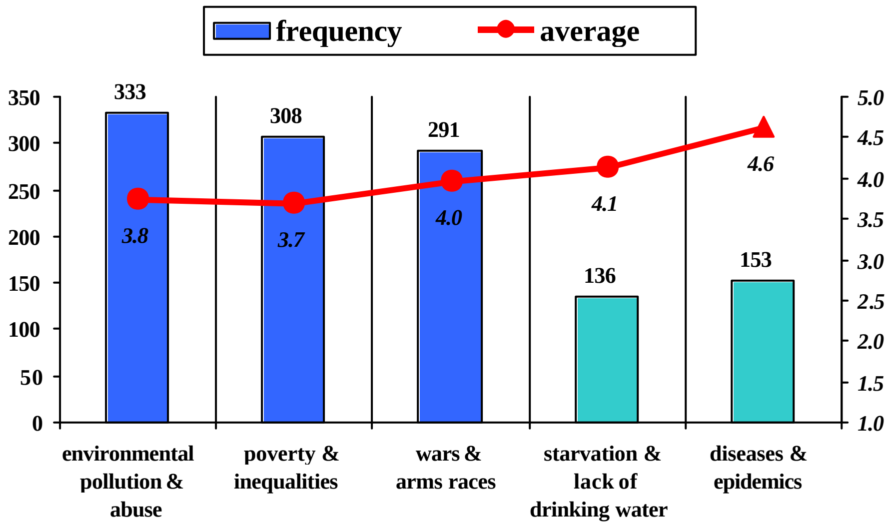
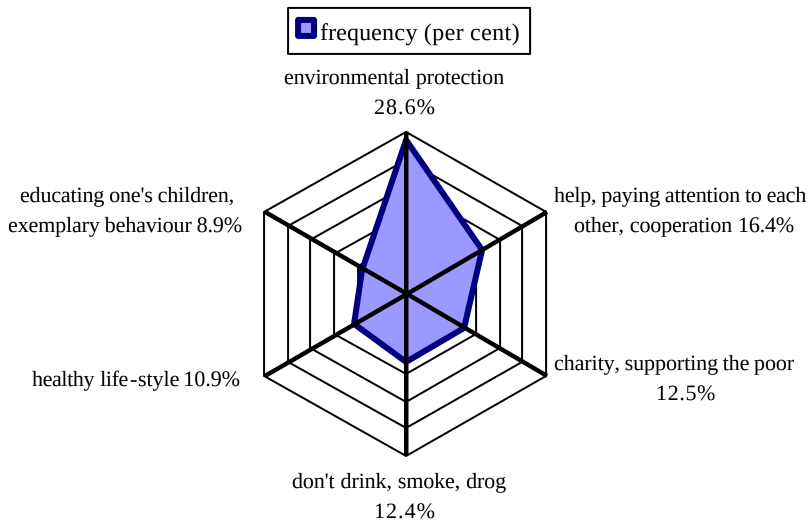


Figure 2. The six most frequently mentioned individual actions by a representative sample of Hungarian adults/frequencies in open-ended questions (N=870, further 116 person answered “don’t know”); the 3-5th group of categories aren’t differ significantly (by Cochran’s Q test, p=0.509)/



The social-psychological dimension of biodiversity conservation and management

Susanne Stoll-Kleemann, Humboldt University of Berlin, GoBi (Governance of Biodiversity) - Research Group, Institute of Agricultural Economics and Social Sciences

SUMMARY: This short keynote contribution addresses and explains how far deeply rooted social-psychological processes are at work in shaping attitudes, outlooks and behaviour towards biodiversity conservation and management.

Firstly, emotional drivers lead to negative perceptions and experiences regarding biodiversity conservation: Residents and land users perceive conservation regulations as restricting personal rights. In this context, Brehm's Theory of Psychological Reactance provides a useful explanation for this phenomenon. This theory states that reactance arises when personal rights to decide and act are threatened, reduced, or eliminated – for example via regulations, prohibitions and controls (Brehm 1966). For example in protected areas people feel restricted in a lot different ways such as their individual preferences (e.g. in their leisure activities, like camping or fishing), in their way of using land (e.g. agriculture, forestry, or hunting), and in their freedom of pursuing their professional activities without co-ordinating with conservationists. Conservation officials also experience emotional underpinning to their management behaviour, e.g. is the fear of losing too much of the core conservation mission when confronted with the demand to include local interests (Stoll-Kleemann 2001 a & b).

Secondly, cultural drivers such as the challenge of traditional values of local people by conservationists play an important role. Conservationists often insufficiently take into account the extent to which biodiversity management interferes with customary behaviours of the local population, and especially, how far that apparent imposition affects their values. Modifications of familiar landscapes and different ways of land use strengthen the dislike of biodiversity conservation (Stoll-Kleemann 2001a & b). If, for instance, the "wilderness concept" means that cultivated forests are to be allowed to evolve into near-nature (virgin) forests without human interference, the local population, which has long been used to exploit and cultivate its forests, perceives such areas left to themselves as "untidy" and contradicting their traditional landscape values (ibid). Local people also tend to overcome any dissonance they may feel about not being custodians of nature (as challenged by conservationists) by justifying their traditional practices as being beneficial for wildlife.

