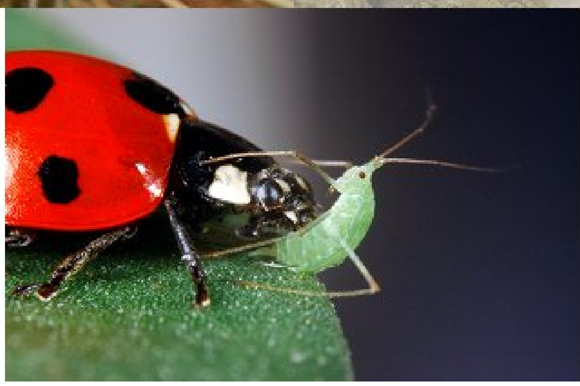
Ecosystem Services and Drivers of Biodiversity Change





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Introduction

Paula Harrison, RUBICODE co-ordinator

The wellbeing of humans is integrally linked to the wellbeing of the other species with which we share the planet. There is now wide acceptance that if the current rate of loss of biological resources is continued, the result will be catastrophic for humankind within a few generations. This loss of species and genetic diversity decreases the resilience of ecosystems, but at the same time ecosystems are experiencing growing pressures from drivers such as climate change, land use change, pollution and invasive species. The challenge is to translate these threats to biodiversity into tangible and quantifiable factors which can be used by policy-makers to promote the development of flexible and effective conservation strategies. Increasing knowledge and awareness of the goods and services provided by ecosystems, and the importance of conserving them for maintaining our own quality of life, aims to address this challenge.

RUBICODE (Rationalising Biodiversity Conservation in Dynamic Ecosystems) is a Coordination Action Project funded by the EU to review and develop concepts of dynamic ecosystems and the services they provide. Methods for relating biodiversity in dynamic ecosystems to the provision of ecosystem services are being compared and evaluated in order to increase our understanding of the value of ecosystem services and, consequently, of the cost of losing them. Frameworks for linking biodiversity traits to service provision and for improving and testing indicators are also being developed and used to explore management strategies and inform priorities for biodiversity conservation policy. Further information on the project can be obtained from the project's website (www.rubicode.net).

The aim of the e-conference was to review and advance discussions involving around 100 scientists who attended the RUBICODE international workshop on "Ecosystem Services and Drivers of Biodiversity Change", which was held in Helsingborg, Sweden from 25 to 28 February 2008. At the workshop, participants were divided into a number of breakout groups where specific issues related to the assessment of ecosystem services were discussed. The e-conference aimed to evaluate the representativeness of these discussions and recommendations with the wider scientific community and expand on the inventory of research needs.

The e-conference focussed on the four following issues:

Session I: Frameworks and approaches for ecosystem service assessment: This session, chaired by Martin Sykes and Paula Harrison, aimed to identify and discuss what frameworks and approaches are currently being used for ecosystem service assessment, whether they adequately capture the linkages and dynamics of our systems and the appropriate temporal and spatial scales, and their relevance to policy and management decision-making.

Session II: Drivers and scenarios for ecosystem service assessment. Drivers represent the underlying causes of environmental change and include social, economic, technological, political, policy and governance factors. Changes in all these factors will, individually and in combination, affect the world around us. This session, chaired by Mark Rounsevell and

Tatiana Kluvankova, aimed to discuss current activities and identify new research directions on the role of drivers and scenario development in studies of ecosystem service provision.

Session III: Valuation of ecosystem services. The main objective of the session, chaired by Michalis Skourtos, was to contribute to an understanding of the state-of-art in the design, empirical application and policy relevance of valuation exercises. The session considered how valuation studies define and communicate changes in ecosystem service provision, how well monetary valuation methods address the needs of conservation strategies, existence values, spatial context, and the complementarity of monetary and non-monetary approaches.

Session IV: Research priorities for ecosystem service assessment. The objective of this session, chaired by Allan Watt, Josef Settele, Martin Musche and Christian Anton, was to start a discussion within the wider research community on research priorities for ecosystem service assessment. Participants were invited to send their suggestions on future research needs related to stakeholder engagement, the development of concepts and frameworks, valuation, drivers of change, and indicators of ecosystem services (amongst others).

This report includes all the contributions made during the e-conference, as well as summaries of contributions and research recommendations identified by participants in the above sessions.

Summary of contributions

Fiona Grant

Summary of contributions: Week 1

Session I: Frameworks and approaches for ecosystem service assessment

The co-chairs of session one, Martin Sykes and Paula Harrison, introduced the main aims of this session: to identify and discuss what frameworks and approaches are currently being used for ecosystem service assessment and to question whether they adequately capture the linkages and dynamics of our systems at the appropriate temporal and spatial scales, and to consider how relevant they are for policy and management decision-making.

In the first contribution to this session, Gary Luck and Richard Harrington outlined the concept of the Service-Providing Unit (SPU). They highlighted the need for future research to follow a systematic approach that identifies explicitly service beneficiaries, the level of need for the service, the spatio-temporal scale of the service need, and the components of biodiversity that provide the service. They also addressed the need to quantify the characteristics of service providers and how changes in these characteristics could impact on service provision.

Terry Dawson and Mark Rounsevell outlined two other frameworks for ecosystem service assessment, the Drivers-Pressures-State-Impact-Response (DPSIR) and Social-Ecological System (SES) frameworks. They proposed that integration of these frameworks for ecological services management could provide a robust approach for the implementation and monitoring of adaptation strategies to reduce systems vulnerabilities. Their contribution addressed the need for future research to concentrate on the identification of suitable indicators in the context of the appropriate spatial and temporal nature of the drivers and pressures on the SES to ensure that the system dynamics and response is adequately captured and that adaptation strategies can be implemented and monitored.

Other contributions to this session concentrated on approaches for ecosystem service assessment. Rob Marrs focussed on ecosystem services assessment in heathlands. He highlighted the need for future research to take a long-term view and include any risk from wildfire/vandalism fires. He argued that quantification and modelling were key to assessing the accumulation and loss of carbon etc from heath systems. Sandra Luque highlighted a different approach, concentrating on forest ecosystems. She discussed the need to integrate ecology and economics to evaluate and design biodiversity management strategies and to consider examples of multi-use forestry that would improve sustainable forest management. Brian Moss' contribution outlined another approach to ecosystem service assessment, focussing on freshwater ecosystems. He argued that scientists should stop classifying ecosystems in terms of individual habitats, and addressed the need to consider ecosystems in terms of biospheres, incorporating all interacting habitats, in order to stand a chance at conserving ecosystem services. Future research needs to therefore consider possible linkages between classified habitats.

Session II: Drivers and scenarios for ecosystem service assessment

Mark Rounsevell opened this session with an introduction outlining the main aims of Session II, namely to discuss current activities and identify new research directions on the role of drivers and scenario development in the study of ecosystem service provision.

This session began with a contribution from Axel Volkery, who outlined the structure behind participatory approaches to scenario development and the potential benefits of the involvement of scientists and non-scientists when considering scenarios for ecosystem service assessment. He discussed the need for more combined approaches and highlighted that although these were often more difficult to manage they promised greater returns in terms of comprehensive, scenario-rich narratives.

Several contributions considered different drivers of ecosystem service assessment. Erik Gomez-Baggethun and Eszter Kelemen focussed on how institutional disruption could act as a key driver of change in the flows of ecosystem services. They highlighted the need for future research to:

- Further understand the links between institutions and ecosystem services to support scenario planning
-Use case studies that identify specific links between institutional change and ecosystem services flow
- Explore how institutional diversity for ecosystem services management can be maintained and promoted in the ongoing process of European environmental policy integration.

Ines Omann's contribution concentrated on human lifestyles as a key driver of ecosystem service assessment. She outlined inter-linkages between socio-economic and natural systems, in particular between scenarios of lifestyles (drivers), their influence on global change (pressure), the consequent changes of ecosystems (state) and subsequent change of ecosystem services (impact on human wellbeing) and societal responses (policies; changed lifestyles due to reduced quality of life). Frank Ewert highlighted another key driver of ecosystem services change: technological development. His contribution focussed in particular on the agricultural industry and called for researchers to look at the role of technological development for ecosystem service change and the assumptions underlying the projected development of technology.

Session III: Valuation of ecosystem services

Michalis Skourtos introduced the main objectives of his session, specifically to consider how valuation studies define communication changes in ecosystem service provision, how well monetary valuation methods address the needs of conservation strategies, existence values, spatial context, and the complementarity of monetary and non-monetary approaches.

George Cojocaru responded to the introduction outlining the difficulties inherent in ecosystem services valuation due to the diversity of services and the diversity of places in which the services are applied. He proposed the need to split ecosystem services into three/four pillars: economic services, environmental services, social services, and, potentially, aesthetic services. He argued that although the monetary valuation method was adequate to assess the value of economic services, this method should not be applied to the other services. As such, he suggested the need to collate lists of the most important social and environmental services and then, from those lists, use a mathematical or empirical method to estimate their values.

Areti Kontogianni and Gary Luck continued this session by outlining the need for standardised methods to measure ecosystem services in order to be able to place a valuation on them. They highlighted the importance of the development of new terminology, such as service providing units (SPUs) and ecosystem service providers (ESPs) in allowing a better understanding of complex ecosystem processes, thus aiding quantification and ultimately valuation of these services. They addressed the need for future research to consider the following questions:

- Will SPUs enhance interdisciplinary collaboration and understanding between ecologists and economists and thus promote more validated, well-informed valuation applications?

- Do SPUs apply equally well to both use and non-use values?
- Could SPUs enhance understanding of the objects being valued and accordingly bypass the cognitive problems giving rise to heuristic devises used by respondents in stated preference surveys?
- Are SPUs more suited to address dynamic issues of value estimation?

Other contributions in this session considered how well monetary valuation methods addressed the need of ecosystem services. Ian Bateman's contribution focussed on this in the context of existence values. He outlined the difficulties associated with measuring existence values, in particular the application of various frameworks for economic analysis of existence values, such as contingent valuation and choice experiment methods. Dominic Moran's contribution concentrated on this issue in a spatial context. He discussed the difficulties of calculating how much of a target ecosystem should be conserved and the consequences that may then follow. In particular he considered the importance of understanding the scale required for relevant ecosystem processes and functions to occur in conserved areas. He addressed the need to be specific about how scale fits into appraisal and valuation of ecosystem services, and to consider whether and how to draw system boundaries around the relevant units of provision of goods and services.

Session IV: Research priorities for ecosystem service assessment

The co-chairs of session four, Allan Watt, Josef Settele, Christian Anton and Martin Musche, introduced the main aims of this session, namely to consider future research needs related to stakeholder engagement, the development of concepts and frameworks, valuation, drivers of change, and indicators of ecosystem services.

This session began with a contribution from Martin Sharman, who outlined the difficulties posed as a result of badly defined terminology. In his particular example he concentrated on the term 'ecosystem', arguing that it had a normative dimension, i.e. that the term emphasised how things should be rather than how they actually are. He therefore highlighted the need to identify concepts and associated behaviours that could allow for a sustainable relationship with nature.

Erik Gomez-Baggethun and Eszter Kelemen also highlighted some relevant research priorities for this session. They addressed the need for future research to analyse how institutions at local, national and international scales could complement each other within multi-level management systems to ensure that protection of ecosystem services was valued by stakeholders at different scales. They also proposed the need to develop a consistent theoretical framework and methodological guideline that would help to identify local institutional guidelines that were successful in the maintenance of biodiversity and key ecosystem services.

Summary of contributions: Week 2

Session I: Frameworks and approaches for ecosystem service assessment

The second week of the e-conference began with a contribution from Sandra Lavorel, Francesco de Bello, Sandra Diaz, Jonathan Storkey and Richard Harrington outlining a framework linking response traits and effect traits across trophic levels. The authors outlined its use in assessing the vulnerability of ecosystem services to changes in environmental pressures. Other contributions in this session concentrated on approaches for ecosystem services assessment. Winfried Voigt highlighted the importance of grasslands in Europe as ecosystem service providers. Bruce Jones and Carl Shapiro focussed on landscape dynamics, highlighting the need for frameworks to allow the inclusion of spatial variation and uncertainty in the composition and functioning of ecosystems. An agricultural approach was considered by John Porter and Lene Sigsgaard, who argued that because such a large proportion of the Earth's land was devoted to agriculture and because its ecosystem services provision had been driven to a low level it was difficult to see how global ecosystem services could increase without significant improvements in ecosystem services from farming. A further contribution was made by Sarah Gardner in response to Rob Marrs' contribution on ecosystem service assessment in heathlands. She concentrated on the effect of livestock grazing on upland moor in England and Wales, highlighting that an increase in sheep numbers had led to a loss of biodiversity in heathlands, and subsequent damage to ecosystem services.

Session II: Drivers and scenarios for ecosystem service assessment

Nico Keilman highlighted the importance of demography as a driver for ecosystem service assessment. He outlined the four main demographic trends expected in the coming decades and discussed two techniques used by demographers in mapping future population trends, namely scenarios and probabilistic projections.

Felix Rauschmayer outlined multi-level governance schemes and highlighted the importance of understanding the social processes underlying the production of socio-political scales and the need to analyse how these social-spatial dimensions are produced by social processes in order to fully grasp cross-scale interactions.

Annette Piorr's contribution assessed both drivers and scenarios in the context of land use change. She outlined the main factors leading to land use changes, and changes in ecosystem services and functions and highlighted the need to develop downscaling procedures for scenarios in order for them to be applied at different scales.

Tatiana Kluvankova-Orvaska's contribution outlined the need to integrate Social-Ecological Systems (SES) into the RUBICODE approach. She offered justification of the importance of having a dynamic concept of biodiversity conservation within the EU and linked it with the existing concept of the Drivers-Pressures-State-Impact-Response (DPSIR) framework.

Session III: Valuation of ecosystem services

In his keynote contribution to Session III , Kerry Turner defined ecosystem services as being those aspects of ecosystems used to improve human well-being. He argued that a classification scheme that could divide ecosystem services into intermediate, final services and benefits would be a useful tool for valuation purposes.

Rob Tinch's contribution was based on a question raised by Allan Watt in his opening statement to Session IV: 'What research is needed to develop indicators that assist decision-makers in managing ecosystems and the services they provide?'. Rob Tinch agreed with Martin Sharman about the importance of identifying concepts and associated behaviours that would allow for a more sustainable relationship with nature. He also highlighted that the relationship between human activity and the natural world was now a fundamental aspect of social choice and drew upon Michalis Skourtos' contribution arguing that economics needed to deal with scarce resource allocation, rather than just monetary or market-line considerations.

Session IV: Research priorities for ecosystem service assessment

Numerous research priorities for ecosystem services assessment were raised this week. Below is a summary of points raised in this session. For a full list of the research recommendations please refer to the 'Draft list of Research Recommendations' document.

Sandra Diaz's contribution concentrated on the role of scale in ecosystem service assessment. She highlighted the limitation of experiments and models carried out at too small a scale and argued for the need for more 'natural experiments' to be carried out. Sandra Luque's contribution also focussed on scale. In her contribution she outlined the issue of scale dependency and understanding mechanisms at different levels, and the resistance and resilience of ecosystems in relation to feedback mechanisms. She highlighted the importance of understanding the interrelation between ecosystems and landscape level mechanisms.

The need for future research on associations between biodiversity and ecosystem goods and services was the subject of Martin Zobel and Mari Moora's contribution, in which they argued that biodiversity was positively associated with an ecosystems' capability to provide goods and services, and that recent experiments had shown that functional diversity, rather than species number per se, was important. Similarly, Francesco de Bello highlighted the need to further assess the links between functional diversity and ecosystem services.

Richard Harrington briefly discussed the difficulties associated with conserving biodiversity for the sake of ecosystem service delivery and attempting to value ecosystem services in order for them to compete with alternative land-use options. He argued for the need to investigate the risks associated with such approaches, including the issue of species which may have no value in the provision of the service for which the approaches were designed or, indeed, any other currently recognised service.

A number of contributions focussed on the need for future research on service provisions. Nikolai Friberg outlined that a biological trait-based approach to ecosystem service provision was a promising step towards a more process orientated understanding, but that this approach did have some limitations: the use of biological traits in nature is still too qualitative and static, and the capability of ecosystems to provide a range of services will be more strongly related to dynamic features such as ecosystem organisation and fluxes of energy. Pam Berry considered the links between human behaviour, species' attributes and supporting habitats in response to environmental pressures, and how these determine whether ecosystem services are provided at an adequate and sustainable level. Similarly, Gary Luck's contribution highlighted the need to link quantitative changes in service providers with impacts on service provision.

Ines Omann's contribution analysed the link between quality of life and ecosystem services. She argued that the Millennium Ecosystem Assessment had demonstrated a strong connection between ecosystem services and human well-being, and that there was therefore an urgent need to study the impacts of reduced biodiversity or disturbed ecosystems on quality of life.

Contributions from Christian Feld and Paulo Sousa stressed the need to consider indicators for the different components of biodiversity, and their use to indicate service provision and/or underlying ecological processes. Paulo Sousa added the need to develop concerted indication concepts and approaches that could be used across ecosystems. Similarly, Rob Bugter highlighted the need to develop a limited set of standardised indicators which could be used universally. He argued that it may be necessary to implement a new directive focussed on biodiversity monitoring and assessment.

A number of contributions highlighted the need to consider the effects of climate change on biodiversity conservation. Richard Johnson outlined problems associated with managing ecosystems, focussing on patterns and changes in species diversity and the added problems faced due to climate change and human-induced stressors.

Rob Bugter posted a contribution from Willemien Geertsema, Paul Opdam, Claire Vos and Koen Kramer, which highlighted the need for a paradigm shift in biodiversity conservation planning due to climate change. In response to this contribution, Olly Watts argued that climate change does not enforce a paradigm shift in biodiversity conservation. He proposed that huge efforts are required before paradigm shifts can be put into place, and instead we should focus on our current frameworks in order to protect Europe's wildlife from a wide range of impacts alongside climate change. In his own contribution, Rob Bugter argued that there was currently a lack of knowledge on how land use change and climate change would affect biodiversity and what this could mean for us in the future. Following on from this contribution, Dominic Moran highlighted the need to translate our limited knowledge on biodiversity damage costs into important climate change calculations that influence how much we spend on greenhouse gas mitigation.