Furthermore, group processes encouraging social identity such as internal bonding processes within social groups (e.g. "conservationists" or "farmers") are an important explanation why conflicts arise while implementing biodiversity strategies (for details on Social Identity Theory, group processes and their role for biodiversity conservation please see Stoll-Kleemann 2001a & b).

How can we use these findings to devise more sensitive biodiversity management processes? A concrete approach to address the feelings of restricted personal and professional freedoms of those directly concerned and affected are the use of sensitive participatory procedures. More egalitarian and network-based communication among all parties may well increase acceptance of biodiversity conservation. Participatory approaches also help to include local particularities, such as different social values and particular cultural norms, into decision-making, and hence avoid potential (negative) consequences of management, of which professional conservationists have not initially been aware (Stoll-Kleemann 2004; Stoll-Kleemann & O`Riordan 2002).

More social psychological oriented research is needed to clarify the motivations underlying the management of biodiversity conservation because this kind of research particularly reveals how various social interests shape prejudices, bond in alliances, and create the scope either for conflict or conciliation.

Re: The Social-psychological Dimension of Biodiversity Conservation and Management

Rainer Muessner, CIMAR- Centre for Marine and Environmental research, Porto, Portugal

Susanne Stoll-Kleemann described in her keynote contribution how deep-rooted social-psychological processes are shaping attitudes towards biodiversity conservation and management. The resulting attitudes she refers to, have led to the fact that in densely populated cultural landscape like most of Europe are, the capacity to implement further protected areas is close to its limits and at least in Western Europe protected areas are likely to level off on the current stage (EEA 2003a, IUCN 2003a+b). The designation of the NATURA 2000 areas is probably the last "boom" of protected areas that will be realised in the near future. Concerning protected areas the trend in Europe is not comparable to the clearly increasing trend globally (see IUCN presentation at the Paris Biodiversity and Governance meeting). That means for biodiversity conservation in Europe three main options are left open:

1. Improve the biodiversity protection/ conservation strategies outside protected areas
2. Improve the effectiveness and efficiency of established PAs rather than calling for new PAs
3. Last but not least try to tackle these "deep-rooted social-psychological processes" that result in negative attitudes towards conservation in many societal groups.

Especially the last option is still not very popular in conservation research due to the fact that it calls definitely for interdisciplinary research with social scientists (with all the well known difficulties and problems of interdisciplinarity) and secondly because it means scientists have to leave the "ivory tower" and try to perceive the wider public about the values of biodiversity and the need to protect it (although this is not research in the narrow sense, it can give some interesting inside views that help to design the next proposal). Currently most scientist leave this later activity mainly for the NGOs.

In the second part of my reply I like to add something to Susanne's comment: "Residents and land users perceive conservation regulations as restricting personal rights". This statement reflects probably the conservation reality and can be confirmed by many sociology studies about the acceptance of conservation activities in Europe and elsewhere.

Even so, it is important to mention that conservation is not the only societal movement that asks for restriction of personal rights for the benefit of the whole society. It seems we are calling for something extraordinary/ unbelievable. But this is not the case. A speed limit on the highway is as well restricting my personal rights to test the technical speed limit of my car on the highway. Every municipality or local development plan restricts my personal rights to build a skyscraper on my own property, but everybody seems to accept this.

People seemed to be very used to blame conservationists for restricting there personal rights on the properties, but what about the personal or social responsibility I have as land owner. These responsibilities are the counterparts of the rights someone seems to have and these responsibilities are much too often neglected in discussions with land owners about the protection of the environment (including Biodiversity). Everybody knows her/his rights as landowner, but seems to forget about some old fashion things like Aldous Leopold´s land ethics which calls for responsibility.

Multi-disciplinarity and biodiversity as a boundary object

Martin Sharman, Biodiversity Sector, Natural Resources Management and Services, European Commission Directorate General for Research

SUMMARY: Biodiversity means different things to different communities. Co-operation between disciplines requires acceptance that biodiversity is big enough for both of us.

As Ekko van Ierland stated, "the relevant aspects and perspectives on a scientific problem vary largely between the gamma and beta sciences". There is indeed an issue of scale involved in approaching biodiversity in a multi-disciplinary way, and it's not just between the "social" and "natural" sciences.

In the Amsterdam meeting of the EPBRS, Dr C.L. Kwa (U. Amsterdam) told us that biodiversity is a boundary object. But what is a boundary object? Susan Leigh Star and James Griesemer introduced the concept in 1989 to describe information used in different ways by different communities. Since then it has been defined in many ways: the "boundary object" is itself a boundary object, but Dr Kwa used it to mean a concept that is shared by different communities, but whose details differ from community to community.

So what "is" biodiversity? Biodiversity is not just what it says in the CBD definition. It goes beyond, and is more far reaching and ambiguous, than that already rather complicated formula. The definition belongs among the set of ideas, beliefs, feelings, objects, relationships between objects, documents and vocabulary that allows people from different backgrounds and perspectives to agree that they are working towards a common understanding of biodiversity. Those different perspectives all contribute to a working arrangement between very different ways of thinking and doing business.

The term, and the concept, allow people to co-operate. Geneticists, taxonomists, sociologists, ecologists, modellers, economists, bio-chemists, conservationists, policy makers, TV documentary makers and many other groups interpret "biodiversity" differently. But they all agree they're talking about biodiversity. This encourages us to interact across communities. And as we interact, our perspectives shift and our own interpretation of the concept evolves. Boundary objects are flexible if the person thinking about the object is willing to accept new angles, flux, and even ambiguity.

But boundary objects are resilient, too, since each community will see valuable elements in their own view that are not included in the perspective from another community. People waste effort when they try to get others to agree to their definition of biodiversity. While I may (possibly) be able to explain it, I cannot impose my vision of biodiversity on you. At best you may like some aspect of my vision, and adopt it as part of your own - but there may well be aspects of my vision that you really dislike, or simply do not understand. Thus a geneticist and an economist, for example, will probably never share the same vision of what biodiversity "is" - not for lack of good will, but simply because their backgrounds and training bring them to the boundary object from different points of the compass. What is most important about biodiversity to the geneticist may be something that the economist legitimately regards as a trivial detail from her perspective.

From a philosophical point of view, it should not concern us that we all understand biodiversity differently. This is is a characteristic of a useful boundary object. But the characteristics of a boundary object, particularly the difference in understanding about what is important about biodiversity, is one of the reasons that it is so hard to get agreement on indicators. What are we measuring? That depends on the details, about which different communities tend to have fundamentally different opinions.

As we change scales from a single-discipline view of biodiversity to a multi-disciplinary one, we are obliged, therefore, to change our understanding of biodiversity in a qualitative way. To do so properly we must absorb a new set of concepts and adopt a new vocabulary, which is part of the reason that it is so hard to find effective multi-disciplinary co-operation.

Re: Definitions of Biodiversity

Tim Kitchin, Glasshouse Partnership, London, UK

I see here a clear distinction between big B, 'B'iodiversity (idea 1) - the boundary object - in other words the overarching 'brand' which describes the concept of nature as a dynamic, precious and fragile phenomenon and small b 'b'iodiversity' (idea 2) - a qualitative description of the health of this natural order.

There seems to be reasonable consensus that this second 'b'iodiversity has something to do with the variability, richness and resilience of ecosystems and their component parts.Within 'b'iodiversity' itself though sits the infamous CDB definition, describing biodiversity as an indicator of natural variability. Call this 'b'iodiversity (idea 3).

Much of the measurement debate here seems to revolve around the need to move from measuring idea 3 to better describing idea 2. But what is driving this need is widespread engagement with idea 1. Economists come in through idea 1. The general public come in through idea 1. Governments come in through idea 1.

I suggest that the new language Dr Sharman proposes should exist in the space occupied by idea 2, but also be meaningful within the frames of reference of ideas 1 and 3.

'b'iodiversity essentially describes the health of nature and is inextricably linked to the health of human beings, as a species and as individuals. We are taking nature's vital signs, and in so doing, taking our own.. No single 'b'iodiversity indicator holds any truth, but taken together they paint a clear picture.

There are clear analogies between natural health and human health - with vital organs; with blood pressure; with oxygen peak-flow rates; with white cell count, with recovery rates etc. It feels to me that an overarching framework is required which takes into account the total system. Measuring 'b'iodiversity alone is like trying to assess someone's health by counting their toes.

Contributions of cultural keystone species to social and ecological conservation

Ann Garibaldi, Garibaldi Heritage and Environmental Consulting, Alaska, United States, and **Nancy Turner**, University of Victoria, British Columbia, Canada.

SUMMARY: Identifying and focusing on culturally critical species may strengthen conservation initiatives that seek to address both the social and ecological aspects of an ecosystem.

Certain species –referred to as keystone species – strongly influence the integrity of an ecosystem through their functional and structural contributions; a similar phenomenon is occurs in social systems. We know that the integrity of social systems is inextricably linked to the integrity and functioning of their associated ecosystems. Furthermore, for local and aboriginal peoples worldwide, particular plants and animals play key roles in shaping the characteristics of a culture. We have termed the organisms that serve these special cultural roles as cultural keystone species and define them as the culturally salient species that shape in a major way the cultural identity of a people, as reflected in the fundamental roles these species have in diet, materials, medicine and/or spiritual practices (Garibaldi and Turner 2004; see also Cristancho and Vining 2004). The significance of these plants and animals to particular cultures may be evidenced in their role in narratives and ceremonies, intensity and multiplicity of use, naming and terminology in a language, and use in trading and resources acquisition (for a discussion of cultural keystone indicators see Garibaldi and Turner 2004). Cultural keystone species vary over temporal, geographic, and social scales. Their use and importance are simultaneously affected by season or history, location, and personal, cultural or societal standing.

The maintenance of biodiversity by aboriginal peoples has been well documented; a decline in biodiversity often results in a decline of cultural diversity, and vice versa. Ecosystem changes are mirrored in human cultures. A decrease in both local and regional biodiversity due to such agents as development, invasive species and resource extraction affects not only the availability of culturally important species but also their accessibility to aboriginal and local peoples. Social and ecological systems have co-evolved with an intricate system of checks and balances to produce biologically diverse landscapes. Attention to the observations of aboriginal peoples to changes in such areas as the circumpolar region has illuminated not only important systemic environmental changes, but also information on the resulting cultural adaptations (Krupnik and Jolly 2002).