Finally, John Haslett's contribution argued that future research should reflect gaps in present scientific understanding in combination with the practicalities of acceptance and implementation by stakeholders. He also focussed on the need for future research to concentrate on ecological corridors and invertebrate conservation.

Research priorities

Fiona Grant, Juliette Young & Allan Watt

1. Research needed to engage stakeholders in ecosystem services

- Analyse: 1) the plurality of decision and communication contexts within societies, 2) the relative merits of different classification frameworks, evaluation methods and decision support tools for these contexts, and 3) the scale dependency and cultural dependency of answers to 1 and 2.
- Analyse how institutions at local, national and international scales can complement each other within multi-level management systems to ensure that the protection of ecosystem services is valued by stakeholders at different scales.

2. Research needed to develop concepts and frameworks for ecosystem services

General

- Develop a consistent theoretical framework and methodological guidelines to identify local institutional guidelines that are successful in the maintenance of biodiversity and key ecosystem services.
- Develop frameworks and models to quantify the effects of relationships between beneficial invertebrates in adjacent grasslands, crop and fruit yield in agricultural fields or orchards.
- Translate scientific uncertainty into the best no-regret strategies, monitor the effects of measures in an adequate way, and communicate the need for flexible, adaptable strategies and get them implemented.
- Develop process-orientated understanding of food web architecture and quantify how energy flows between the nodes within a food web, which can then be related to the provision of services.

Models

- Model functional diversity responses to different factors.
- Model biological feedback onto ecosystem services, across spatial scales.
- Model thresholds beyond which the level of ecosystem service delivery changes dramatically and perhaps irreversibly.
- Integrate agent-based modelling of human behaviour (Ecosystem Service Beneficiaries) with models of the species and habitat which provide the service (Ecosystem Service Providers).
- Model externalities (positive and negative) across the landscape.

Experiments

- Take advantage of 'natural experiments' and also incorporate real land-use situations in which carefully controlled, factorial design is not suitable.

- Develop experiments on the effects of different components of biodiversity on ecosystem processes at the scale in which management is done in practice.

Scale
- Consider different scales in a more integrative landscape level approach.
- Study pattern-process levels and scale-dependent mechanisms at different spatio-temporal scales.
- Address governance and conflict resolution at different scales.

Grasslands
- Define specific SPUs and general measures for the amount or a quantitative value of the palatability in grasslands for specific livestock.
- Integrate the function of grasslands providing fresh clean water into economic models and decision frameworks.
- Incorporate more quantitative information to link SPUs with the control of soil erosion.

3. Research needed for valuation of ecosystem services

- Improve estimates of ecosystem services from agriculture, examine alternative methods of valuation and establish the connection between farmland design, management and ecosystem services.
- Investigate the risks associated with conservation based on ecosystem service delivery.
- Develop a valuation classification scheme that divides ecosystem services into immediate services, final services and benefits.

4. Research needed to understand and identify drivers of change of ecosystem services

General
- Understand the fundamental relationships between a) community diversity and composition and b) ecosystem services (support services in particular). These studies should address the dynamic status of ecosystems, i.e. take into account different capabilities of ecosystems under different human pressures.
- Quantify ecosystem responses to human-induced changes in biodiversity dynamics and the actions necessary to maintain delivery of ecosystem function and services in the face of changes in climate, land use and social attitude.

Institutional drivers
- Identify specific links between institutional change and ecosystem services flows through case studies.
- Explore how institutional diversity for ecosystem services management can be maintained and promoted in the ongoing process of European environmental policy integration.

Scale
- Understand horizontal interactions and scaling functions of biotic, abiotic, and human factors that influence ecological processes and services at multiple landscape scales.
- Develop downscaling procedures for change in land use scenarios in order for them to be applied at different scales.
- Understand how horizontal interactions across multiple landscape scales influence ecological services.
- Analyse how socio-spatial dimensions are produced by social processes.

5. Research needed to identify indicators of ecosystem services

- Develop biodiversity indicators that account for the different aspects of diversity, address ecosystem functions, processes and ecosystem services, and are applicable at regional, sub-global and global scales.

6. Research needed for ecosystem management

- Understand the intricate linkages between systems and how these interactions are important for resistance and resilience to stress.
- Develop decision support systems to make information readily available to land managers.

7. Research needed on the use of traits

- Analyse relationships between species' traits and taxonomic distinctness and develop predictions of trait responses to different types of stress.
- Produce shortlists of key traits for different organisms and define standardised protocols to assess these lists.

List of contributions

Session IV: Research priorities for ecosystem service assessment

Session I: Frameworks and approaches for ecosystem service assessment

Introduction to Session I: Frameworks and approaches for ecosystem service assessment

Martin Sykes, Department of Physical Geography & Ecosystems Analysis, Lund University, Sweden & **Paula Harrison**, Environmental Change Institute, Oxford University Centre for the Environment, UK

Many current conservation strategies are developed around a static and uniform view of nature and the environment. Ecosystems are however dynamic. Thus for successful current and future conservation objectives it is important that new conservation strategies are developed and implemented that concentrate on managing dynamic ecosystems for maintaining their capacity to undergo disturbance, while retaining their functions, services and control mechanisms (Gunderson, 2000). Ecosystems are also multifunctional systems which provide humanity with vital services for example, food, wood, water, soil protection, climate regulation as well as many cultural and aesthetic services. Continued provision of these services, particularly in a rapidly changing world, requires that the multi-functionality of ecosystems is taken into account in their management.

The RUBICODE project has reviewed the current state-of-the-art with regard to concepts and frameworks for the assessment and quantification of services within dynamic ecosystems (Vandewalle et al., 2008). The framework for ecosystem service assessment proposed by the Millennium Ecosystem Assessment (MA, 2005) is perhaps the most well known. The RUBICODE project is building on the work of the MA and has developed a number of concepts and frameworks for ecosystem service assessment which will be introduced during the e-conference. These include:

- The service-providing unit (SPU) concept defined as the components of biodiversity necessary to deliver a given ecosystem service at the level required by service beneficiaries.
- A conceptual framework for quantifying the effects of drivers of change on ecosystem service provision based on coupling the DPSIR (Drivers-Pressures-State-Impact-Response) and SES (Socio-ecological systems) concepts.
- A framework for linking ecosystem service provision to biological traits.

These approaches to the assessment of ecosystem services were presented at the RUBICODE international workshop held in Helsingborg, Sweden from 25-28 February 2008. A background report for the workshop summarises the frameworks, and comments received from workshop participants are documented in the workshop report (both available from http://forums.ceh.ac.uk:8080/~rubicode). During this session of the e-conference we wish to solicit further comment on these approaches and widen the debate to discuss some key issues relevant to ecosystem service assessment. We invite contributions from all participants in the e-conference on what conceptual frameworks are relevant to ecosystem service assessment.

Specifically, the following questions are posed:
- What frameworks/approaches are being used?
- What is the added value of the different frameworks/approaches?
- Do they adequately capture the linkages and dynamics of our systems?
- What are the appropriate temporal and spatial scales?
- Are they relevant for policy and management decision-making?

The session is based on keynote contributions from a number of international experts on the subject of frameworks and approaches for ecosystem service assessment. Topics to be covered include: introduction to the three RUBICODE concepts/frameworks, ecosystem service assessment and landscape dynamics, and approaches for ecosystem service assessment in forest, heath & shrub, grassland, agricultural, mountain, soil and freshwater ecosystems. We hope that this collection of contributions will stimulate debate, identify research needs and promote the development of a research community with interests in advancing methodologies for the assessment of ecosystem services.

Quantifying the contribution of organisms to the provision of ecosystem services: The SPU concept

Gary Luck, Institute for Land, Water and Society, Charles Sturt University, Australia & **Richard Harrington**, Department of Plant and Invertebrate Ecology, Rothamsted Research, UK

The service-providing unit (SPU) concept is a framework for quantifying the biotic components of ecosystems that supply services to humanity, something that is desperately needed to understand the implications of changes in ecosystems for human wellbeing.

Ecosystem services are the benefits humans derive from ecosystems. Provision of these benefits occurs at multiple spatial and temporal scales. Quantifying the link between the characteristics of organisms and service provision is of crucial importance to effective land management. A service-providing unit (SPU) can be defined as the collection of organisms and their characteristics necessary to deliver a given ecosystem service at the level required by service beneficiaries. The concept is scale independent, but context specific. Application requires clear identification of service beneficiaries and the level of need for the service, and the capacity to identify and quantify the organisms and their characteristics providing the service.

The SPU concept was initially introduced as an approach to explicitly link species populations with the services they provide to humans (Bird et al., 2000). For example, a density of 33 mallard ha-1 over a 180-day period was sufficient to improve the decomposition of rice straw (compared to treatments with no mallard) (Kremen et al., 2002). A certain population density is crucial for service provision, although there was no indication of the consequences of lower densities (other than zero) or the level of need for the service. The SPU concept can be extended to functional groups and beyond (e.g. ecological communities). For example, watermelon crops in California were pollinated by several native bee species. Maintaining the diversity of the native bee community is essential because of temporal fluctuations in the population of any one species and variation in pollination effectiveness among species (Luck et al., 2003). The SPU is the composition of the functional group, the functional traits of each member, the population characteristics of each member and appropriate spatial and temporal dynamics to deliver the service at the desired level.

The SPU concept implies that if a collection of organisms is not contributing to service provision at the desired level it does not constitute an SPU. That is, there is a threshold level of service delivery above which a group of organisms is considered an SPU. Invoking this threshold is important because ecosystem services must be defined by both the contribution of [potential] service providers and the requirements of service beneficiaries. For example, the threshold may be a desired level of natural pest control that reduces the reliance on pesticides and results in crop yields at a given profit margin. However, thresholds are blunt instruments that potentially draw attention away from the need to understand how incremental changes in the characteristics of service providers impact on service delivery. The latter is very important because it helps to identify the trade-offs in obtaining a given outcome through ecosystem services or anthropogenic alternatives (e.g. the cost-benefits along a continuum of options for controlling pests based on various combinations of natural control from native and/or exotic species and pesticides).

Quantifying the characteristics of organisms that contribute to service provision is crucial to guiding land management and policy development. However, such examples are extremely rare in the scientific literature. Researchers may identify ecosystem service providers, but not quantify the units required for service provision, or research on ecosystem function may provide detailed quantification of functional units, but not elaborate on their potential for the provision of ecosystem services. Future research must follow a systematic approach that identifies explicitly service beneficiaries, the level of need for the service, the spatio-temporal scale of service need and the components of biodiversity that provide (and support provision of) the service, in addition to quantifying the characteristics of service providers and how changes in these characteristics impact on service provision.

Approaches to ecosystem service assessment in heath and shrub systems

Rob Marrs, Applied Vegetation Dynamics Laboratory, University of Liverpool, UK

Heathlands are usually anthropogenic systems, and are ideal model systems for investigating the conflicts between (1) management for conserving biodiversity and (2) contributing to ecosystem services. Research must take a long-term view, and include any risk from wildfire/vandalism fires.

Background:

1. In many parts of the world heath and shrub communities (hereafter heaths) are anthropogenic systems, maintained in a sub-seral state by management (grazing, cutting, burning) (Gimingham 1972). Such systems have a high biodiversity as cultural landscapes and are obliged to be protected under EU/national designations ((Habitats Directive-92/43/EEC; Birds Directive-79/409/EEC, Anon, 1995, MAFF, 1993). THUS, there is a policy driver that forces the management of heaths towards maintenance of the status quo.

2. Heathlands, at least in the UK, are predominantly found in the uplands (cold and wet), and the soils are either peats or have a well developed organic layer. If this is the case the following statements must be TRUE:

a. There is a Carbon sink of unknown magnitude in these upland heath areas.
b. All downstream ecosystems must be viewed as the sewers of these upland heaths.

3. All heathland management will disrupt the ecological processes within the communities; biotic control (Bormann & Likens 1978) will be reduced and there will be a loss of carbon/nutrients to the air or sewers to a greater or lesser extent depending on the management. This will impinge on water quality and purification costs.

4. If a conservation manager decides to implement management then there must be an acceptance that damage will occur. The important point is to quantify the relative risks, and to balance this against other potential risks to the system.

Case-study 1: Heathlands or late-successional communities. Marrs et al. (2007) assessed these potential conflicts using a Heather-dominated heathland invaded by bracken, where the aim was to control bracken/restore Heather. Management (especially cutting twice/yr) successfully increased plant biodiversity but at the same time there was a significant loss of C and N. Bracken management had two opposing effects, (1) positively on biodiversity and (2) negatively on ecosystem services. Management for ecosystem services will reduce biodiversity.

Case-study 2: Heathlands & burning. Many heathlands in the UK are burned on rotation during winter/spring to provide new growth for sheep/grouse. These prescribed burns must be distinguished from fires started deliberately/accidentally, which often occur in the summer and are much more damaging. There are two current schools of policy-thought:

1. The no burn approach – here carbon losses will be reduced during the point where biotic control is lost after the burn, the system will aggrade, carbon will be sequestered and peat will develop. The vegetation biomass will increase a lot. This is a policy designed primarily for carbon sequestration.

2. The management-burn approach - here there is an acceptance of small losses of nutrients during burning when biotic control is lost, but there is rapid recovery. If this is done in small rotations then losses will be minimized and the vegetation biomass will be maintained at a relatively low level. This is a policy designed primarily for maintaining the ecosystem in its present condition for biodiversity.

The issue facing policy makers is to place these two options in a scenario where there is exposure to accidental/vandalism fire risk: e.g. the Peak District in the UK has had very severe vandalism fires in the past and it must be assumed that all heathland there is at risk. We can postulate that: 1. Under the no-burn approach, fuel loads could be very high, if these were burned then fire temperatures are also likely to be very high and we would expect severe damage to the peat and massive carbon and nutrient loss. As vegetation damage will be severe, recovery will take a very long time. 2. Under the management-burn approach, fuel loads will be reduced, fire temperatures are likely to be lower and hence damage should be less. Fire control will also be easier. We would expect some vegetation damage but re-sprouting might be possible so there should be rapid recovery.

Unfortunately, we do not have the data yet to assess the amounts of carbon and nutrients lost during management burns, or indeed carbon aggrading during post-fire recovery.

Gaps in our knowledge: The real difficulty in assessing the role of heath systems is quantifying the losses of carbon etc from process studies set against accumulation in the longer term and site diversity. The heathland example above can be viewed simply: Is it better to lose a little carbon annually and a modest amount if there is a vandalism fire, or is it better to lose almost no carbon in the medium term, but be at risk to losing a very large amount during vandalism fires? Quantification and modelling are the keys to answering this question based on rational grounds. Heathlands offer a good model system for studying ecological processes such as these (Vitousek 2006).

Re: Ecosystem service assessment in heathlands

Sarah Gardner, GardnerLobo Associates UK

In his posting on Ecosystem Services (ES) assessment in heath & shrub systems, Rob Marrs highlights potential trade-offs between biodiversity conservation and ES provision brought about by management activities. I would like to add some further thoughts to this discussion from recent work on livestock grazing on upland moor in England & Wales (Critchley et al. 2007).

It has long been recognised that grazing animals are important in maintaining the mosaic of dwarf shrub, grass and mire communities, characteristic of upland moor. However, significant increases in sheep numbers, stimulated by government livestock support policies in the 1970's & 80's, have been associated with the loss of biodiversity (particularly dwarf shrubs & moorland birds) and damage to ES such water quality, carbon sequestration, game production and landscape amenity (English Nature 2001). Replacement of these policies with ones to encourage greater extensification of grazing management has resulted in variable benefits (& frequently little change) to biodiversity (e.g. Hope et al. 1996; Marrs et al. 2004) and some benefits to ES such as water quality.

A principal driver of livestock management on moorland is farm economics and the influence on this of livestock prices, government support mechanisms and the availability of alternative income streams. Thus the 'health' of moorland biodiversity and ES services and the impact of management on this, cannot be evaluated without considering the role of economics in influencing land management choices.

In considering Frameworks & Approaches for ES assessment I would therefore agree with Sandra Luque (see posting on Forest Ecosystems) who highlighted the need to integrate ecology and economics into the evaluation of biodiversity and ES management strategies. Indeed for ecosystems where production of a particular Provisioning commodity (livestock, game, timber etc.) is an important driver of ecosystem management, the role of economics in driving management decisions will be an important element in Frameworks for monitoring the health of ES.

Finally I would also support the point raised by Bruce Jones (Landscape posting) that any such Frameworks should allow for the inclusion of spatial variation and uncertainty in the composition and functioning of ecosystems. In a study of sustainable grazing regimes for moorland, it was clear that heterogeneity in the abundance and spatial arrangement of habitats had a significant impact on the outcome of specific grazing practices for biodiversity (Critchley et al. 2007; Gardner et al. in press; see also Vandvik et al. 2005).

In summary I would anticipate that any Frameworks & Approaches for ES assessment should enable consideration of the following:

1. Trade-offs between services and, as appropriate, between services & biodiversity arising from management or other human impacts;
2. The role of specific economic drivers in determining ES health
3. The effect of spatial organisation in mediating the robustness of ES responses to human impact.

The integration of DPSIR and SES frameworks

Terry Dawson, University of Southampton, UK & **Mark Rounsevell**, Centre for the Study of Environmental Change and Sustainability, School of Geosciences, University of Edinburgh, UK

The integration of the Drivers-Pressures-State-Impact-Response and Social-Ecological System frameworks for ecological services management provides a robust approach for the implementation and monitoring of adaptation strategies to reduce systems vulnerabilities.