The utility of the cultural keystone species concept lies in the identification of uniquely critical species that contribute to the maintenance of cultural integrity. A loss of these species result large impacts to the cultures that rely on and identify with them. If we begin our conservation efforts by focusing on cultural keystone species in a given setting, often limited resources are targeted at socially critical species. Furthermore, if local people connect and participate in conservation and restoration practice through a conduit of species that have high importance to them, the conservation endeavors will be much more relevant to them, and consequently, they become more engaged in ecosystem conservation and restoration. We anticipate that linking conservation to cultural concerns will result in a positive feedback loop of increasing effectiveness in maintaining and restoring both human and ecosystem health.

Protected Landscapes: The role of communities in conservation of biodiversity

Jessica Brown, QLF/Atlantic Centre for the Environment (USA) and Protected Landscapes Task Force, IUCN World Commission on Protected Areas

SUMMARY: This paper briefly discusses the role of communities in sustaining and protecting biodiversity in landscapes. There is rich experience world-wide with the protected landscape approach, which offers insight into the linkages between biological diversity, cultural practices and other social factors, and provides models for engaging communities in stewardship.

In discussing the role of communities in biodiversity conservation at the landscape-scale, let me begin with the idea of "landscape," since by its very nature it integrates ecological and social factors. Landscapes, the places where people and nature meet, are often rich in both biological and cultural diversity.

Community involvement and inclusive approaches to conservation are central to an emerging new paradigm for protected areas world-wide as summarized in figure I (Phillips 2003) It sets the stage for stewardship by those closest to the resource. There is growing recognition that conservation must tap the wealth of knowledge, traditional management systems, innovations and love of place that indigenous and local communities bring to the landscapes they inhabit.

This is particularly true for protected landscapes. Protected landscapes are protected areas based on the interactions of people and nature over time. Living examples of cultural heritage, these landscapes are rich in biological diversity and other natural values not in spite of but rather because of the presence of people. It follows that their future relies on sustaining people's relationship to the land and its resources. It requires an approach that is interdisciplinary, inclusive, and that engages people and communities. (Brown, Mitchell and Beresford 2005). In such an approach, traditional ecological knowledge and management systems are key. The protected landscape approach links conservation of nature and culture, and fosters stewardship by people living in the landscape. While grounded in experience with Category V Protected Landscapes/Seascapes (Category V in the 1994 IUCN Guidelines for Protected Area Management Categories), this approach is broader than a single protected area category or designation. Rather, it relies on different tools and designations to achieve protection, and on an array of processes to sustain people's relationship to the land. Examples include Category V Protected Landscapes, Category VI Protected Areas, UNESCO World Heritage Cultural Landscapes, as well as private protected areas and community-conserved areas. Protected landscapes are often part of a mosaic of protection tools, and can help strengthen linkages between more strictly protected areas and the broader landscape. It is important to stress here that an approach that emphasizes lived-in landscapes should in no way be seen to reduce the importance of strictly protected areas. Rather it is a complementary model – one that is particularly appropriate in settings where biodiversity and cultural practices are linked, and where management must accommodate traditional uses, land ownership patterns and the need to sustain local livelihoods.

One important way that indigenous and local communities conserve biological diversity in the landscapes they inhabit is through community-conserved areas. These encompass the array of strategies that indigenous and local communities have been using for millennia to protect land and natural and cultural resources important to them. Long ignored by governments, and not included in the accounting of official protected areas, community-conserved areas are now receiving growing attention in the protected areas field. These areas, which are found worldwide, can be defined as: modified and natural ecosystems, whether human-influenced or not, and which contain significant biodiversity values, ecological services, and cultural values, that are voluntarily conserved by communities, through customary laws and institutions. (Barrow and Pathak 2005). Examples include sacred groves, watersheds protected for communal water sources, coastal areas protected for fishing, traditional agricultural systems, and areas reserved for grazing and forage by pastoralist peoples. For more on CCAs, see for example, Barrow and Pathak 2005; Borrini-Feyerabend, Kothari and Oviedo, 2005; Jaireth, H. and D. Smyth. 2003).

Though thousands of years old in practice, the community-conserved area model has only recently been gaining broad recognition by practitioners in the fields of protected areas management and biodiversity conservation. Further research would help to deepen our understanding of CCAs and the conditions that will help sustain them. Assessing the biodiversity resulting from these practices, on a case-by-case basis, is an important step.

More generally, two interesting areas for further inquiry include: models for participatory and community-led governance of protected areas, and arrangements to ensure the intellectual property rights of communities' traditional knowledge systems and management practices.

The IUCN Task Force on Protected Landscapes (part of the World Commission on Protected Areas – WCPA) has been researching and documenting case-studies of Protected Landscapes and Seascapes (See: Brown, Mitchell and Beresford, 2005; Beresford 2003; Phillips 2002). There is rich experience world-wide with the protected landscape approach, which offers insight into the linkages between biological diversity, cultural practices and other social factors, and provides models for engaging communities in stewardship.