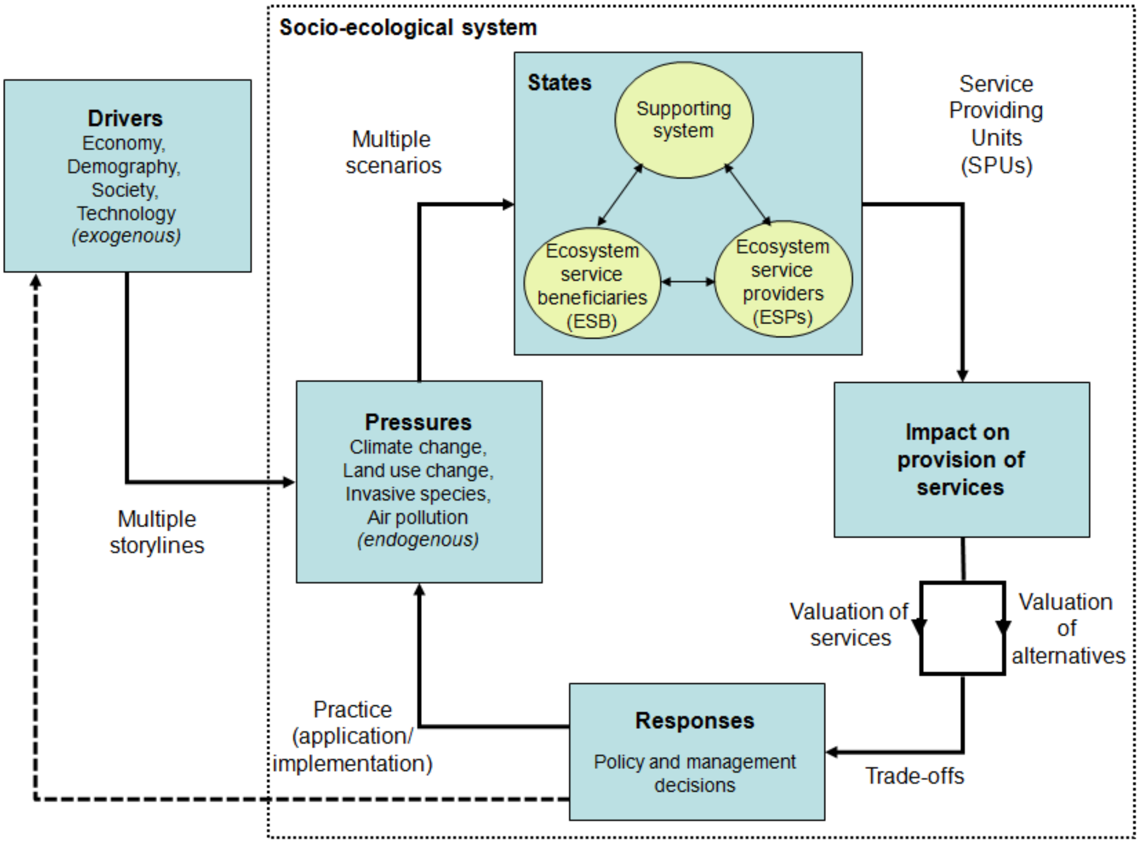
Predicting environmental change and its impacts on human well-being and natural ecosystems at local to global scales remains a significant challenge for the international scientific community (MEA 2005). Uncertainty on the interactions and feedbacks between the natural and human drivers of environmental change that may operate at different spatial and temporal scales make it difficult for societies to resolve an appropriate course of collective action to pursue sustainable livelihoods. The Drivers-Pressures-State-Impact-Response (DPSIR) framework has evolved into an interdisciplinary tool for environmental analyses (EEA 1995) and assumes cause-effect relationships between interacting components of social, economic, and environmental systems.

However, criticisms have been raised about the linearity of cause-effect schemes such as the DPSIR (Svarstad et al., 2007). The interactions between drivers, pressures and responses are much more complex as a result of positive and negative feedback responses existing between different activities, economic and social mechanisms, and policy responses having multiple effects, etc (Fusco, 2001). To effectively capture social and ecological dynamics and the crucial human dependence on ecosystems, the Social-Ecological System (SES) has been defined (Gallopin, 1991) that articulates social and ecological systems as strongly coupled and complex. Social systems include the economy, actors and institutions in mutual interaction. Institutions are durable systems of established and embedded social rules (convention, norms and legal rules) that structure social interaction (Hodgson, 2002) and thus are different from organisations and other actors. Ecological systems include self-regulating communities of organisms interacting with one another and with their environment (Folke, 2003).

SES vulnerability, defined as an exposure to threats affecting the ability of the SES to cope (e.g. failure in food production, ecological services provision, etc.), can arise from endogenous and exogenous factors across multiple time-scales and can range from transient shocks or disruptions through to chronic or enduring pressures. A highly resilient SES would be able to recover and retain its structure and function following a transient and exogenous shock. Stability refers to a system's tolerance to transient and endogenous disruptions. Durability represents a system's ability to recover or maintain its social-ecological functions in the face of a chronic endogenous stress. Robustness is the property expressed when a system is able to cope with an external and chronic pressure. As Stirling (2007) states, each property is individually necessary and collectively sufficient for achieving sustainability. If these system components have been eroded, a disturbance may be more likely to push the system beyond a threshold state from which it may not recover or may take many years to return to its previous state through natural processes (Kinzig et al. 2006).

Integrating the DPSIR and SES frameworks (see Figure 1) provides many advantages for the management of ecological services. The indicator system of the DPSIR is well established and embedded in a number of policy decision-making organizations and institutions. Identification of suitable indicators in the context of the appropriate spatial and temporal nature of the drivers and pressures on the SES will ensure that the system dynamics and response is adequately captured and that adaptation strategies to reduce vulnerabilities can be implemented and monitored.

Figure 1. A proposed coupled DPSIR and SES framework for the assessment of the effects of environmental change drivers on ecosystem services



Approaches to ecosystem service assessment in forest ecosystems

Sandra Luque, Cemagref - Institute for Agricultural and Environmental Engineering Research, France

The need to consider both economical and ecological values of forest habitat in order to establish sustainable management methods is discussed, highlighting the importance of comparing expected monetary gain from harvested products with the values associated from ecosystem goods and services lost as a result of harvesting when making forest management decisions.

Ecosystem goods and services and their continued delivery are essential to our economic prosperity and well-being (Millennium Ecosystem Assessment 2005). Modifications of ecosystems to enhance one service have generally come at a cost to other services due to trade-offs. For instance human interventions to increase food and timber production have often resulted in changes to services such as water regulation and recreation activities. An example of where the effect of a change in an ecosystem in one location can have impacts in other locations is the management or harvesting of forest in one region that affects water quality in downstream areas. Since changes in the quantity or quality of various types of natural resources and ecosystem services have important impacts on human welfare and the economy, comprehensive methods to measure and value biodiversity and ecosystem services are needed.

A key reason behind degradation of biodiversity and ecosystem services is that benefits of ecosystem goods and services are not fully captured in the commercial market or adequately quantified in terms comparable with economic services and manufactured capital. Therefore they are often ignored or given too little weight in policy making (Costanza et al. 1997). Decisions on use of natural resources should be based on a comparison of the expected monetary value of the harvested products and the values associated with the ecosystem goods and services lost as a result of harvesting. Forests in particular provide timber material through well-established markets, but the associated value of forest habitat is also gained through un-marketed recreational activities, forest carbon sequestration, maintenance of biodiversity, microclimate, protection against natural hazards and water quality. Decisions on the use of forest resources should be based on a comparison between the expected monetary value of the harvested timber and the costs associated with the ecosystem goods and services that are lost as a result of timber logging.

However, ecosystem goods and services that do not have monetary value are generally not accounted for in the decision making process. There is a need to develop quantitative measures of biodiversity and ecosystem goods and services, in order to achieve sustainable use of forest resources (Kallio et al 2006; Kallio et al 2008). Traditionally, commercial forests are managed to maximize timber output. Our recent work suggests a methodology for integrating economic efficiency and biodiversity value. An integrated approach in forest conservation could provide environmental managers with considerable cost savings while increasing biodiversity protection (Kallio et al 2008).

Earlier research on land management typically keeps economic issues separate from purely ecological issues. In this way, timber production will be considered independently of issues like species conservation and its accompanying value to the system. Research integrating ecology and economics to evaluate and design biodiversity management strategies is scarce, but is slowly gaining ground (e.g., Ando et al., 1998; Polasky et al., 2001; Perrings and Walker, 2004). Many of the studies focused on maintaining the maximum number of species on a given land area or the minimum land area with a given number of species (Ando et al., 1998; Polasky et al., 2001). According to Ando et al. (1998) significant cost savings could be made by integrating economic costs into ecologically based selection of conservation sites compared to traditional ecology-based selection. Another issue of the ecological-economical research has been to help determine site-specific cost-effective compensation payments to land-owners (Johst et al., 2002; Yang et al., 2003).

The tradeoffs between biodiversity and timber harvest value can be derived by the production frontier method,- see for example Pukkala et al. (1997), who developed biodiversity indices and applied them at the forest level for harvest planning. Calkin et al. (2002) explored tradeoffs between the likelihood of persistence of a wildlife species and timber production by applying a model integrating spatial wildlife population, timber harvest and growth models. Nalle et al. (2004) evaluated land-use decisions and looked for cost-effective land-use alternatives. They combined a wildlife population simulation model with the economic model. The aim of the model was to calculate the present value of the sum of consumers' and producers' surpluses from timber harvest. Polasky et al. (2005) analyzed the consequences of alternative land-use patterns on the persistence of species and the economic returns.

Within the framework of this e-conference I would like us to focus on services assessment methods in forest ecosystems, research that integrates ecology and economics to evaluate and design biodiversity management strategies, and to use case studies and experiences in particular for multi-use forestry that may be a valid alternative in order to improve sustainable forest management.

Approaches to ecosystem service assessment in freshwater ecosystems

Brian Moss, School of Biological Sciences, University of Liverpool, UK

Experience from freshwater ecosystems exposes the fatal flaws in the concept of assessment and management of ecosystem services.

Arthur Tansley contributed much to the founding of ecology yet may have completely undermined its importance by establishing the concept (Tansley, 1935) of the ecosystem, the more or less self-contained unit, perhaps through the conditioning of living in the compartmented landscape of England. It would have been better had he been preceded by James Lovelock, who thinks in terms of planets (Lovelock, 1988), in which case the confused thinking about 'ecosystem services' might not have arisen. Thinking in terms of ecosystems, rather than the biosphere, encourages the idea that services and ecosystems are commodities that can be bought, sold, redesigned, lifted off the shelf or discarded at human whim.

Lovelock produced evidence that the Earth is in a non-equilibrium chemical state compared with what it would be like had living organisms not evolved (Lovelock, 1988; Lovelock & Margulis, 1974). It has far more oxygen and nitrogen, far less carbon dioxide, in the atmosphere, far more water and far less sodium chloride and nitrate in the ocean. It is also much cooler; conditions are maintained for the persistence of liquid water, an absolute requirement for life. Its non-equilibrium composition is maintained by biological processes, apparently within an equable range for survival of their biochemistry. It follows that the nature of the system that these organisms constitute, has meaning. It is a continuum of continually changing communities, tempered to local appropriateness by a ruthless, continuous natural selection (Dawkins, 1986). The continuum of the biosphere (which out of habit, we can call the ecosystems, though these are artefacts of our selective destruction of parts of their former continuum) was thus the most efficient system for maintaining equable (fashionably, now, sustainable) conditions. We cannot improve it and we do not know to what extent we can push the system before its regulatory mechanisms (a requirement for maintenance of non-equilibrium chemistry) act to maintain it in a way consistent with the principle of Le Chatelier (Atkins, 1993).

Points in the continuum are recognizable by a characteristic biological community but its species are (biodiversity) ephemeral, though their trait characteristics are more lasting but also subject to change. At any one time, some species will be in natural decline, others on the rise, in response to change. We waste a great deal of time and effort trying to conserve the former, when we should be concentrating on maintaining the fundamental features of the system: its abilities to use scarce nutrients parsimoniously; its characteristic structure (both physical and trophic (Leopold, 1949; Terborgh, 1988); its interconnectedness; and its need for large extent, which underlies its ability to cope with change (resilience) (Moss, 2007).

We should thus talk of ecosystem services as those that maintain the equable state and rename them 'biosphere services', including all those considered as regulatory. All the rest (provisioning, cultural and supporting) are not biosphere (ecosystem) services at all but commodities provided to humans. The concept of 'service providing units' combines the fallacies of discrete ecosystems in the first place and the idea that we can draw on them more or less at will, so long as we manage them, on the other. Freshwater provides a good example of the problems with this thinking.

Freshwater is not a separate habitat. The hydrological network is like the blood-vascular system of an animal. Its separate existence is inconceivable, a concept understood, through Portia, by William Shakespeare, five hundred years ago (Shakespeare, 1623). The valid unit by which freshwater ecologists work is not the wetted perimeter of a river or lake, but the catchment area and provision of human goods and services can only be seen on that basis. Even this is limited because there is also a meta-catchment provided through the atmosphere that brings substances and inocula of organisms from other catchments, indeed the whole planet. A good example of how consideration can only be at a biosphere level is that of the relationships between Pacific salmon species, bears, rivers and the ocean (Calman et al., 2002). Upland rivers, surrounded by forest have low concentrations of nutrients for the forests also require these nutrients and have evolved mechanisms to conserve the limited supply within the forest system. Salmon grow to maturity in the ocean, concentrating nutrients from a large, equally nutrient-scarce system into their bodies. On their migration to their spawning grounds in the river headwaters, they deliver these nutrients, as carcasses following exhaustion or spawning, to the forests, through the medium of brown bears that scavenge the carcasses. About a quarter of the nitrogen content of the riparian trees comes ultimately from the ocean via salmon, and bear excreta. In turn the riparian trees, delivering debris and leaf material to the rivers, provide mechanisms for retention of carcasses, recycled nutrients, and energy to support the invertebrate community on which the salmon parr feed and surviving individuals of which migrate back to the ocean and complete the cycle.

There are other examples of such linkages emerging (wolves, elk and river structure (Ripple & Beschta, 2004a, b); flooded forest, fish migrations, caimans and successful spawning in the Amazon (Fitkau, 1970)) and they were probably the rule in an undisturbed biosphere. The classic experiments at Hubbard Brook in which the effects of deforestation on sediment load and water chemistry were graphically demonstrated (Likens et al. 1971) also illustrate the folly of seeing freshwater systems and services in separation. Rivers and wetlands do not provide pure water as a ‘service’. Whole catchments, if undisturbed, do.

The extent at which we take goods and services from all habitats, but particularly freshwater ones, because we usually fail to see the importance of connections, is a measure of the damage we do. Damming a river for hydroelectric power interrupts spawning migrations that refurbish upland forest systems with nutrients. Draining a floodplain to use its fertile soils for agriculture obliterates the hydrological regulation the floodplain provided for the entire river downstream. Using a lake for water supply has the ultimate consequence that the water is returned as waste water, loaded with nutrients that impair a system that has evolved characteristics of nutrient parsimony.

Our unfortunate karma is that we have evolved the ability, through our enlarged cerebral hemispheres, to overcome, temporarily, the restraints that maintain an equable system for our biochemistry. Our problem is that, despite our cleverness and huge numbers, we cannot control the colossal amounts of energy and materials involved in the regulatory systems that maintain an equable state on Earth. Our delusion is that we think we can, and one of the ways that we do this is to compartmentalize the biosphere into ‘service providing units’ like a bunch of shopkeepers, when our true nature is that of pirates. A manifestation of this is our inability to control climate change for we do not operate in ways that will be able to do this. For all our cerebral hemispheres, we mostly use the parts of the brain that contribute to immediate survival: gleaning of water, food, shelter and mates, for we evolved by natural selection and our nature is determined by it. The environmental damage that we do is thus inevitable and understandable, for it is what a burgeoning herd of elephants, a swarm of locusts or an invasion of water hyacinth will do.

And as our population and its impact rises, the equal inevitability of the mechanisms that maintain the Earth in a non-equilibrium state, equable for biochemistry, but not specifically for just one species, to move with Le Chatelier may become manifest (Lovelock 2006). As a species yet present for only one four hundredth of one percent of the life of the biosphere, we are truly arrogant.

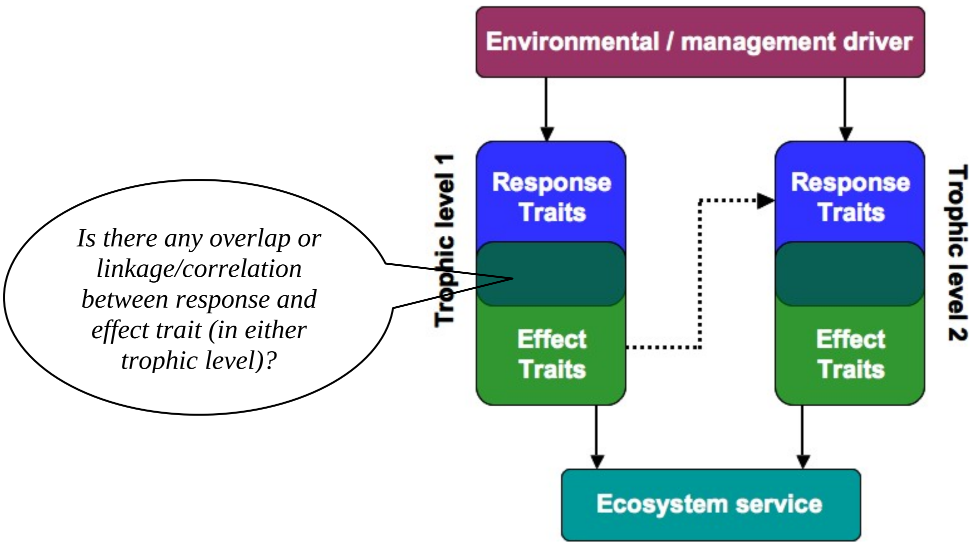
A trait trophic interaction framework for assessing the vulnerability of ecosystem services to changes in environmental pressures

Sandra Lavorel and **Francesco de Bello**, Laboratoire d'Ecologie Alpine, France; **Sandra Díaz**, IMBIV (CONICET-UNC), Argentina; **Jonathan Storkey** and **Richard Harrington**, Plant & Invertebrate Ecology Division, Rothamsted Research, UK

Most ecosystem services depend directly or indirectly on more than one trophic level; in this contribution the authors outline a framework that links response traits and effect traits across trophic levels and outline its use to asses the vulnerability of ecosystem services to changes in environmental pressures.

The RUBICODE project has recently reviewed evidence linking functional traits to ecosystem processes that may be important to service provision. Analysis of nearly 250 published studies showed that a given trait may contribute simultaneously to the control of a range of processes, whilst many traits may contribute to the control of a given process. Most of the work reviewed concerned a single trophic level, but most ecosystem services depend directly or indirectly on more than one trophic level and little has been done to examine links between pressures and processes under such circumstances. The framework pictured (Figure 2) is designed to help.

Figure 2. A trait trophic framework for assessing the vulnerability of ecosystem services to changes in environmental pressures



Response traits are those that determine how organisms respond to a given environmental pressure or how they respond to organisms with which they interact (dotted line). Effect traits are those required for provision of the service, or those that affect organisms in a different trophic level. The hypothesis is that, if linkages can be identified right the way through the scheme, service provision is likely to be affected by the pressure.

Stages in using this framework are as follows. First, within each trophic level, identify response traits to the pressure and effect traits on the service. Then identify traits determining interactions between trophic levels (dotted line) and the processes or properties that are the bases of those linkages. Finally, within each trophic level, analyse the overlap between response and effect traits. There are three possible types of overlap within each trophic level: (i) a response trait may also be an effect trait; (ii) a response trait may be correlated with an effect trait through functional linkage and (iii) a response trait may be linked with an effect trait non-functionally, through phylogenetic constraints.

A given pressure may result in some trait values changing favourably with respect to service delivery and others changing unfavourably. In such cases, the direction of change in service delivery may not be easy to assess. Different pressures may have different effects on traits and services. The framework should be used for one pressure at a time, but a methodology for linking impacts of different pressures will be required.

The framework does not take into account spatial issues. For example, at a local scale, losses of mobile organisms may be compensated by immigration. Thus, it should be used, when appropriate, in conjunction with other theoretical frameworks (e.g. meta-population dynamics). The framework may be adapted to consider different life stages as different levels of analysis. Population-level properties such as abundance and biomass may be necessary to explain service delivery levels and cannot be captured by aggregating impacts on individuals.

We consider that the framework has potential value in conceptualising links between pressures and service delivery, summarising knowledge, testing hypotheses and identifying missing data. It could be used to compare the impact of pressures under contrasting conditions (e.g. high and low fertility) and may aid identification of key traits as predictors of service delivery levels.

As yet, there are very few published studies where the linkages in the framework are described, and we would be grateful for examples.

Approaches to ecosystem service assessment in grassland ecosystems

Winfried Voigt, Institute of Ecology, University of Jena, Germany

Grasslands represent an important and common type of ecosystem in Europe. Even though there has been extensive research on grasslands, there is still a great need for more accurate information about how to quantitatively link ecosystem services (ES) that are provided with Service Providing Units (SPUs). Biodiversity research of grasslands is in principle a good starting point for ES approaches (or studies). Grasslands and the ES they provide can sometimes conflict with forests and their ES. Some issues in the most significant ES in grasslands are touched on which show deficits and are referred to in recently written literature.

Grasslands are estimated to cover about 7 million km² throughout Europe, of which only 3.5% are designated as conservation areas. Semi-natural grasslands are usually species-rich ecosystems. They are an inherent and often unique feature of many European landscapes. Except under extreme (negative water balance), harsh conditions, grasslands are transient man-made ecosystems kept in a productive state by permanent or periodic management. With land-use changes, climate fluctuations and increased air-borne nitrogen input, amongst other factors, there is a dramatic turnover and loss of species. This is followed by fragmentation or perhaps a general decrement of grassland areas negatively affecting ES typical for grasslands. Abandoned grasslands previously managed in various ways promptly undergo ecological successions resulting in species turnover, species loss and, eventually a total loss in area.

Grasslands respond and alter promptly as a result of disturbance or change in management. On the other hand (and fortunately) after the initial conditions have been restored they are able to recover or regenerate quite quickly (Heinrich et al. 2001). So, because of its general and regional importance as well as its fast response to changes, but also because of its comparatively simple and small spatial structure, it is not surprising that extensive ecological work has been done on various grasslands in Europe but also in many other parts of the world. There is actually a lot of data and knowledge accumulated to date. Nevertheless, there are still many issues about the role of biodiversity in grasslands and a lack of understanding of the mechanisms behind them in connection to the maintenance of the ESs they provide. They are going to be answered step by step in a couple of recent experiments (e.g. see Roscher et al. 2004).

Some issues concerning the most significant ES provided by grassland ecosystems:

1. Provisioning services

Provision of fodder for livestock: This service has clearly been the most important one to human societies for a long time but it is also the most effective way to maintain grasslands. Reduction or the fragmentation of fodder by various processes can lead to economic inefficiency and therefore cause a change in management or even lead to abandonment. While in the past the enrichment of the most productive plant species was considered to cause an increase in total productivity in some grasslands, it turned out that species rich grasslands produce permanently more and, importantly, more reliable biomass than would be produced with less diversity (Tilman et al. 2006, Fargione et al. 2008). Species turnover can cause species compositions with lower numbers or lower biomass of palatable plant species. There is a strong need for defining specific SPUs and general measures for the amount or a quantitative value of the palatability (a feeding value index) in grasslands for specific livestock.

Reservoir and refugium for natural enemies of agricultural pests, and for (“wild”) pollinators: Until now there have been no convincing studies to show quantitative relationships between beneficial invertebrates in adjacent grasslands, crop and fruit yield in agricultural fields or orchards. Frameworks and models to quantify such effects need to be developed.

Providing clean freshwater: While forest areas are usually considered to be the main provider and regulator of freshwater, recent studies show that upland or mountain grasslands, in particular, increase water yields compared to forested areas (Viviroli 2003, Mark and Dickinson 2008) due to less transpiration of grass vegetation. This (providing) function of ES in grasslands needs to be integrated into economic models and decision frameworks. The South African National Water Act of 1998 is a recommended model (Macdonald 2004, Görgens and van Wilgen 2004).

2. Regulatory services:

Control of soil erosion: Since extreme weather events are predicted to become more frequent, the value of this ES will probably increase in hillside locations and other sloped areas over the coming decades. Changing management and decreased diversity can reduce sward density or even create open gaps (Heinrich et al. 2001) and so reduce the resistance against soil erosion. There is a lack of quantitative information (quantitative species composition) to link SPUs with this ES.

Most but not all ecosystem services provided by grasslands have been proved to be related to plant and consumer species diversity. The conservation of species rich grasslands often conflicts with the expansion of forests and the services that they provide. Here a concerted debate of scientists, stakeholders and politicians will be necessary to decide which priorities are most significant and to find societal consensus.

Ecosystem service assessment and landscape dynamics

Bruce Jones and **Carl Shapiro**, U.S. Geological Survey, Virginia, USA

Considerable research is needed on horizontal interactions and scaling functions of biotic, abiotic, and human factors that influence ecological processes and services at multiple landscape scales.

Landscape composition and pattern strongly influence fluxes and flows of the four primary ecological elements: water, nutrients, biota, and nutrients and materials (Turner 1989). These fluxes and flows in turn determine the quality and diversity of ecological services derived from a landscape, catchment, or river basin (Rapport et al. 1998). The spatial intersection of biotic (vegetation) and abiotic (soils) factors often determine the quality of any specific service provided in the landscape. The cumulative fluxes and flows associated with entire basins often influence the quality of and impairment to estuaries, lagoons, and near-shore habitats (Basnyat et al. 1999). The landscape matrix determines the effectiveness and importance of the individual biotic components rather than simply adding up the individual components to obtain a range of benefits (Ricketts 2001). Yet the position of the landscape elements within the matrix is also important. For example, forests located along stream margins may yield greater benefits for water related services than forests in upland areas (Jones et al. 2006). Horizontal flows and fluxes are also influenced by the position of biotic and abiotic elements in the landscape (Reiners and Driese 2001, Urban and Keitt 2001, Ludwig et al. 2005).

Landscapes comprised of relatively large amounts of natural vegetation tend to maintain greater variety and quality of ecosystem services, primarily because they tend to reduce energy from wind and water, increase water filtration, maintain soils (nutrients, elements, biota), maintain native habitats for terrestrial and aquatic species, increase photosynthetic capacity, and resist invasive species establishment (Turner et al. 2007). Conversely, landscapes with large amounts of anthropogenic cover tend to lose their capacity to (1) filter nutrients and contaminants from water, (2) abate flood waters associated with extreme climate events, (3) retain water, soils, and nutrients, (4) resist invasive species establishment, and (5) provide for natural predators of pests (Turner et al. 2007). The key issue is the degree to which people and communities can be distributed within landscapes without impairing or losing important ecosystem services.

A landscape perspective (spatial composition, pattern, and position) also provides for a common framework to evaluate social, economic, and cultural dynamics and their relationship to ecological services (Naidoo and Ricketts 2006). Economic conditions themselves can predict landscape patterns of plant species diversity (Wamelink et al. 2003). However, considerable additional research is needed to understand linkages between ecological services and socioeconomic, demographic, and cultural drivers of landscape change at multiple scales (Hein et al. 2006).

Current efforts to map ecological services tend to be vertical in nature where spatial data are intersected in a geographic information system to evaluate an individual grid cell or area’s relative importance for multiple ecological services (Troy and Wilson 2006). This approach ignores important horizontal interactions among landscape components and differential importance based on position in the landscape. Moreover, different ecological processes and associated services for any given area on the landscape are linked to different functional units and spatial scales, such as water flows through catchments, terrestrial biota through vegetation matrices, ground water through aquifers, and air through airsheds. Therefore, the importance of an area or habitat patch for a particular service may vary, depending on its position within a specific functional unit. Finally, the scale of these functional units will determine the magnitude of change of a particular service for a specific area given changes in broad-scale environmental drivers such as climate. For example, there is a 50% chance of Las Vegas, Nevada losing most of its water supply by 2021, due in part to climate change on the upper part of the Colorado River Basin (Barnett and Pierce 2008).

Research is needed to increase our knowledge of how horizontal interactions across multiple landscape scales influence ecological services. Such an approach is important in deploying spatially explicit options for biological traits to optimize ecological services and in adapting to broader-scale drivers that might influence a range of services (e.g. climate change). Moreover, mapping approaches that capture process- and service-specific functional units will enhance ecological service assessments, restoration, and conservation.

Ecosystem services from agriculture

John Porter and **Lene Sigsgaard**, Faculty of Life Sciences, University of Copenhagen, Denmark.

Ecosystem services (ES) from agro-ecosystems have hitherto been assigned relatively low values partly due to lack of data, but it is difficult to see how global ES can increase without significant improvements in ES from farming, given the proportion of the Earth's land devoted to agriculture and because its ES provision has been driven to a low level.

Ecosystem services (ES) are the benefits humans derive from ecological processes and ecosystem function. By recognizing the value of ES we accept that our largely non-marketed ecological wealth underpins our marketed economic wealth. Agricultural ecosystems produce food, fibre, and non-marketed ecosystem services (ES). Agriculture typically also involves high negative external costs associated with, for example, fossil fuel use. Estimation of ES from agriculture has shown benefits from low-input systems (Singh et al, 2008) but ES from agriculture have hitherto been assigned relatively low values (Costanza et al. 1997, Bjorklund et al. 1999) when compared with other terrestrial and aquatic ecosystems, partly due to lack of data. However, agro-ecosystems cover between 28% and 37% of the Earth's land surface and are divided about 70:30 between pastures and crops. Although agricultural ecosystems may have low ES values per unit area when compared with other ecosystems such as estuaries and wetlands (Costanza et al. 1997), they offer the best chance of increasing global ES via definition of appropriate goals for agriculture and the use of land management regimes that favour ES provision. Agriculture can thus be considered the largest ecological experiment on Earth with a large potential to damage global ES but also to promote them via ecologically informed approaches to the design of agro-ecosystems that value both marketed and non-marketed ES (Bjorklund et al. 1999, Porter 2003).

Therefore research is needed to gain a more accurate estimate of the ES from agriculture and to examine alternative methods of valuation and to establish the connection between farmland design, management and ES. This includes the level of ES provision achievable by agriculture as governed by the intensity of use and diversity of crop land. It is also important to examine the ES from agriculture at a variety of scales and to link such data to GIS. Other issues are how and the extent to which agricultural systems at the local, country and European regional scale can provide ecosystem goods and services, how the value of such services may change with land-use over time and how far agro-ecosystem services are reflected in the current level of societal support given to European farming.

Session II: Drivers and scenarios for ecosystem service assessment

Introduction to Session II: Drivers and scenarios for ecosystem service assessment

Mark Rounsevell, Centre for the study of Environmental Change and Sustainability, School of Geosciences, University of Edinburgh, UK

Change on earth has been taking place for billions of years, but of new concern is the scale, the magnitude and the speed at which change in ecosystems has been occurring since the industrial revolution and more recently over the last sixty years. It is important to analyze and, as much as possible, quantify the importance of ecosystems to human well-being in order to make better decisions regarding the sustainable use and management of ecosystem services (Millennium Ecosystem Assessment, 2005). But, to prevent and reduce further environmental degradation, it is important to understand how and why change is occurring. The identification of environmental change drivers is an important first step in understanding more about the 'how' and the 'why'. Drivers represent the underlying causes of environmental change and encapsulate the ways in which people live their lives. This includes social, economic, technological, political, policy and governance factors, changes in all of which will, individually and in combination, affect the world around us.

The RUBICODE project has set out to review existing knowledge about the drivers of ecosystem change. The review demonstrates that the majority of studies focus exclusively on one spatial scale, in spite of the axiom that drivers act differently at different spatial (and temporal) scales. Demography is the most frequently cited indirect driver of environmental change, with land use and land cover change, and climate variability and change the most commonly cited direct drivers (or pressures). Natural, physical and biological phenomena, diseases and wars are the least discussed direct drivers. The review highlights the problems that arise from the use of different terminology in describing similar or even identical concepts. A commonly accepted definition of the notion of a driver simply does not exist. Better definitions and the standardising of terminology would help to reduce confusion and facilitate the rapid exchange of comparable information.

Whilst we have some understanding of the types of processes that affect ecosystems, we are simply unable to know what the future has in store for us: we do not have a 'crystal ball'. This uncertainty about the future derives from the basic complexity of social systems and our lack of insight into the fundamental laws that govern human behaviour and development. Faced with this uncertainty, scenarios provide an opportunity to explore alternative, but plausible futures, as a combination of assumptions about interacting drivers. We can use scenarios to ask 'what if?' questions, but not to make predictions.

The participants at the recent RUBICODE Workshop held in Helsingborg, Sweden (25-28 February 2008) raised a number of important points about the role of drivers and scenario development in studies of ecosystem service provision. These included:

- The need to promote consistency in the definition of system boundaries (and the associated establishment of exogenous drivers and endogenous pressures);
- Identification of those components of scenarios where uncertainty can be quantified and for which variables have high or low uncertainty;
- Development of participatory approaches to scenario construction that build on a range of stakeholder perspectives and policy relevance;
- Development of scenarios of drivers/pressures that affect ecosystem service beneficiaries;
- Development of conditional probabilistic futures for different sectors;
- Development of shock or 'wildcard' scenarios as explorations of extreme events and 'surprises'.

This raises a number of further questions. Are we able to develop probabilistic futures? How can we account for the uncertainties of human actions and behaviour? Does the inherent unpredictability of the future make our efforts worthless? These and many other questions will be the subject of this e-conference discussion.

The session is based on keynote contributions from a number of international experts on the subject of drivers and scenarios in ecosystem service assessment. Topics to be covered include: participatory approaches to scenario development, institutional change and the flow of ecosystem services, lifestyle drivers and the consequences for human well-being and quality of life, technology as a driver of ecosystem service change, governance and the importance of addressing ecosystem service beneficiaries, probabilistic approaches in population projections, innovative approaches to drivers of ecosystem service assessment and change. We hope that this collection of contributions will both stimulate debate and raise awareness about the issues surrounding drivers and scenarios. The keynote presentations may be contentious, but this is intended to provoke a reaction. We hope to use the e-conference discussion as a means of identifying new research directions and fostering the development of a research community with interests in drivers and scenarios for ecosystem service provision. We want to find new ways of thinking about complex problems, and your contributions to this process are appreciated. In the words of Marcel Proust, “the voyage of discovery is not in seeking new landscapes, but in having new eyes”.

Participatory approaches to scenario development and their relevance

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This contribution outlines the structure behind participatory approaches and discusses the potential benefits of the involvement of scientists and non-scientists when considering scenarios for ecosystem service assessment.

Participatory approaches to scenario development are often advocated with two main arguments:

1. Tapping relevant knowledge and experience of stakeholders allows for a broader, more innovative analysis of alternative futures.
2. Involving stakeholders fosters the buy-in and acceptance among potential end-users. But it can also trigger wider (social) learning processes and provides the grounds to reveal conflicts, exchange information, build mutual trust and form consensus over controversial issues.

One way to distinguish the variety of approaches is by classifying them according to the depth of stakeholder involvement and to the degree of using formalized methods to frame discussions (Volkery et al. 2008):

- Stakeholder involvement can be purely consultative. In the co-design approach stakeholders have their say on the design of the process. Within co-decision approaches, stakeholders and scientists both determine process and content. In the full-decision approach stakeholders are fully responsible.
- Formalized methods such as detailed questionnaires, information matrixes or agent-based modelling elicit information effectively, but don't allow for much discretion. Non-formalized methods such as open interviews or open-space workshops allow for more leeway. They are more frequently employed in co-design, co-decision- or full-decision modes.