Figure 1. Contrasting paradigms

Topic	***As it was: protected areas were …***	***As it is becoming: protected areas are …***
Objectives	• Set aside for conservation • Established mainly for spectacular wildlife and scenic protection • Managed mainly for visitors and tourists • Valued as wilderness • About protection	• Run also with social and economic objectives • Often set up for scientific, economic and cultural reasons • Managed with local people more in mind • Valued for the cultural importance of so-called "wilderness" • Also about restoration and rehabilitation
Governance	Run by central government	Run by many partners
Local people	• Planned and managed against people • Managed without regard to local opinions	• Run with, for, and in some cases by local people • Managed to meet the needs of local people
Wider context	• Developed separately • Managed as 'islands'	• Planned as part of national, regional and international systems • Developed as 'networks' (strictly protected areas, buffered and linked by green corridors)
Perceptions	• Viewed primarily as a national asset • Viewed only as a national concern	• Viewed also as a community asset • Viewed also as an international concern
Management techniques	• Managed reactively within short timescale	• Managed adaptively in long term perspective

	• Managed in a technocratic way	• Managed with political considerations
Finance	Paid for by taxpayer	Paid for from many sources
Management skills	• Managed by scientists and natural resource experts • Expert led	• Managed by multi-skilled individuals • Drawing on local knowledge

Source: Phillips, Adrian. 2003. Turning Ideas on their Head: The New Paradigm for Protected Areas. In: Jaireth, H. and D. Smyth. Innovative Governance: Indigenous Peoples, Local Communities and Protected Areas. Ane Books, New Delhi, India.

Transboundary Landscapes: Paradigm shift in conservation of biodiversity

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Over the past decades, there have been ample discussions towards more participatory forms of management for effective and sustainable conservation and social justice (Secretariat to CBD 2004). There is strong thought amongst conservationists that community involved participatory conservation at a landscape level should be the approach for sustainability and effective conservation of biodiversity (Vanclay et al. 2001, Sharma and Chettri 2003a, Velazquez et al. 2003, Bawa et al. 2004, Secretariat to CBD 2004).

In spite of decades of conservation efforts, biological diversity throughout the world continues to dwindle. It is experiential that during the last three decades, efforts to conserve biodiversity have gradually begun to shift away from law enforcement and use restrictions towards more participatory approaches emphasizing equitable and sustainable use of natural resources (MacNeely and Miller 1984, Sharma and Chettri 2003b, Borrini-Feyerabend et al 2004). This change in approach is important in the remote rural areas of mountains and developing countries where biodiversity is concentrated, where poverty tends to be pervasive, and where the reach of development programs is often limited. Wells (1992) has stated that beyond the economic principles involved, it is recognized that neither is it politically feasible nor ethically justifiable to attempt to deny the poor for using natural resources without providing them with alternatives. We still need to devise approaches to conserving biodiversity that recognise the dynamism of systems, the dependence of local people on their natural resources and need to build redundancy into our system to conserve biodiversity (Bawa et al. 2004).

Securing the conservation of biodiversity while at the same time promoting sustainable economic development is one of the greatest challenges of our time. Ways to achieving these two goals are becoming the focus of increasing attention, particularly within the conservation and developmental communities. Formal conservation in most countries has, for the last century or more, been treated as the domain of centralized government agencies. Predominant focus has gone to the creation of protected area seen as islands of biodiversity, which need to be protected from the human intervention. More recently, however, there is increasing recognition of the value that local communities can bring to the process of conserving biodiversity, and of the need for a range of conservation types from strict protection to multiple sustainable uses. There is also recognition that protected areas need to be related to their surrounding, and planning process need to go into broader landscape level. This paradigm shift has seen the development and application of management models that are design to integrate conservation and sustainable use (Bennett 2004).

Integrating Ecological and Social Scale: Summary Week 2

György Pataki and **András Lányi**, Hungarian Academy of Sciences and **Ekko van Ierland**, Wageningen University, The Netherlands, Session IV Chairs.

The second week of Session IV planned to discuss issues related to biological and social diversity at the individual and community level. The main message of the contributions, I believe, points to the importance of social justice in biodiversity conservation and management. Many contributions explicitly referred to the deeply political nature of biodiversity issues as they relate to giving voice to people and communities being in marginalised positions in society; involving people and communities in conservation efforts in a strongly democratic way; respecting people's and communities' lived-in cultural experience of particular landscapes. In a sense, Week 2 contributions pushed us to recognise, as some commentators put it, that if there is a global crisis of biodiversity or ecology, as we believe it in the conservation profession, there is, at the same time, a global crisis of justice, too.

Our first keynote contribution, by Ferenc Jordán, highlighted the possibility of a methodological cross-fertilisation between sociology and ecology through the tool of network analysis. In addition to the job network analysis may do by identifying so-called topological keystone species, it convenes the important message that not only the "stars" with many connections in a network may play a key role in a community, be it biological or social, but a marginal position could have an important role in the sense of providing the only connections for groups having no other way of interacting.

The second and third keynote contributions, by Mózes Székely and Susanne Stoll-Kleemann, addressed issues of biodiversity from a social psychological perspective. Previous surveys, conducted by Székely in the Hungarian context, revealed not only the knowledge of people in general about ecological problems, but also a motivation of act, a feeling of some extent of self-efficacy. Stoll-Kleemann's analysis, however, revealed the emotional and cultural drivers, as well as group processes of social identity formation as they provide an explanation for specific barriers to nature conservation in general and biodiversity preservation in particular. Importantly, she, as well as Erling Berge in his contribution, pointed to the need for more politically sensitive biodiversity policies, ones that prefer participatory procedures, more egalitarian and network-based communication, and aware of giving voice to local particularities in decision-making.