The level of involvement and choice of methods should be based on the primary purpose of the scenario development, which can be distinguished into a) scientific exploration and research, b) public outreach and learning and c) decision-support and strategic planning (Alcamo and Henrichs 2008):

a) If scenarios should be relevant for this purpose, they need to be perceived of as scientifically credible. This requires a structured, formalized approach. Participation of stakeholders can broaden the information basis, but will not leave the consultative modus.

b) For public outreach and learning, the openness and importance of the process is of prime concern. Ideally, a broad range of stakeholders should be involved from an early stage to create relevant scenarios, employing at least co-decision approaches and rather non-formalized methods. Scientists are in the role of providing input.

c) In order to be relevant for decision-support and strategic planning, scenarios also need to be perceived of as significant, but more importantly, as legitimate. Who has been involved in the development becomes important. Due to the prevalent time constraints within the policy business, it is highly desirable to clearly highlight the time requirements and organize high-level political backing at the beginning of the process.

Before starting a participatory process, it is useful to carefully consider why and how stakeholders should be involved, who represents a core target audience, what their interests are and what is expected from them to avoid stalemate and frustration?

Roles and responsibilities need to be clearly defined. Especially if scenarios are meant to support public outreach or decision-support, one should resist the urge to develop the scenarios around the expertise of the experts only. There is, however, no standard recipe for organizing such a process. Much of it depends on the actual context, the people involved and the availability of resources. In a nutshell, a process would start by establishing a core scenario team that takes overall responsibility. This team should include some of the key stake-holders. A scenario panel should then be established that comprises of all participants and experts. The role of the panel is to identify the main concerns and questions about the future, discuss drivers of change and uncertainties and to rank them according to their importance. It is important to apply methods that encourage long-term creative, "out-of-the box”-thinking. This can include provocative input from non-experts or even role-plays. Scientists should provide input, clarify questions and monitor the consistency of assumptions. The panel generates alternative scenario storylines, often in an iterative process. If a qualitative-quantitative approach has been decided on, this activity needs to be coupled with model runs. Such combined approaches are more difficult to manage, but they also promise greater returns in terms of comprehensive, scenario-rich narratives. They provide better grounds for insightful analysis and scenario comparison (MA 2005, EEA 2007).

Institutional change as a key driver in the transformation of the flows of ecosystem services

Erik Gómez-Baggethun, Department of Ecology, Universidad Autónoma de Madrid, Spain and **Eszter Kelemen**, Department of Environmental Economics, St. István University, Hungary

We discuss the need for research to further understand the existing links between institutions and ecosystem services to support scenario planning. In particular, we focus on how institutional disruption can act as a key driver of change in the flows of ecosystem services.

Scenarios are plausible sets of stories about how the future may unfold. Changing human conditions drive changes, both directly and indirectly, in biodiversity, in ecosystems, and in the services ecosystems provide. Identifying drivers of change is a central issue in developing scenarios for ecosystem services (ESs) as these are the key factors, trends and processes required to increase the probability of possible outcomes to the stories (Gallopin and Rijsberman, 2003). Millennium Assessment scenarios suggest that the ongoing trend of deterioration of ESs could be mitigated through substantial changes in policies, practices and institutions (Carpenter et al. 2005). However, although it is sometimes addressed within the category of indirect socio-political drivers, institutional change is rarely mentioned as a significant driver itself.

Institutions, understood as formal rules, norms and conventions structuring social interaction, regulate the relationships between social and ecological systems (Vatn, 2005; Ostrom, 2006), and determine how biodiversity and ecosystem functions are managed to obtain ESs. Due to non-linear dynamics in socio-ecological systems, resource regimes (institutional guidelines for resource use) can abruptly change from one state to another, leading to changes in the flows of ESs within short periods of time. These changes take place when critical thresholds are reached either as a consequence of cumulative pressures or through sudden disruptions caused by inner or external agents, such as market pressure, state intervention or natural disturbance.

Interventions from large scale organizations can rapidly break down locally developed institutional guidelines, leading to new resource regimes being established that prioritise different sets of ESs to be used or protected. Case studies on top-down declaration of protected nature areas in Spain and Hungary show how resource regimes, consisting mainly of informal norms and conventions based on locally developed traditional knowledge, were displaced in a short period of time by formal rules based on scientific knowledge (Gómez-Baggethun and Kelemen, in press). This abrupt change led to different criteria and priorities for resource use and conservation, inducing changes in ecosystem service flows. Ecosystem service demands at national and international scales, such as tourism or scientific research, were enhanced at the expense of services that were valued by local stakeholders, such as harvesting and hunting. Biodiversity conservation priorities moved from hunting species needed by local inhabitants, towards key species required by the society at large.

Europe is currently undergoing a process of deep institutional change. As a consequence of integration in the EU, many countries in Europe are facing institutional change for resource use, which will have impacts on biodiversity and ecosystem services. We argue that further research is needed for assessing the links between institutional change and ecosystem services, an issue that has attracted limited attention from research agendas.

Key challenges which need to be addressed in order to fill this gap in our knowledge include:

1. Using case studies that identify specific links between institutional change and ESs flows
2. Developing a consistent theoretical framework and methodological guidelines that help to identify local institutional guidelines that are successful in the maintenance of biodiversity and key ESs
3. Exploring how institutional diversity for ESs management can be maintained and promoted in the ongoing process of European environmental policy integration

4. Analysing how institutions at local, national and international scales can complement each other within multi-level management systems to ensure the protection of ESs are valued by stakeholders at different scales.

The interrelations between lifestyles, ecosystem services and quality of life

Ines Omann and **Lisa Bohunovsky**, Sustainable Europe Research Institute, Vienna

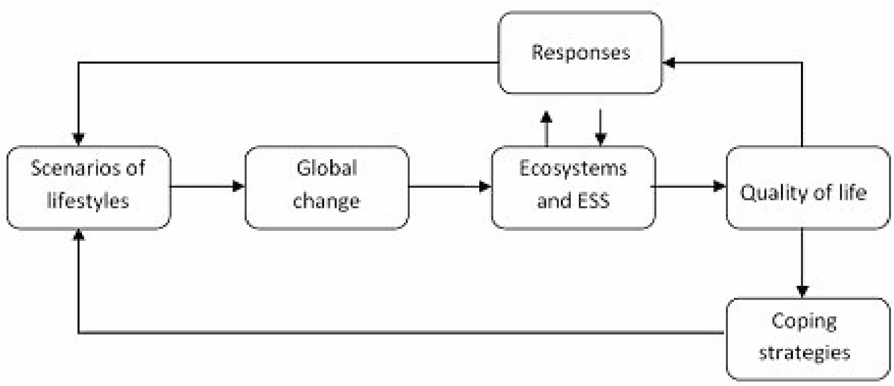
The aim of this keynote is to show the inter-linkages between socio-economic and natural systems, in particular between scenarios of lifestyles (drivers), their influence on global change (pressure), the consequent changes of ecosystems (state) and subsequent change of ecosystem services (impact on human well-being) and societal responses (policies; changed lifestyles due to reduced quality of life).

The DPSIR scheme, developed by the EEA and used quite widely in the fields of global change, ecosystem research and sustainable science, seems appropriate to show how lifestyles, global change, ecosystem services and quality of life are interlinked.

Quality of life comprises of various areas, (such as material wealth, health, social relations, safety, access to resources, freedom, spirituality) and is influenced by different interacting factors (such as environmental change, change of social relations, changes in financial circumstances etc.; see UNEP, 2007; Porritt, 2007). The European sustainable development strategy names the maintenance and improvement of the quality of life as one of the aims for sustainable development: Sustainable development "…aims at a continuous improvement of the quality of life and well-being on Earth for present and future generations" (EU SDS 2006).

Human lifestyles are influencing natural systems and can lead to global change, such as climate change: In our modern societies, a good life is mainly defined in material terms. Modern lifestyles lead to consumption patterns, which are a main driver for global change as they result in increased greenhouse gas emissions induced by use of resources and energy. We know that climate change has a strong influence on ecosystems and biodiversity (this is one of the reasons why the project RUBICODE was initiated). The change of these systems leads to changes of ecosystem services (ESS) used by humans (see for instance Anastasopoulou et al., 2008; Harrison et al., 2006; CBD, 2003; IPCC, 2002; Spangenberg, 2007; Thuiller et al., 2005). A reduction or elimination of ESS threatens human well-being and in some cases even human life per se. Thus, modern lifestyles put pressure on natural and socio-economic systems (i.e. on quality of life) via global change. The impacts differ over spatial scales and require appropriate strategies (responses). Climate change is thus in turn influencing quality of life (see Figure 3).

Figure 3. Interlinkages between lifestyles, global change, ESS and quality of life



There are a growing number of people, who consciously lead other, less consumptionoriented lifestyles, trying to keep their resource and energy use as low as possible. Research results show (Hofstetter et al. 2005; Jackson 2005) that sustainable lifestyles and thus sustainable consumption patterns are directly linked to life satisfaction. Thus by changing human lifestyle's to be more sustainable can not only directly influence one's happiness, but also indirectly influence it via natural systems and ESS, because of the strong links between these issues.

Summarising, I would like to bring the following two hypotheses into the discussion:

- Global change has a direct impact on humans and their quality of life, for example by changing or reducing the number and quality of ESS or through changes of social relations. Depending on the vulnerability of different population groups, the strength of the impacts can vary.
- Different lifestyles contribute differently to global change through different consumption patterns. Those whose lifestyles contribute most to climate change (North-Americans, Europeans) are often less vulnerable to climate change, whereas those who live life with a low ecological footprint (i.e. using few resources and land) are often most affected (Indians, Bangladeshi, Inuit). However the first group is not necessarily happier than the second.

Technology as a driver of ecosystem service change: assumptions made in developing technological change scenarios

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The importance of technological developments in economic and environmental scenarios is discussed, with particular focus on the agricultural industry.

Ecosystems provide essential services to nature and society but are affected by changes in natural and socio-economic conditions. Understanding and assessing changes in ecosystems and related service provisions is therefore of vital interest for people and society (Schroter et al., 2005).

Agricultural production determines a large part of the terrestrial biosphere. Primary services provided by agriculture are the production of food, feed (for animals) and energy (e.g. biofuels) as well as income to farmers to maintain their livelihood. In addition, the role of agriculture within the wider context of sustainable development including effects of agricultural production on biodiversity, ecosystems' functioning and landscape development is increasingly recognized.

Assessing changes in agricultural production, including land use, requires firstly identifying the drivers of this change. A set of drivers that affect agricultural production have recently been identified (Rounsevell et al., 2005; Ewert et al., 2006). These can be grouped into natural (e.g. soils and climate), socio-economic (e.g. technological development and management, population growth, consumer preferences) and policy (e.g. environmental policies, market interventions) related drivers.

Analysis of historic data and future projections (based on the IPCC SRES scenario framework) have shown that from all the drivers considered changes in technological development had the largest impact on food production and land use (Ewert et al., 2005; Rounsevell et al., 2005). Technological development was considered to be particularly large for economic scenarios and less for environmental scenarios. Intensification (improved crop management and use of fertilizers and pesticides) and breeding for new varieties (including GMO crops) were assumed to further increase current productivity rates (Ewert et al., 2005). Associated effects on land use were large with a projected decrease in agricultural land use by more than 50 percent within the next 80 years (Rounsevell et al., 2005). This was particularly interesting as some of this land would be available for other use such as nature conservation, recreation, biofuel production etc. (Rounsevell et al., 2006) with potential benefits for sustainable development. Technology development and changes in productivity and land use were less pronounced in the environmental scenarios with more emphasis on the sustainability of agricultural production.

The advantages and disadvantages of the assumptions about technological development within the context of sustainable development in environmental and socio-economic terms could not be assessed and may be subject to further discussion. Also, the assumptions about technology development were made without considering feedback mechanisms. It remained unclear what exactly drives technology development. Free market scenarios may not necessarily push technology development to the extent assumed as markets will regulate production according to demand without overproduction. Another important issue is the increasing demand for biofuels which starts to affect food production and may drive technology development and land use change in the future.

Clearly, drivers affecting ecosystem service change can be very dynamic and are often difficult to estimate. Drivers will evolve interactions with other drivers and through feedback with the ecosystem(s) and the services provided. Technology development has been identified as an important driver of agricultural production and land use and assumptions about its change are crucially important for future projections. The present note attempts to encourage discussions about the role of technology development for ecosystem service change and the assumptions underlying the projected development of technology.

Unclear multi-level governance schemes as drivers for biodiversity loss

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In this contribution the author outlines multi-level governance schemes and highlights the importance of understanding the social processes underlying the production of socio-political scales and the need to analyse how these social-spatial dimensions are produced by social processes in order to fully grasp cross-scale interactions.

In recent years, scholars in environmental research have increasingly discussed the appropriate scale of environmental assessments and of government strategies capable of addressing environmental problems. Although some advances have been made, recent discussions still lack an awareness of how social scales are produced and which power relations are involved. Including the driving effects of, for example, EU decisions on biodiversity in the Western African coastal ecosystems, is a first step. Without understanding the social processes underlying the production of socio-political scales, a remedy with political impact might not be found.

Trans-boundary environmental problems raise questions of how to connect the scale, the interactions and the fit of environmental regimes and institutions (Young 2002).

During the last decade, Multi-Level Governance (MLG) has become a buzzword, and not only for environmental policy. Influenced by previous research on federalism, this new form of political steering (Hooghe and Marks 2001; Heinelt et al. 2002; Bache and Flinders 2004) became paradigmatic for the European integration process and decision-making in that supranational entity. If EU environmental policy represents a 'unique system of multilevel environmental governance' (Jordan 2005: 2), it is questionable whether the EU model is transferable to other regions.

Moreover, in research about environmental governance, in the EU and beyond, the term 'level' denotes existing institutional systems or procedural processes at specific spatial dimensions such as international or supranational institutions (MEA, EU-commission etc.), national authorities or democratic institutions (for example. national parliaments), or local decision-making processes. The different levels are simply taken as given! The production of these spatial levels – e.g. the production of Europe in a historical process – and its meaning for the relationships between the different levels is regularly excluded. Analyses therefore underestimate or often simply neglect the processes of up-scaling and down-scaling of decision-making through the strengthening or weakening of existing levels and/or the construction of new levels (Görg and Rauschmayer, in preparation).

Nevertheless, in their present form, the policy options, for example those discussed within the Millennium Ecosystem Assessment, do not acknowledge that the societal externalization or 'misplacement' of environmental effects is a specific 'beyond-the-border' or 'trans-local' environmental problem of particular importance to industrialized countries. The term 'misplacement' addresses the driving function on biodiversity loss in other parts of the world while providing human well-being for a particular society in a specific region or nation state. These misplacements, emphasizing the relevance of cross-scale interactions and the power relations involved for governance strategies at the regional and/or local level, cannot easily be avoided. To fully grasp these cross-scale interactions, it is necessary to analyze how socio-spatial dimensions are produced by social processes, rather than dealing with them simply as givens. Otherwise, proposed policy options will not be able to really deal with this important driver for global biodiversity loss, i.e. decisions taken on different levels in industrialized countries.

Scenarios of demographic change: the role of probabilistic approaches in population projection

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This brief note first summarizes the four main demographic trends that are expected for the coming decades, and discusses two techniques that demographers use for mapping future population trends, namely scenarios and probabilistic projections.

Demography is an important driver of changes in ecosystems. Individuals and households use scarce resources such as fuel, water, and arable land, thereby contributing to biodiversity loss and other environmental changes. Good policy prepares for undesirable future developments, and takes action when needed and possible. Thus it is important to analyse the demographic future at the global, regional, national, and sub-national scale.

Four major demographic trends:

1. Population growth rates are falling – most growth in Less Developed Countries
The United Nations expect continued population growth to 2030, but the global annual growth rate is predicted to fall from 1.1 per cent on average in 2000-2015 to 0.8 per cent in 2015-2030 (see UN's 2006 Revision of World Population Prospects). More developed countries will show hardly any growth (many countries in Eastern and Southern Europe will actually experience falling populations), while the rates in Less Developed Countries are predicted to fall from 1.3 to 1.0 per cent for the two periods. Decreasing birth rates and, to some extent, AIDS-related deaths are important for slowing population growth.

2. Smaller households: Numbers of households are increasing faster than population numbers. A UN-Habitat report of 2001 expected that the number of household's world wide will increase by 2.2 per cent and 2.1 per cent over the periods 2000-2015 and 2015-2030, respectively (see UN's Cities in a Globalizing World: Global Report on Human Settlements 2001). Even with constant population size, more households imply higher per capita demand for energy, water, arable land, etc., due to economies of scale.

3. Ageing: The populations of a growing number of countries are ageing rapidly. Between 2005 and 2050, half of the increase in world population will be accounted for by a rise in the population aged 60 years or over, whereas the number of children (under the age of 15) will decline slightly. Furthermore, in the more developed regions, the population aged 60 or over is expected to nearly double whereas the percentage of people under the age of 60 is likely to decline. The subsequent implications for environmental change are unclear. Elderly consume less, but they live in smaller households, compared to other age groups.

4. Urbanization: The 2007 Revision of the UN's Urbanization Prospects corroborates that the world population will reach a landmark in 2008: for the first time in history the urban population will equal the rural population of the world and, from then on, the world population will be urban in its majority. Nevertheless, major parts of the world remain largely rural, in particular in Africa and Asia. Between 2007 and 2050, the world population is expected to increase by 2.5 billion, increasing from 6.7 billion to 9.2 billion. At the same time, the population living in urban areas is predicted to gain 3.1 billion, increasing from 3.3 billion in 2007 to 6.4 billion in 2050. Thus, the urban areas of the world are expected to absorb all the population growth expected over the next four decades while at the same time drawing in some of the rural population. Most of the population growth expected in urban areas will be concentrated in the cities and towns of the less developed regions. This rapid urbanization implies less self-sufficiency and increased transportation demand in more densely populated areas.

Two techniques for mapping demographic futures:

The demographic future of any population is uncertain, but some (of the many) possible trajectories are more probable than others. The United Nations expect a world population equal to 7.3 billion in 2015, up from 6.7 billion in 2007. 7.3 billion is just one possible – but plausible - number out of many, yet a population around 7.3 billion is much more likely than one around 3 billion, or 13 billion, say. A probabilistic population projection tells us how much more likely, because such a projection gives the future population as a probability distribution. Hence to every range of possible outcomes (e.g. 5-9 billion for world population in 2010) is attached a certain probability (e.g. 60 per cent). The probability that world population will equal 7.3 billion (i.e. between 7.25 and 7.35 billion) is extremely small.

Demographers and statisticians have developed methods for computing probabilistic projections, because the traditional deterministic scenario approach does not yield an accurate picture of forecast uncertainty. A traditional deterministic population projection starts from a recently observed age pyramid, usually taken from a census. Assumed rates for mortality and fertility project that age pyramid one year into the future. Separate assumptions for migration flows broken down by age and sex complete the projection for the first year. Repeated application of this procedure for later years takes the population into the future as far as one wishes.

The traditional approach deals with uncertainty by formulating more than one path for future fertility - usually a high, a medium, and a low variant. For instance, the UN assumes in their medium variant that world fertility levels will eventually converge towards 1.85 children per woman on average. High and low variants are based upon ultimate fertility levels that are half a child higher or lower, respectively. The latter two variants result in world population sizes of 7.5 and 7.1 billion in 2015. Projection results for various age classes, distinct countries, and other projection years also differ between projection variants. Mortality and migration variants are also commonly used.

The correct view is to consider these projection variants as purely conditional ‘what if’ computations. Modellers use such sensitivity analyses to trace important variables, but they are of little help to users and policy makers, because the probability that one or the other variant will materialize is close to zero. Moreover, the gap between the high and the low variant gives a false impression of forecast uncertainty, because each variant assumes perfect serial correlation for the birth rates.

Probabilistic projections are based on randomly varying birth and death rates and migration numbers. Uncertainty parameters for the random processes are estimated from errors observed for historical projections and time series analyses, possibly modified by expert judgement. These parameters quantify uncertainty correctly, and force the user/policy maker to consider a whole range of outcomes in the form of a probability distribution. This is in contrast with the traditional scenario approach, where one or just a few outcomes are selected, often based on vested interests.

A probabilistic projection requires long time series for fertility, mortality, and migration, which are of good quality. Thus in practice, probabilistic projections are limited to industrialized countries. Techniques for computing probabilistic household projections are only just emerging, as these pose even higher data demands.

Changes in urban and peri-urban land use: approaches and problems of assessing drivers and scenarios

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In this contribution the author outlines factors causing changes in land use which in turn cause changes in ecosystem services and functions. The author highlights the need to develop downscaling procedures for scenarios in order for them to be applied at different scales.

Changes in land use generally cause changes in ecosystem services and functions (Costanza et al. 1997 Lambin/ Geist 2006.). Recent research has increasingly focussed on land use changes between urban and rural regions. An urban-rural region is a system of functions, resulting in patterns of activities that create certain land uses in response to those activities. The balance between open space and urbanisation, effects interlinking agro-ecosystems and urban land uses; this is increasingly discussed in terms of the provision of ecosystem services (Loibl/ Köster 2008). One reason why land use change in urban-rural regions often proves distinct and highly dynamic is due to the plurality of land uses along urban fringes. It is connected with a plurality of mutual demands and supplies, services and functions, therefore indicating that it is sensitive to a high number of driver-pressure relationships.

Demography and global economic trends are considered to be the most relevant drivers of land use change, and specifically affect urbanisation and counter-urbanisation. They are a result of activities related to space consumption: production/work, transport/commuting, housing, water and food supply, lifestyle, and recreation. Beyond regional characteristics, land use change is driven by planning policies and governance, environmental quality, climate change, social issues (such as migration), technology and accessibility. Those regional characteristics and different types of urban-rural patterns provide different spaces of choice for land use changes and accordingly show different driver-impact relationships. This simple insight has challenging implications for research on land use change impact assessment.

Information on future trends and impacts or changed driver groups is usually derived by implementing scenario settings to economic, bio-economic, environmental, demographic etc. modelling procedures. Demography and global economic trends are usually modelled by implementing scenario-based assumptions at European or national (NUTS1) levels. For other drivers named above, either national or regional characteristics, like geo-morphological and bio-physical site conditions, socio-cultural background, or legal frameworks are crucial factors, that have to be considered at lower spatial levels: national, regional, related to the settlement pattern or the population density. Hence, it is essential to develop downscaling procedures for scenarios.

The grouping of regions, based on the degree of sensitivity to certain drivers or scenario storylines was applied in several projects: ESPON 1.3.1 (Schmidt-Thomé 2006), Scenar 2020 (Nowicki et al. 2007), SENSOR (Stuczynski 2007). Using a rural-urban-region (RUR) typology, related to settlement morphology and applied at the EU-27 at NUTSX level (mix between NUTS2 and NUTS3 (Alterra 2005)), Zasada et al. (2008) deal with two different elements of regionalisation or downscaling for scenario modelling outputs: On the one hand drivers are treated as determinants differentiating a European entity into Regional Clusters and hotspots of driving forces, showing either macro-regional clustering effects (e.g. migration movement pattern, pattern of national administrative and planning systems) or hotspots with structural impacts (from climate, technology), depending on scenario settings. On the other hand six Rural-Urban-Region (RUR) types (Very large monocentric (1.0), Large monocentric (1.1), Medium monocentric (1.2), Urban polycentric (2), Dispersed polycentric (3) and Rural (4) (Loibl et al. 2007)) show distinctive elasticities to these different driving forces (determinants) regarding land use change. When referring to the dynamics of change, often a central component of scenario analyses, a third element has to be taken into consideration: the spatially explicit allocation of different land uses. Usually a framework of rules acting via restrictions (e.g. legal) and ethnic behaviours (e.g. migration) leads to growth/decline patterns displaying increased fragmentation or centralisation. Translation of high-scale modelling outputs on changed spatial demands on different land uses to small-scale land use distribution in urban, peri-urban and rural regions requires the definition of spatial allocation procedures/rules based on expert knowledge.

Returning to the question of ecosystem services or functions, it is obvious that apart from the change in space being claimed for particular land use (industry, services, housing, agriculture) the allocation (limited by spatial conditions and connectivity) of such trends and related impacts on land use functions also determine whether a system is receptive towards changing trends of different drivers. By identifying not only trends, but also the land use related impacts supplied or demanded by societal or institutional groups, and any regulatory framework performance towards better targeting, research can support policy makers to direct the competition on the resource of land in a sustainable way.

Innovative approaches to drivers of ecosystem services assessment and change

Tatiana Kluvankova-Oravska, Slovak Academy of Sciences, Slovak Republic

In this contribution the author outlines the need to integrate Social-Ecological Systems (SES) into the RUBICODE approach and offers justification of why it is important to have a dynamic concept of biodiversity conservation within the EU and links it with the existing concept of the Drivers-Pressures-State-Impact-Response (DPSIR) framework.

Many recent studies on ecosystem - social system interaction recognised that interactions exist between people, biodiversity and ecosystems. Changing human lifestyles drive, both directly and indirectly, changes in biodiversity, changes in ecosystems, and ultimately changes in the services ecosystems provide (MA, 2005). However to capture social and ecological dynamics, human dependence on the capacity of ecosystems to generate essential services, and the vast importance of ecological feedback for societal development (Galaz et al., 2007) it is assumed that relationships between social and ecological systems is complex, based on mutual partnership and not domination over each other. To emphasize this concept of humans-in-nature Berkes and Folke (1998) use the term social-ecological system (SES). Our aim is to integrate SES into the RUBICODE approach and offer justification of why it is important to have a dynamic concept of biodiversity conservation within the enlarged EU, and in particular to link it with the existing concept of the Drivers-Pressures-State-Impact-Response (DPSIR) framework presented in the current version by the European Environmental Agency (EEA, 1995).

During the last two decades DPSIR has evolved in to the interdisciplinary tool for environmental analyses, however multidisciplinary and linear origins remain. Recent findings emphasises such limitations. Svarstad et al. (2007) underlines the static relation between drivers, pressures and state in respect to biodiversity. Drivers as the underlying causes of environmental change are often understood as external negative forces to ecosystems and species, lacking complexity of interdependent ecological- social processes linked via dynamic interactions and outcomes.

The main objective of our approach is to concentrate on social–ecological dynamics and to understand ecology and society interdependence. The central component of the of the DPSIR framework is that it is formed of ecological and social systems (SES) framed by the dynamic concept of interdependent systems, where dynamics is understood as the state of multiple equilibrium –(socio-ecological resilience) and replaces state (S) from the DPSIR framework. The basic components of ecological systems are communities of organisms interdependent of each other and their environment. Social systems include economy such fishing or agriculture, actors or resource users and institutions such governance structure.

Drivers are key factors of change often with cascading effects across various scales, which result in the changes (impact) to ecosystems and social systems. Pressures are variables of such processes interacting within ecosystems and social systems. Similarly state is understood by the concept of dynamic stability, known as socio-ecological resilience (Folke, 2006) or the adaptive capacity of an ecosystem to remain within multiple-equilibrium or its ability to reorganise into a possible adaptation. Responses are adaptations to socio-ecological dynamics such as adaptive governance.

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Session III: Valuation of ecosystem services

Introduction to Session III: Valuation of ecosystem goods and services

Michalis Skourtos, Department of Environmental Studies, University of the Aegean, Greece

Valuation of environmental assets has been practiced actively since the 70s. Although a pure academic curiosity at its beginnings, this particular line of research has gained a momentum since the popularisation of the concept of ecosystem services by the Millennium Ecosystem Assessment (MA). It is today considered a cornerstone of sustainability assessment procedures both within national and international agencies as well as by private business and conservation practitioners. At present, research on ecosystem service valuation is rooted mostly on economic premises and methods. Alternative approaches based either on multicriteria approaches and deliberative/inclusionary approaches are also being developed. Ecosystem valuation is a highly controversial issue not only because of its technicalities but foremost because of the underpinning philosophical and moral strands.

The main objective of the session on valuation of ecosystem goods and services is to contribute to an understanding of the state-of-art in the design, empirical application and policy relevance of valuation exercises. The following topics give an indication of research questions that need further exploration:

1. How do valuation studies define and communicate changes in ecosystem service provision? To help society make informed decisions in using space and resources we have to start by quantifying ecosystem functions and identifying needs. Valuation studies of ecosystem goods and services use a palette of conceptual and visual devices to define and communicate changes in ecosystem goods and services. The challenge is portrayed by what J. Boyd (2007) has termed "endpoint problem': Boyd defines ecological endpoints as concrete statements, intuitively expressed and commonly understood, about what matters in nature. The road most usually taken is to focus on nature as an asset and thus produce static value estimates. The recent popularisation of the concept of ecosystem services (MA) and its refinement as the Service Providing Unit concept (senso Luck et al., 2003) promises a more coherent, dynamic treatment of ecosystem benefits through time. What can scientific knowledge offer for the resolution of such conflicts, especially at local scales and within ecosystem entities that mediate multiple functions? And what can the social sciences contribute?

2. How well have monetary valuation methods addressed the needs of conservation strategies to date? Do people care for the environment? The published evidence for monetized expression of human preferences for conserving ecosystems, as it has been documented up to now for individual biomes and the totality of earth's ecosystems, suggests the answer must be 'yes.' D. Pearce (2007) has questioned the validity of this assertion by comparing estimates of global ecosystem values with global expenditure on ecosystem protection. For people in the conservation arena this is a truly alarming assertion since it documents a 'global deficit of care' to resolve global warming and conservation problems. For others it may simply indicate the shortcomings of value estimates and related economic methodologies. In the first case, valuing ecosystem services turns out to be irrelevant for policy purposes irrespective of being methodologically solid whilst in the second case it turns out to be methodologically flawed irrespective of being policy relevant. A possible explanation though might be linked to the dual nature of the valuation process: estimation and capture. The former answers the question of the magnitude of social benefits but the latter is actually concerned with translating the value estimates into motivation for behavioural changes. How successful has the environmental decision-making process been to date in capturing ecosystem value estimates?

3. The existence (and measurement) of existence values. Focussing upon pure existence value, is it credible that these could dominate the value of some non-unique resource? The definition of non-use value embraces more than just pure existence value including also bequest value. The big challenge to any claimed existence value is that such values are typically not particular to the goods under evaluation but reflect the 'warm glow of moral satisfaction' associated with supporting any good cause. Does this matter? And how does any value estimate prove it is not a mere warm glow?

4. The spatial context of ecosystem service valuation. In the context of environmental valuation, a basic problem is that we often do not know the spatial incidence of many (environmental) public goods including ecosystem goods and services. Even if we did, these may rarely be congruent with the jurisdictional boundaries that can mandate conservation or supply through public policy. The appraisal underlying such policies (including implicit or explicit valuation) is therefore always likely to be partial in the extent to which it considers any system boundary that might be meaningful in biological or ecological terms. This is not always the case, but this spatial/governance mismatch characterises many important local and global environmental problems. If the world were one country then this would not matter. But this is not the case, and we have to be specific about how scale fits into appraisal and valuation; whether and how we can draw system boundaries around the relevant units of provision of goods and services, or whether partial valuation is the best we can do.

5. The complementarity of monetary and non-monetary approaches to ecosystem service valuation. In the context of "social choice" (which is an economics problem) we are obliged to mobilise information of all sorts - qualitative, quantitative multi-metric and monetized - in structured ways. Economics is not uniquely concerned with monetary or market-line considerations. Many people though consider the treatment of preferences in economics as too formalistic, devoid of any substantial content. They propose instead the use of deliberative techniques: Deliberative decision-making is the term used to cover a wide variety of techniques that have some element of detailed consideration. It could hence be useful to supplement monetary estimations with more qualitative, in-depth insights from inclusionary techniques. What would such a mixing of methods look like in the realm of ecosystem valuation?

Re: Introduction to Session III

George Cojocaru, TIAMASG Foundation, Bucharest, Romania

Valuing ecosystem services is obviously a difficult task, mainly because of the diversity of the services and the diversity of the places where the services are applied, which are sometimes antagonistic. I think that we have to split the services according to three main pillars: economic services, environmental services and social services (maybe the aesthetic services are also important). Within each pillar we should define metrics (hierarchical or not-hierarchical) which can help us to estimate the importance of each service considered. And, finally, we need to aggregate the three/four pillars according to the importance we consider that each pillar should have.

Although for the economic services the monetary valuation method is clearly the best one, I do not think that this method should be applied to the social and environmental services. An exercise for finding a good method in these cases could start from the lists of the most important social and environmental ecosystem services we know. Looking at these lists we can then use a mathematical or empirical method to estimate their values.

What's new in the ecosystem service approach to valuation? The challenge of SPUs

Areti Kontogianni, University of Aegean, Greece; and **Gary Luck**, Institute for Land, Water and Society, Charles Sturt University, Australia

This contribution outlines the need for standardised methods to measure ecosystem services in order to be able to place a valuation on them. It highlights the importance of the development of new terminology, such as SPUs and ESPs in allowing a better understanding of complex ecosystem processes, thus aiding quantification and ultimately valuation of these services.

To help society make informed decisions about using space and resources we need to quantify ecosystem functions and identify ecosystem needs. Valuation studies of ecosystem goods and services use a range of conceptual and visual techniques to define and communicate changes in ecosystem goods and services. The challenge is portrayed by what J. Boyd (2007) termed the "endpoint problem': Boyd defines ecological endpoints as concrete statements, intuitively expressed and commonly understood, about what matters in nature. The view often taken is to focus on nature as an asset and thus produce static value estimates. However, the recent popularization of the concept of ecosystem services (MEA) and its refinement as a Service Providing Unit (senso Luck et al., 2003) promises a more coherent, dynamic treatment of ecosystem benefits through time.

As the endpoint problem suggests, both biophysical and social sciences are today in desperate need of a method to measure ecosystem services. Quantifying ecosystem services is a prerequisite for valuation and it is unfortunate that until now this has not been provided by ecology. Carpenter et al. (1995) suggests that the time hysteresis of quantifying ecosystem services is due to the fact that ecosystem level experiments are difficult, costly and timely. Meanwhile, environmental economists have carried out their valuation applications hoping to gleam the best out of the ecological research available and to input it into their work. Economists have been mostly concerned with testing methodologies for potential biases and the applicability of their own methodologies of non-market valuation. What they were actually measuring in many cases was a combination of talent and intuition due to insufficient ecological information.

Recent advances in ecological science of defining terms like service providing units (SPU) (Luck et al., 2003) and ecosystem service providers (ESP) (Kremen, 2005) allow for better understanding of complex ecosystem processes, quantification of ecosystem services and thus better integration of ecological concepts into social sciences generally. If ecologists are successful through the SPU in "decoding" the functions of nature and consequently in transforming disparate factors of production into a collection of attributes which, taken together, are valuable and so command a certain price (which is the very nature of production processes in an economy) (Vatn and Bromley, 1995), then the basic condition for successful valuation will be met.

A number of questions need to be considered:

- Will SPU enhance interdisciplinary collaboration and understanding between ecologists and economists and thus promote more validated, well-informed valuation applications?
- Does SPU apply equally well to both use and non-use values?
- Could SPU enhance understanding of the objects being valued and accordingly bypass the cognitive problems giving rise to heuristic devises used by respondents in stated preference surveys?
- Is SPU more suited to address dynamic issues of value estimation?

The existence (and measurement) of existence values reconsidered

Ian Bateman, University of East Anglia, UK

In this contribution, the author highlights the difficulties associated with measuring existence values, in particular the application of various frameworks for economic analysis of existence values, such as contingent valuation and choice experiment methods, are discussed.