The fourth keynote contribution, by Canadian researchers Ann Garibaldi and Nancy Turner, introduced the concept of cultural keystone species. This concept has the clear advantage of merging the biological and social-cultural side of nature conservation in the spirit of a co-evolutionary perspective. The concept of a cultural keystone species point to the fact how a particular ecosystem has her imprints in a particular culture, as well as how a particular culture make sense of the landscape it has co-evolved with. Also, the temporality and spatial nature of biodiveristy issues, both the biological and cultural, may be considered. Garibaldi and Turner emphasise the possibility of creating positive feedback loops between conservation efforts and preserving local cultural diversity.

The final keynote contribution of Week 2, by Jessica Brown, brings a community perspective into the discussion. The "protected landscape approach" she advocates understands landscapes as the meeting point of nature and people, characterised by a richness in both biological and cultural diversity. This perspective points to the importance of seeing nature and culture in their co-evolutionary interrelatedness: the inevitable policy lesson is thus sustaining people's relationship to the land; tapping the wealth of knowledge and love of place residing in local communities. The importance of models for participatory and community-led governance of protected areas are emphasised, as well as of designing arrangements to ensure the rights of traditional ecological knowledge and management systems of aboriginal and indigenous communities.

Space Matters in Conservation: A Spatial Approach to Understanding Local Environmental Knowledge and their Institutions

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SUMMARY: The task of biodiversity conservation is increasingly one of negotiating amongst a variety of actors and interests, many of whom operationalize different knowledges produced at different scales and with different agendas in mind. It is therefore critical for the task of biodiversity conservation to pay close attention to environmental knowledge, how it is produced, and the conflicts arising from different actors making distinct knowledge claims.

Environmental knowledge is not something merely used by resource management institutions, but is something produced, maintained and adapted through the practices of such institutions. It is thus embedded in political, cultural and economic context and its study requires a holistic approach (Berkes, 1999). What I want to draw our attention to is the importance of paying attention to the spatiality of environmental knowledge. By this, I do not mean bounded spatial scale, but the quality of space associated with the production of environmental knowledge, for instance spatial patterns, spatial behaviour and the meanings of the boundaries associated with management regimes. I conduct research in Northern Thailand, but I believe the lessons I have learned there around the importance of space in conservation mechanisms translates to other circumstances. In my field site I found that local environmental knowledges were characteristically spatially complex (see Roth 2004). For instance, environmental practices associated with the use and production of knowledge (such as the tools and techniques used to harvest non-timber forest products, graze livestock, and monitor forest quality) are expressed spatially, not as predictable, fixed and easily categorized patterns of use and management, but rather as seasonally dynamic networks that are dependant upon social relations, political climate and environmental conditions. So different places within a community's territory are accessed differently throughout the year and serve different sectors of the community. Likewise, social institutions associated with environmental management (such as rules of access, enforcement and tenure arrangements) are expressed spatially, not as neatly bounded zones – such as what we witness daily on management maps – but as overlapping, porously bounded zones subject to social relations. For example, households and communities often share areas of land for common uses such as grazing or common activities such as harvesting. Boundaries are often porous to particular people, for particular activities at particular times. (Contrast this with the favoured impermeable boundaries of conservation zones!)

A key point that arose from the research was the issue of spatial flexibility. Because local managers were interacting more directly with the environments they were managing, they had produced knowledge and institutions which could adapt more readily to changing environmental, social and political circumstances. Whereas the demands of centrally managing a large area (e.g making land use legible and management transferable to people without first hand knowledge) meant that managers operating at a larger spatial scale often used less flexible spatial tools (e.g. statically bounded zones.) Conflicts arise when the static spatial tools developed to meet the requirements of centralized management are imposed at the local level, replacing and homogenizing the previously more complex spatial arrangement. Such changes in the spatial arrangement of local environmental practices and management institutions create conflict not only between local communities and the forest service but also amongst households and amongst neighbouring communities. Such conflict is not conducive to the goals of greater integration amongst the various stakeholders of biodiversity conservation. The task of biodiversity conservation requires better integration of various environmental knowledges (understood, not only as bits of information, but including management institutions and practices) and a clearer understanding of their spatial form can aid in such a task. We need to be aware of the different spatiality associated with different scales of management so that spatial complexity can be scaled up when necessary, and local managers can be more aware of why certain spatial form might be useful for certain conservation goals.

Intellectual property rights and biodiversity

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SUMMARY: The Rio Convention on Biodiversity brought linkages between traditional knowledge, intellectual property rights and biodiversity to the international agenda.

Most of crucial regions in biodiversity preservation are situated in the third world countries, many of them being inhabited by culturally heterogeneous groups of indigenous peoples and other traditional communities. Their traditional ecological knowledge and their skill to use a wide variety of biological resources can turn out to be useful to the larger human populations. As an indication of it, there is an intensifying attention to (plant) genetic resources known as bioprospecting. However, traditionally national laws have not assigned any special status to indigenous knowledge systems and know-how – but continuous discussions have recently taken place how their interests would be protected and consequently a number of reforms have been done. In particular, the Rio Convention on Biodiversity (CBD) brought these linkages between traditional knowledge, intellectual property rights (IPRs) and biodiversity to the international agenda.