Not naming names but, a few years ago an official government agency of one EU country undertook a cost-benefit assessment of a scheme to prevent eutrophication at a major lake. This assessment claimed that 95% of the total value of that scheme would be non-use value, a result which, for me at least, fails the 'laugh test' (Fisher, 1998). Of course it could be that the definition of non-use used here embraces more than just pure existence value (that associated with non-human species) to include bequest value (essentially an altruistic concern for the benefits of others including future generations). There are a number of practical problems with the assessment of bequest not least because any value based upon altruism is liable to double counting (Jones-Lee, 1992). More fundamentally, it is clearly linked to eventual use and so is erroneously labelled as non-use (Weikard, 2002).

So, focussing upon pure existence value, is it credible that these could dominate the value of some non-unique resource? A recent test noted that making a resource open to public access roughly tripled its value (Luisetti, et al., 2008); suggesting in turn that it is use rather than existence values which dominate.

The big challenge to any claimed existence value is still that proposed by Kahneman and Knetsch (1992). This argues that such values are typically not specific to the goods under evaluation but reflect the 'warm glow of moral satisfaction' associated with supporting any good cause. So, the value of the rare turtle, the donkey sanctuary and the dog's home all become interchangeable. Does this matter? From a purely financial perspective the answer may well be 'no' and we return to this later. However, for economic analyses the answer is 'yes' – because otherwise there is a serious risk of double counting all those warm glows. But how does any value estimate prove it is not a mere warm glow?

The criterion for admission into economic analysis is that a value estimate conforms to the characteristics laid down by economic theory. Unfortunately there actually are not many expectations to satisfy here or act as criteria. One that was highlighted by the NOAA blue ribbon panel (Arrow et al., 1993) on contingent valuation (CV) was the 'scope sensitivity' test. This requires that values, typically expressed as willingness to pay (WTP), should not decline as the 'size' (or quality) of the good under evaluation increases. This is not a particularly stiff test to pass. For example, ranges over which total WTP remains constant are perfectly compatible with both this test and common sense (simple satiation would yield such a result). More crucially the reasonableness of the degree of any sensitivity is not typically known a-priori, at least not for existence values. This has led to the mere observation of statistically significant scope sensitivity being interpreted as proving the validity of such estimates.

While scope tests for non-market goods are a necessary condition for validity, they remain insufficient in the absence of prior expectations regarding the appropriate degree of sensitivity. Furthermore, even these analyses are often compromised by relying on internal (within-person) tests. Clearly this faces a problem of respondents forcing consistency upon themselves by anchoring upon initial responses and adjusting these in the direction of the scope change presented to them. Such approaches were generally rejected for CV analyses in favour of external (across-respondent) tests where different individuals are presented with different scope goods and tests conducted across these responses. However the recent rise of choice experiment (CE) methods has reversed this trend. CE studies offer great potential in terms of flexibly identifying the relationship between the scope of a good and its consequent values. However this is achieved by asking respondents repeated questions involving different scope-money trade-offs. Pooling these responses together does not change the fact that these are within-respondent analyses and therefore inherently liable to self-enforced consistency. In short the observation of scope sensitivity within such studies signifies nothing more than a basic level of intelligence within respondents. Further, expectations based tests are required.

One possibility which should be compatible with both the CV and CE framework is to examine not only scope sensitivity, but also certain other relationships which the theory predicts. One which may be worthy of further consideration is that concerning substitutes and complements. One would expect that the value of a given level of a specified good would reduce in the presence of near substitutes and be relatively unaffected (income effects aside) by the presence of unrelated goods. Similarly values for one good should be heightened by the presence of complements. A raft of expectations based tests of use and non-use values are being conducted as part of the ongoing ChREAM project funded by the UK RELU programme (for further details see www.uea.ac.uk/env/cserge/research/relu/index).

Although such expectations based additions to scope sensitivity are, I feel, worthy of investigation, analysts and policymakers should not get their hopes too high. There are use-value analyses for cases where there are fairly clear prior expectations derived from basic economic principles. Perhaps the clearest of these is within the valuation of health risk reductions. Here, given that the avoidance of fatality can fairly reasonably be interpreted as the avoidance of losing virtually all utility (concerns about future generations aside), then the only constraints upon WTP should be those imposed by income. For example, the utility of gaining an extra two months of life expectancy should be roughly double that of gaining one month's life expectancy. Unfortunately, high quality, split sample (cross-respondent) tests of this expectation are not encouraging; the degree of scope sensitivity is less than expected (Chilton, 2004).

There may be some light at the end of the tunnel. Recent research suggested that bringing the experience and familiarity of market repetition can yield values which satisfy expectations based tests in some circumstances (Bateman et al., 2008a). Furthermore, from the perspective of conservation organisations it is arguable that the niceties of defensible economic analyses might not always be necessary for the aims of such groups to be met. Even if the WTP for existence values are not specific to any particular good, providing the individual will actually pay up for their warm glow then it is simply the conservation groups' goal to ensure that it is their donkey sanctuary/dog's home/etc., which secures that payment. In a recent study we show that by easing the mode of payment from individuals and ensuring that this provides incentives to the producers of conservation goods, values which are likely to fail tests of economic validity can nonetheless be turned into real funds for ensuring the conservation of endangered species (Bateman et al., 2008b). From the perspective of that species this is perhaps a more important and real-world test of validity!

The spatial context of ecosystem service valuation

Dominic Moran, Scottish Agricultural College, Edinburgh, UK

Difficulties with calculating, in a spatial context, how much of a target ecosystem should be conserved and the consequences that may then follow, are discussed. In particular the importance of understanding the scale required for relevant ecosystem processes and functions to occur in conserved areas is considered.

Spatial scale is not something that economics deals with particularly well. In the context of environmental valuation, a basic problem is that we often do not know the spatial incidence of many (environmental) public goods including ecosystem goods and services. Even if we did, these may rarely be congruent with the jurisdictional boundaries that can mandate conservation or supply through public policy. The appraisal underlying such policies (including implicit or explicit valuation) is therefore always likely to be partial in the extent to which it considers any system boundary that might be meaningful in biological or ecological terms. This is not always the case, but this spatial/governance mismatch characterises many important local and global environmental problems. If the world were one country then this would not matter. But this is not the case, and we have to be specific about how scale fits into appraisal and valuation; whether and how we can draw system boundaries around the relevant units of provision of goods and services, or whether partial valuation is the best we can do.

This wish list is obviously one that economists therefore need to bounce back to the biologists and ecologists. When it comes to local and global ecosystem services, we would like to be specific about the spatial extent of what it is we need to conserve. This then means that within any target ecosystem we would ideally like to be specific about the scale of the relevant processes and functions that give rise to the services that are socially relevant. There are two consequences of getting this wrong. If we conserve too much, then there is an opportunity cost and resources that could be directed to conservation elsewhere are wasted. Too little, and we risk cutting off conservation at critical thresholds and getting discontinuities. Either way, society loses out. It is demonstrably the case that the ecological consequences of poor designation have not been made as clearly as the economic case for truncating space given over to conservation. While there is much talk of thresholds, system discontinuities, and irreversibility, there is much less work that pinpoints where these are, and which ecosystem functions really need to have an alarm bell ringing for them. In contrast, the opportunity cost of conservation is only too apparent.

The need for clarity on providing units has real-world applications; for example the design of protected areas on land or at sea, the formulation of agri-environment payment regimes, and systems of so-called payments for environmental services. In all cases we need to provide policy with convincing scientific evidence based on the “providing unit”, or more crudely, the relationship that links processes to functions to economic endpoints. This challenges science to untangle the complexity in natural systems and to demonstrate what is vital and where we need to be taking precautionary arguments to justify conservation. We cannot save everything and we need to avoid a tendency to fall back on intrinsic arguments or emotive calls that imply that we can. This is my pragmatic interpretation of what the Ecosystem Approach is meant to be about. It is also the only useful way I can interpret the purpose of the Millennium Ecosystem Assessment.

Measuring benefits of ecosystem service provision in monetary terms

Kerry Turner, University of East Anglia, UK

In this contribution the author defines ecosystem services as the aspects of ecosystems utilized to produce human well-being, and argues that for valuation purposes a classification scheme that divides ecosystem services into intermediate services, final services and benefits would be useful.

The concept of ecosystems services has become an important model for linking the functioning of ecosystems to human welfare benefits. Understanding this link is critical in decision-making contexts. While there have been several attempts to come up with a classification scheme for ecosystem services, there has not been an agreed upon, meaningful and consistent definition for ecosystem services. Any attempt at classifying ecosystem services should be based on both the characteristics of interest and a decision-context. Because of this there is not one classification scheme that will be adequate for the many contexts in which ecosystem service research may be utilized.

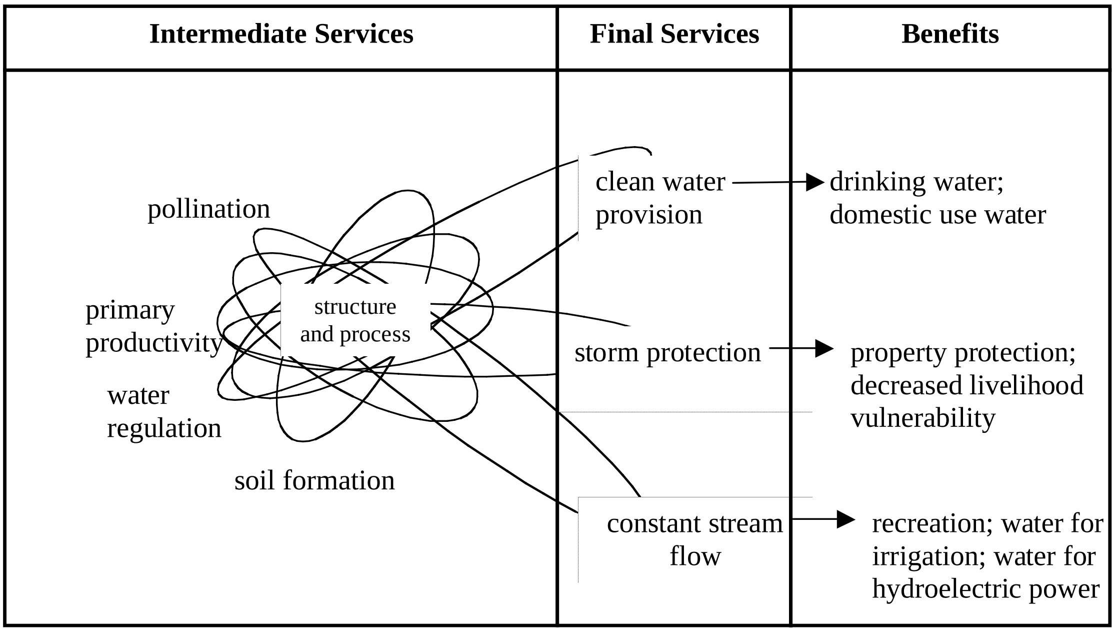
Drawing largely on Boyd and Banzhaf (2007) we propose that ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being. The key points include that services must be ecological phenomena and that they do not have to be directly utilized. Defined this way, ecosystem services include ecosystem organization or structure as well as process and/or functions if they are consumed or utilized by humanity either directly or indirectly. (Boyd and Bazhaf see services as only the directly consumable end point). The functions or processes become services if there are humans that benefit from them. Without human beneficiaries they are not services.

Ecosystem structure (called component in Boyd and Banzhaf 2007) is a service to the extent that it provides the platform from which ecosystem processes occur. How much structure and process is required to provide a diversity of ecosystem services in a given ecosystem context is still an active research question. Clearly some minimum configuration of structure and process is required for ‘healthy’ functioning and service provision. This ‘infrastructure’ has value in the sense that its prior existence and maintenance is necessary for service provision, and is therefore a service in itself (Turner 1999).

If the goal or decision context is to value ecosystem services then the MA classification is not appropriate and some other scheme should be utilized. This is due to the fact that the MA classification could lead to double counting the value of some ecosystem services. For example, in the MA, nutrient cycling is a supporting service, water flow regulation is a regulating service, and recreation is a cultural service. However, if you were a decision maker contemplating the conversion of a wetland and utilized a cost-benefit analysis including these three services, you would commit the error of double counting. This is because nutrient cycling and water regulation (both means) help to provide the same service, providing usable water, and the MA’s recreation service is actually a human benefit of that water provision.

For valuation purposes a classification scheme that divides ecosystem services into intermediate services, final services, and benefits would be more appropriate (Figure 4).

Figure 4. A classification scheme that divides ecosystem services into intermediate services, final services, and benefits



With the definition above, all ecosystem processes and structure are ecosystem services, but they can be considered as intermediate or as final services, depending on their degree of connection to human welfare. The same service can also be both intermediate and final depending on the benefit of interest. This classification scheme recognizes that ecosystems are complex, and rather than understanding all of the complexity we just have to be clear about some final services and benefits with which we are concerned. In doing so it also appreciates the benefit dependence characteristic. This classification avoids a double counting problem because you would only value the final benefits (i.e. the things that directly relate to changes in human welfare), and hence is fit for purpose in a valuation context.

This contribution is based on B. Fisher, R.K. Turner et al. Defining and Classifying Ecosystem Services for Decision Making in press at Ecological Economics available as a CSERGE working paper at http://www.uea.ac.uk/env/cserge

Managing ecosystems and the services they provide

Rob Tinch, Environmental Futures Ltd., UK

This contribution aims to address aspects of one point in Allan Watt's opening list in the introduction to session four: "What research is needed to develop indicators that assist decision-makers in managing ecosystems and the services they provide?", drawing on points made by various other contributors to the e-conference.

Martin Sharman stated "We must identify concepts and associated behaviours that allow us to make our relationship with nature (I will not say "ecosystems") sustainable, and above all to forestall catastrophic shifts in those ecological regimes that maintain us."

Quite right - the fundamental question is how to ensure sustainability. This question arose because the scale of human numbers and activity grew enough to seriously impact on the natural world, the resources and support systems we rely on. These impacts have carried on growing, and the urgency of the question has grown in parallel.

This has brought the natural world and our relationship with it within the subject area of economics. As Michalis Skourtos pointed out, "[e]conomics is not uniquely concerned with monetary or market-line considerations", but rather with questions of resource allocation under scarcity. How do societies (or governments, organisations, firms, households, individuals) use the scarce means at their disposal to achieve, as best they can, an essentially unlimited wish-list of objectives. Now, increasingly, the key scarcities are environmental. The relationship of human activity with the natural world is now a fundamental aspect of social choice, and we are working towards frameworks and classifications for assessing the interactions and incorporating them within decision making. Economics / choice theory brings a large toolkit, including but not limited to market-based methods, but no one method is universally applicable.

For example, monetary pricing within a market structure has proven to be extremely useful in many settings. Markets are highly efficient mechanisms (institutions) for allocating resources among competing ends. Prices will tend to rise as supply is restricted, so there is an element of self-regulation in the face of increasing scarcity; however there is nothing that guarantees any sort of sustainability constraint. Also, equity/fairness issues are essentially determined by initial allocations (property rights), not directly through the operation of the market.

This might suggest that markets for environmental services, and prices/values associated with them, could be useful guides for allocation, provided that sustainability constraints, and equity concerns, are addressed in other ways. In other words, economic valuation might be useless for answering the fundamental question of international sustainability at the top-down level (and the total economic value of the global environment may be a meaningless concept), but values based on a framework solution to the sustainability problem might be extremely useful for helping local or regional decision makers (in some cultures/places/times) to ensure their decisions are consistent with global sustainability, or to create structures within which individuals behave in ways which are sustainable in the aggregate.

Martin Sharman argued that the ecosystem concept "is not just a projection onto reality - and perhaps an (imperfect) reflection of it - but it actively constructs our perceptions of it". This holds even more for the concept of ecosystem services, which are fundamentally subjective (arguably, the whole point is to construct / influence human perceptions of nature and the way we benefit from and use it); and even more still for any attempts at an ecosystem service evaluation framework (monetary or not).

I don't think that this weakens the usefulness of the concepts. Martin suggested that "By using the adjective [dynamic] we betray our unconscious attitude towards an "ecosystem" - an attitude that is saturated with ideas of stasis, equilibrium, and resilience." In contrast, I'd argue that the use of "dynamic" (or of "complex", "non-linear", "far from equilibrium" etc) is a conscious stressing of the limitations of the static view (and that it can be viewed as an oxymoron suggests that the concept "ecosystem" is not in fact saturated with ideas of stasis). Similarly, our debates over ecosystem service taxonomies and valuation frameworks reflect conscious efforts to develop appropriate techniques for framing and assessing the natural world and our interactions with it in ways that can be useful in different decision and communication settings. Martin is right to view this as active construction of our concepts of the natural world and our uses of it, and right to argue that there is no telos in the natural world ("ecosystems have no purpose"), but I think wrong to imply that this undermines the ecosystem services framework and is "cynical and unethical". The absence of purpose in nature does not mean nature can not serve a purpose for us - these are fundamentally different concepts.

The research needs here are partly technical (in economics and in ecology) but to a greater extent involve moral issues. Kerry Turner's contribution is important in this context, with his message that "there is not one classification scheme that will be adequate for the many contexts in which ecosystem service research may be utilized." We need further efforts to recognise and research: (1) the plurality of decision and communication contexts within societies; (2) the relative merits of different classification frameworks, evaluation methods and decision support tools for these contexts; and (3) the scale dependency and cultural dependency of answers to (1) and (2).

Session IV: Research priorities for ecosystem service assessment

Introduction to Session IV: Research priorities for ecosystem service assessment

Allan Watt, Centre for Ecology and Hydrology, Edinburgh, UK; and **Christian Anton**, **Martin Musche** and **Josef Settele**, UFZ - Helmholtz-Centre for Environmental Research Leipzig, Germany

The goal of RUBICODE is to provide support to decision-makers in developing conservation policies to halt biodiversity loss. It considers two areas of concern in particular, firstly, the translation of threats to biodiversity into tangible and quantifiable factors for use by policy-makers in their decision-making processes. RUBICODE does this by developing innovative methods for quantifying ecosystem services so that the full consequences of biodiversity loss can be better assessed. Secondly, RUBICODE recognises that ecosystems are dynamic and that decision-makers need to take this into account, particularly in an era of global change. Management of dynamic ecosystems means maintaining their capacity to undergo disturbance. RUBICODE links these two basic concepts by focussing on the ecological resilience of biological units that provide specific services.