The standard explanation for the loss of biodiversity pays attention to the institutional conditions under which the exploitation of genetic resources occurs. It is claimed that open accessibility leads to underinvestment and overutilisation. When intellectual property rights are recognized in plant genetic resources, nations and private landowners would have incentives for investing in these resources (see e.g. Mugabe et al. (eds.) 1997; Vogel 1994). As to communities, Gupta (1998) has suggested that "the solution [to the biodiversity loss] lies in establishing intellectual property rights in favor of local communities and individuals [--]." For instance, if plant breeders have exclusive, patent-like rights to crop cultivars they have developed in their laboratories, then we should also accept the idea of "farmers' rights" or "community IPRs" and legally acknowledge the role of the farmers in the development of crop cultivars.

There are a number of conceptual and practical difficulties in this reasoning and in extending IPRs to communities. Is it compatible with the traditional idea of IPRs? If not, should a new idea of intellectual property be developed? What are then intellectual property objects? Who exactly are their owners? Does proprietary ideology match up with indigenous worldviews? Is it just to reward one party, while larger groups of people may have been involved in the development? And finally, does this really result in conservation practices? Some of these problems have been tried to tackle in WIPO (see http://www.wipo.int/tk/en/genetic/index.html) and CBD meetings (see http://www.biodiv.org/programmes/socio-eco/traditional/default.asp).

When we focus on biodiversity conservation, there are two main themes to be distinguished: 1) the conservation of the units of biodiversity; 2) the conservation of knowledge about the use of units of biodiversity. In the case of domesticated and cultivated forms of biodiversity the two themes are intertwined in the sense that if knowledge about the use is lost, the same is likely happen to the unit of biodiversity. This is not the case concerning undomesticated units of biodiversity; when the knowledge disappears, it is likely to be the only loss. Some conservation biologists have suggested that domesticated units of biodiversity are irrelevant to the biodiversity protection. However, it is rather difficult to make distinction between domesticated and undomesticated forms of life in practice. And even if human-dependant units are not a proper focus in conservation biology, they are a matter of great importance to us. If we focus on wild organisms, then we face a conceptual problem how to provide IPR protection to objects and characteristics that are evidently natural. This is clearly in conflict with the traditional requirement of artificiality. One stance is, probably promoted in the CBD, that possession of organisms implies a kind of exclusive control to all of their features.

Alternatives to the exclusive IP system are also evolving. Some of the inspiration comes from information technology debates, for instance some speak of “open-source biology” (http://www.wired.com/news/medtech/0,1286,66289,00.html) and in other case, the model to protect two thousand potato varieties in Andes region reminds me of the copy-left movement (http://www.grain.org/bio-ipr/?id=429 and http://www.gnu.org/copyleft/copyleft.html). The international legislation is very much under development regarding genetic resources. In general, the issue as a whole is extremely complicated and therefore I am rather hesitant to make wide-ranging claims in this short presentation, but what worries me most is the disappearance of biocultural heterogeneity. Perhaps these alternatives, still in need of further development and conceptual analysis, could work better in conservation and not accelerate the loss of biodiversity. Moreover, at least some of these traditional communities prefer them to the exclusive systems, which is an important aspect in moral and political considerations.

Biodiversity as Global Commons and the Limits of the Market

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An important development has occurred in the past decade about how to manage the global environmental commons. According to this new approach, the "tragedy of the global commons" should be avoided through the extensive use of market-based instruments. This can take several forms. First, new markets should be created for environmental goods. For instance, an international marketplace for greenhouse gases has been created, or new intellectual property rights regimes should help biodiversity (embodied in genetic information) to become a market good. Water resources and supply are also being privatised all over the world. Second, environmental conventions should avoid using command-and-control mechanisms and must deal with questions of competitiveness. Voluntary self-regulation of business and "green market forces" (green investment and green consumption) are to be promoted, instead of setting global standards for business.

The greening of the market is certainly to be promoted, and market mechanisms can, obviously, play some role in international environmental measures. However, pushing this logic too far has serious shortcomings. The market in itself can never secure the social and ecological needs of sustainable development. From the social point of view it can be predicted that the market will create new kinds of inequalities related to environmental goods: an enclosure of the commons will take place on a global scale. For instance, local people may be denied to have access to economically valuable resources or areas. From the ecological point of view, it can be predicted that the market will not provide proper and sufficient incentives to protect the global commons. For instance, property rights regimes for genetic resources may create incentives to save some "botanical gardens" in developing countries, but they will certainly not be enough for saving the Amazonian rainforests.

We need a different approach. Justice as fairness should be the guiding principle for international regimes protecting the global commons. In a sense, nations are in a similar situation to the one described by John Rawls as being behind the "veil of ignorance". Nobody can perfectly know the effects of environmental degradation upon her country. Therefore, there is a need and room for a fair agreement. The right of each individual to clean and healthy environment must be secured, on the one hand. International institutions should promote the greatest benefit of the least advantaged, on the other hand. This needs concerted actions of the nations and the construction of a complex institutional setting, instead of relying extensively on individual players and market forces. Property rights regimes must be part of this setting with the aim of helping social and ecological needs to be realized. Two basic questions should be considered when creating property rights regimes: Whose property? What kind of property? Regimes should secure the rights of local communities and protect or promoting small-scale as well as common property systems at both local and global levels (for the latter see the concept of the Common Heritage of Mankind), instead of pushing exclusively for individual property and large-scale properties.

Re: Biodiversity as Global Commons and the Limits of the Market

Hans-Peter Weikard, Environmental Economics and Natural Resources Group, Wageningen University, The Netherlands

The issue of justice has been addressed in a number of contributions to this conference. Indeed is worthwhile to explore what fairness requires in terms of conservation and conservation policies on a global and international scale. I agree with Zsolt Boda on most issues he raises. And I agree that exploring conservation "behind a veil of ignorance" is an attractive option. I would like to point out, however, going along with Sen and Roemer, that Rawls's primary goods approach may not be appropriate. The preferable alternative that emerged in the literature is that fairness should be defined in the space of opportunities. I elaborate this idea below.

I shall call a “conservation policy” every course of action that has an impact on the survival or extinction of species. A conservation policy may be good or bad in terms of loss of species. This notion of a conservation policy covers activities like the protection of a coastal area, the conversion of a rain forest to grassland, the conversion of wetlands into an industrial area, etc. For the purpose of this note each conservation policy is associated with the set of species that survives under this policy. Thus, formally, a conservation policy can be identified with a subset of all currently existing species. A comparison of different conservation policies requires both, a comparison of the values of the subsets of surviving species associated with each policy and a comparison of costs associated with a policy. Since the latter is not the main concern here, I simply assume that one could rely on standard evaluation methods which give a monetary measure of the cost of each conservation policy under consideration. The focus here is on the comparison of the values of sets of species. A ranking of those would be a large step forward towards a rational choice of conservation policies, even if no monetary measure is provided.

In a seminal paper Martin Weitzman (1992) has developed a measure of biodiversity defined for each set of species. The framework I propose is close to Weitzman’s, but it is broader in scope. Weitzman’s diversity measure account for the value of species, while the framework developed here can accommodate both, the value of biodiversity and the value of species.

The task is to establish principles upon which subsets of the set of all currently existing species can be compared. In social choice theory a structurally similar problem has been considered under the heading of ranking opportunity sets. Pioneering literature is by Kannai and Peleg (1984), Sen (1988, 1991), Bossert (1989), Pattanaik and Xu (1990), and Bossert, Pattanaik and Xu (1994). In this literature an opportunity set is offering freedom of choice and it is discussed on which principles a measure of freedom of choice can be based. My claim is that the problem of choice of conservation policy can be described in a similar way. The challenge is to establish principles on which a ranking of conservation policies (i.e. sets of species) can be based. Of course, deep valuation problems are inherent in such approach as in any other. The advantage is that valuations are clear and transparent, and we gain insights in which ways certain values will affect conservation choices. How can rational conservation choices ever be possible without an understanding of the “better or worse”? Admittedly, a lot of work is necessary to put such framework into practical use.

Adaptive science for biodiversity conflict resolution

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SUMMARY: To support biodiversity conflict resolution means to select the assessment and evaluation process according to conflict characteristics, including biological and social issues.

There are many discussions on the preferentiability of specific methods and methodologies in natural and social sciences. Some of them have been led in this e-conference (biodiversity indicators, monetary evaluation etc.). The arguments exchanged are important and certainly advance science, but are they of use when trying to resolve concrete biodiversity-related conflicts (I assume that science can and should be used to help resolving conflicts)?

What are the characteristics of such conflicts? They necessarily deal with a complex issue, both in a natural and a social dimension. Different groups wish to use the same resource (e.g. the same fish species is to be used by fishermen and anglers, preserved by fish preservationists and eaten by a bird which bird watchers and ornithologists want to have around). These groups have different perspectives on the conflict, on its spatial and temporal scale, and on the relevant information available and necessary for the resolution.

The first role of science is to give information on the issue at hand. Which information are the conflict parties looking for? Can they agree on their need for information and, later, on the validity of the results of scientific assessment? Going into such a highly specific and contextualised conflict with a method ready to use, and with no flexibility (as this method is supposed to be “the one and only”) will not support conflict reconciliation. Whenever science intends to help resolving the conflict, it has to be open for an adaptive and appropriate information management. Only the conflict parties can know how they want to handle the complexity, how much they need to reduce it, which information can be used by them etc. Factors determining the need for information are the institutional openness (e.g.: is there an institutional procedure available for dealing with the issue? Is there an administration accountable for any resolution of the conflict?), the phase in the conflict cycle (e.g.: has there just been an attempt to handle this? Has it failed in the view of one of the groups? Is a conflict emerging?), the power of the different stakeholders (e.g. has one group the impression of never being heard? Who dominates the discourse? Which group uses how much money in this process?), trust, legitimacy of the overall framework for biodiversity conflicts, funds available for the conflict resolution process, etc.

Characterising the conflict before giving information helps to focus not only the information input from science, but also the selection of appropriate conflict resolution procedures. An evaluation of different procedures along different criteria (information management, legitimacy, social dynamics, and costs) aims at selecting more effectively an appropriate procedure. In most cases and due to the double complexity of biodiversity conflicts, this procedure has to include participatory elements and an analytical handling of the issue. How much and in which way, is less open to scientific truth, but more to the power of judgement of the mediator of the conflict at hand.

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