The aims of RUBICODE are more fully discussed by Paula Harrison in her introduction to the e-conference and are being discussed by the e-conference participants in parallel sessions. The objective of this session is to start a discussion within the wider scientific community on research priorities. We need to consider questions such as:

- What research is needed to develop more effective interfaces between researchers, policy-makers and other stakeholders?
- What are the priorities in developing concepts of dynamic ecosystems, functional diversity, ecosystem services and values?
- Where should we concentrate our efforts in identifying and quantifying socio-economic and environmental drivers that affect ecosystems and their services?
- What research is needed to develop indicators that assist decision-makers in managing ecosystems and the services they provide?
- What are the priorities for research on biological traits, particularly in relation to the ecological functions underlying ecosystem service provision and the conservation of biodiversity?
- What research is needed to develop improved habitat management strategies, with particular emphasis on the need for strategies that acknowledge the dynamic nature of ecosystems and their role in providing multiple services?
- How can research inform priorities for habitat, ecosystem and landscape biodiversity conservation policy, particularly in the context of the delivery of ecosystem services?

In the coming months we will develop a roadmap for research on the conservation of dynamic ecosystems and the services they provide. Please use this opportunity to include your opinions in this exercise. Send your views, however short, ideally focussing on one research priority with a short justification.

The term 'ecosystem': misleading and deceitful?

Martin Sharman, European Commission, Belgium

In this contribution, the author outlines the difficulties posed as a result of badly defined terminology and highlights the need to identify concepts and associated behaviours that allow for a sustainable relationship with nature.

The initial priority in developing concepts of dynamic ecosystems, ecosystem services and values is to be sure that we all agree on the concepts behind the terms we use. We all agree - I think - that an ecosystem is an assemblage of organisms interacting with their environment, the major determinants of the assemblage being the physical characteristics that limit the distribution of the component species, while the most significant elements of the interactions involve other organisms. Whatever the bounds of the system, it is typically characterised by a complex web of interactions governed partly by stochastic events from beyond its boundaries, and feedback between the elements internal to the assemblage itself.

I argue that the word "ecosystem" deceives and misleads. The construction and acceptance of this word reflects a very human desire to comprehend, classify and predict; it overlays on nature a perception of how things should be rather than how they are. It suggests stasis and objective reality, boundaries and integration. On the scale of human lifetimes, ecosystems can often - quite legitimately - be regarded as existing in a steady state. This perception has lead to the adoption of ideas like the "balance of nature" and the "health" of an ecosystem, and to faith in regulating mechanisms that have somehow evolved to keep the ecosystem in that healthy, balanced, stable state. Words like "climax community", "stability", "sustainable", "function", "disturbance", and "resilience" all concord with this comforting view of nature.

Can we see the concept of "ecosystem" as a social construct invented and institutionalized by ecologists in a particular scientific culture? It is not difficult to believe that an indigenous Australian or African hunter-gatherer would not recognise the Western "ecosystem". Their world views incorporate a spiritual world quite foreign to us, but which permeates and informs all aspects of their life. Spirits determine fecundity, health, dearth, plenty, sickness and death. By influencing the spirits, they hope to turn aside misfortune. Their spirits do not include carbon cycling or ecosystem boundaries. Their world view could conceivably include concepts analogous to these, but the codes they use to signify these ideas emerge from and sustain their cultures, not ours. In our own scientific culture, too, meaning is dependent on ideological and political positions.

Why does this matter? Because the word "ecosystem" is not neutral or value-free, it expresses and maintains a power relationship with non-human worlds. It is not just a projection onto reality - and perhaps an (imperfect) reflection of it - but it actively constructs our perceptions of it. This is easily demonstrated: the phrase "dynamic ecosystem" is an oxymoron - all systems are by definition dynamic. By using the adjective we betray our unconscious attitude towards an "ecosystem" - an attitude that is saturated with ideas of stasis, equilibrium, and resilience.

If we agree to behave as if ecosystems exist and follow rules, then they will necessarily obey. After all, a system is a set of entities that through their relationships and interactions form an integrated whole, so this is what an ecosystem must be and do. It says so in their name. We control the discourse; and by doing so we imagine we control the reality, but in fact we control only our understanding of the reality.

We are now 21st century scientists and policy makers, and we move with the times. We understand that much of the feedback between the elements of an ecosystem is non-linear, so that a given amount of change in one element will not always produce the same response in other elements. In some cases the feedback is also positive, so that the response to a change tends to amplify the change, but the response may only occur after a delay whose length depends on the initial state of the system. We know that these characteristics - complex systems with state-dependent delay to nonlinear positive feedback - warn us that we are dealing with the potential for catastrophic and cascading ecosystem change. We call this, somewhat smugly, “regime shift”, a term that in my mind at least finds a disquieting echo in the calmly predicted “regime change” that has led to six chaotic and bloody years in Iraq. Words have power. We should be careful not to believe them without reflection.

Artificial systems are usually designed to work coherently to achieve some purpose. By contrast, most ecologists would agree that ecosystems have no purpose - yet we are all keen to jump on the “services” bandwagon. “Ecosystem service” reifies the ecosystem and gives it value. Yet at the same time we overtly acknowledge, at least between ourselves, that ecosystem services are a device to bring to policy-makers’ attention some of the ways in which nature contributes to human welfare.

As a direct and bitter consequence of this cynical and unethical approach, valuation is increasingly driven by short-term economic interests. If we could step into the future and look back at this suicidal time in the history of ecology, we might agree with Shakespeare that this approach “lighted fools the way to dusty death”. The urgent priority must be to refocus our intellectual efforts to prevent us from slithering our way down this restricted, dogmatic plughole.

We must identify concepts and associated behaviours that allow us to make our relationship with nature (I will not say “ecosystems”) sustainable, and above all to forestall catastrophic shifts in those ecological regimes that maintain us.

Experiments and models: scale and complexity

Sandra Díaz, CONICET-UNC and FCEFyN, Argentina

We need experiments on the effects of different components of biodiversity on ecosystem processes at the scale in which management is done in practice. Most experiments so far have been carried out at too small scales, and with a very limited set of organisms (those easy to manipulate in a controlled environment). While controlled experiments will continue to be relevant to answer some specific mechanistic questions, there is a need to move towards experimental settings that take advantage of 'natural experiments' and also incorporate real land-use situations in which a carefully controlled, factorial design is not suitable. This will pose statistical challenges, but will get us closer to the real scale at which ecosystems are managed, and at which ecosystem services are delivered.

In the same way, we need models that address the complexity of managed ecosystems and the services they deliver, and the way in which they are connected with each other. Specifically, we need to model in a more satisfactory way: (a) functional diversity responses to different factors, (b) biological feedback onto ecosystem services, across spatial scales, (c) thresholds beyond which the level of ecosystem service delivery changes dramatically and perhaps irreversibly; (d) externalities (positive and negative) across the landscape.

Pattern-process levels and scale dependent mechanisms

Sandra Luque, Cemagref - Institute for Agricultural and Environmental Engineering Research, France

Some of the most obvious landscape related omissions from the MA scenarios are pattern-process feedbacks, scale dependencies, and the role of landscape configuration. I would like to point out two main subject areas for future research that are interrelated. The first is the issue of scale dependency and the understanding of mechanisms at different levels. The second is related to the resistance and resilience of ecosystems in relation to feedback mechanisms in particular. In both cases, understanding the interrelation between ecosystems and landscapes level mechanism is critical.

While the MA has set a new standard for biodiversity scenarios, future exercises would benefit from a more multi-scale and mechanistic framework. Integrative landscape-driven research should be envisioned to relate ecosystem processes and global changes including climate changes and socio-economic processes across different governance levels.

Predicting biodiversity change involves understanding not only ecology and evolution, but also complex changes in human societies and economies. One of the most important challenges for future research will be to integrate research across different scales, including spatio-temporal scales within an interdisciplinary and multidisciplinary framework. As Dominic Moran pointed out last week, spatial scale is not something that economics deals with particularly well. In my experience working at the habitat to landscape level with economists, it was actually the spatial representation and modelling-based approach that allowed a common understanding to achieve applied results for sustainable use of forest resources. As landscape ecologists we can actually provide the spatial context to bridge differences between economists and ecologists.

The capacity of ecosystems to deliver services needed by society depends on resistance and resilience of associated habitat types. The resistance and resilience of ecosystems is dependent on the complexity of the ecosystem functioning (especially feedback mechanisms), production capacity and frequency of the disturbance and stress factors. Our limited understanding of ecosystem dynamics is reflected most forcefully in our inability to predict where thresholds separating two alternative, qualitatively different ecosystem states (well functioning and degraded) lie. The relationship between the condition of ecosystems and the services they provide is not a simple linear one, but rather more complex. Ecosystem services may show considerable resilience to stress, but also exhibit rapid and indeed catastrophic, change upon the transgression of certain, yet poorly identified thresholds. Future research should aim to establish methods to determine the sustainability of various intensities of use of components of biodiversity and of ecosystems. We should be able to evaluate different habitat types in terms of disturbance frequency and intensity that can be imitated in the management and use of such ecosystems. A novel integrative approach considering different scale levels should help to detect when ecosystems are approaching the limits of their natural functioning or productive capacity.

In summary, in order to integrate different approaches, future research should consider different scales into a more integrative landscape level approach. Pattern-process levels and scale dependent mechanisms need to be studied at different spatio-temporal scales. In the same way, different levels, in terms of governance and conflict resolution also need to be considered.

Assessing the links between functional diversity and ecosystem services

Francesco de Bello, Cemagref - Institute for Agricultural and Environmental Engineering Research, France

1. Biodiversity indication approaches based on functional traits can potentially be estimated from rapid field assessments and/or remote sensing tools. Several RUBICODE reviews on traits highlight the potential for integrating the functional traits approach into predictive models of ecosystem services assessments.
2. The type, range, and relative abundance of functional traits in biotic communities, collectively referred to as ‘functional diversity’, strongly influence different ecosystem services in a range of organisms and habitats. Future research priorities should focus on producing shortlists of key traits for different organisms and defining standardized protocols to assess these lists.

The need for biodiversity indicators

Christian Feld, University of Duisburg-Essen, Germany

Halting the loss of biodiversity by 2010 is a principle aim of European as well as global policies. However, looking at the set of indicators proposed to monitor sub-global and global biodiversity, we quickly realise that very few indicators are proposed i) that are biological and ii) that account for genetic, structural and functional components of biodiversity.

What is biodiversity? The question seems anachronistic, since it has been exhaustively addressed by numerous papers and standard textbooks since the 1970s. We should know the answer. Therefore, let's change the wording: What biodiversity do we need for Earth's well-being? And what do we need for human well-being (not necessarily the same!).

The latter is subject to the Millennium Ecosystem Assessment (MA) and other related initiatives to monitor the Earth's services and their benefit to humans and human well-being. So, again the question, slightly changed: What and how much biodiversity do we need to maintain the services and the underlying functions and processes? Well, we don't know! But we should, since this is crucial to the goals of the MA, the Convention on Biological Diversity (CBD) and other policies—and finally to our well being.

What we know, and what RUBICODE has nicely summarised through a number of review papers, is that the ecosystem functions are key to many ecosystem services and that these functions can be approached by ecological traits of the ecosystem actors, i.e. plant and animal characteristics that reflect its linkage to the biotic and abiotic environment.

What we also know is that the vast majority of large-scale biodiversity indicators published in peer-reviewed literature refers to abiotic measures of biodiversity: habitat area and fragmentation, for instance. Those measures are easy to gather from remote sensing data, and that is likely to be the reason why they are broadly applied. But: that's not biodiversity, the measures act as proxies for biodiversity, as surrogate measures of biodiversity. They are surrogates for species richness, which is the most widely applied biological measure of biodiversity.

Yet, coming back to the questions above, what we should apply is indicators for functional biodiversity, for structural biodiversity, for ecosystem processes that make regulatory and supporting services. We should address genetic biodiversity as well. And we also may keep the species and community diversity indices that dominate the textbooks.

Thus: What is biodiversity? It is the combination of different aspects: structural diversity, functional diversity and genetic diversity. What biodiversity do we need to sustain Earth's and human well-being? We don't know, but we know that functions and processes are the key to the answer. We also know that we can address the latter with ecological traits.

Consequently, future research priorities should address the development of biological (!) biodiversity indicators that i) account for the different aspects of diversity, ii) address the ecosystem functions and processes - and ultimately - ecosystem services, and are applicable at regional, sub-global and global scales.

Positively linking biodiversity with ecosystem goods and services

Martin Zobel and **Mari Moora**, Institute of Ecology & Earth Sciences, Estonia

Recent experiments with artificial systems have shown that biodiversity is positively associated with an ecosystems' capability to provide goods and services. At the same time, experiments have also shown that in many cases, not the species number per se, but rather functional diversity or presence of a particular species (including so-called keystone species or ecosystem engineers) is important.

The results of these experiments, however, do not tell us much about what is going on in nature. We lack almost any information on how the structure and diversity of natural communities is related to their capabilities to provide services. With respect to supporting services, which provide a basis for other types of services, we have some (but certainly not sufficient) information about the relationship between biodiversity and primary productivity. We have extremely scarce information about how biodiversity and community composition influence nutrient and water cycling, and soil formation. How do plant and animal communities influence nitrogen leaching, total evapotranspiration or water infiltration, humus accumulation and podzolization, etc.?

Since biodiversity, structure and composition of communities change along gradients of human impact, an important issue is also how the dynamic status of ecosystems (their position on the human-impact gradient) influences their capabilities to produce goods and services. Ejrnaes et al. (2002, Ecological Applications) introduced the concept of 'nativeness', characterizing the position of plant community on the gradient of human impact (distance from an undisturbed state). There is a lack of information about how 'nativeness' of ecosystems influence services. For example, are nitrogen leaching and water runoff more intensive in strongly disturbed forest ecosystems (clearcuts), intermediately disturbed (plantations, economic forests) or in old growths, etc.?

We suggest that research priorities need to address the fundamental relationships between a) community diversity and composition and, b) ecosystems services (support services in particular). These studies should address the dynamic status of ecosystems, i.e. take into account different capabilities of ecosystems under different human pressures.

Key ecosystem process rates as predictors of service provision

Nikolai Friberg, Catchment Management Group, Macaulay Land Use Research Institute, UK

Applying a biological trait based approach to ecosystem service provision is a promising step towards a more process orientated understanding, but it has some limitations. The use of biological traits in nature is often still qualitative and based on a deduction from a species list which renders it somewhat static when compared to the more dynamic concept of "ecosystem services". The capability of ecosystems to provide a range of services will be more strongly related to dynamic features such as ecosystem organisation and fluxes of energy. A way forward would therefore be to carry out more process orientated research into food web architecture and to quantify how energy flows between the nodes within the web, which can then be related to the provision of services. This could be done by characterising the food web and mapping energy fluxes in the same type of ecosystem (e.g. streams, which I like!) along a well defined stressor gradient with known differences in ecosystem service provision.

Research priorities needed for ecosystem management

Richard Johnson, Swedish University of Agricultural Sciences, Sweden

To manage systems we need to define them first, which is easier said than done. A birds-eye view of the landscape is often needed to fully understand the systems that we are trying to manage.

Global patterns and changes in species diversity have been at the forefront of ecology for decades (e.g., Connell 1978; Vyverman et al. 2007). Yet, recognition of what drives these patterns has been difficult, despite recent attempts using theoretical frameworks like neutral ecological theory (Hubbell 2001) and the metabolic theory of ecology (Brown et al. 2004). At present, for many regions understanding whole-scale change in diversity is not feasible due to inadequate knowledge of species pools and the effects of spatial and temporal factors on distribution. Consequently, recent focus has shifted to try and understand loss of diversity at regional and landscape scales. Another reason for this shift in interest is that declines in regional species pools occur at a much higher rate than global extinctions (Hughes et al. 2000). However, understanding patterns in diversity at regional scales can be problematic, since human activity often alters the environmental drivers affecting these patterns. These problems are further exacerbated by climate change due to changing baselines and altering interactions between natural and human-induced stressors.

However, one of the most serious problems to understanding patterns in species diversity is simply our lack of knowledge or ignorance of ecological processes and mechanisms. In two earlier contributions, Martin Sharman questioned the use of the term ecosystem, outlining many conceptual weaknesses and perceptions, and Brian Moss questioned the idea of sorting nature into boxes and then quantifying their "value" - "one of the ways that we do this is to compartmentalize the biosphere into 'service providing units' like a bunch of shopkeepers, when our true nature is that of pirates." Both of these contributions echo arguments raised by O'Neill (2001) in his R.H. Macarthur award lecture "Is it time to bury the ecosystem concept? (With full military honours, of course!)". Unfortunately, these ideas are not percolating into policy and management.

To properly manage systems we need to understand the intricate linkages between systems (e.g. terrestrial-aquatic connectivity) and how these interactions are important for resistance and resilience to stress. Recent advances in landscape ecology and the use of ecologically-based, multi-species models is clearly a step in the right direction (e.g., Burman et al. 2005). The next step is to make this information readily available to managers through decision support systems. However, models require data and we still lack adequate data on many groups of organisms, knowledge of taxonomic coherence, and the importance of diversity for ecosystem function. Working with species has limitations and recent use of traits should be explored further, in particular studies focused on analyzing relationships between species' traits and taxonomic distinctness and predictions of trait responses to different types of stress.

Climate change enforces a paradigm shift in biodiversity conservation planning

Willemien Geertsema, Paul Opdam, Claire Vos and Koen Kramer, ALTERRA, the Netherlands

Current conservation planning defines targets for conservation sites that are based on historic references of vegetation types and local species occurrence or density (Margules et al. 1994). The long term effectiveness of such a static, site-oriented strategy is currently challenged by new scientific insight on ecosystem functioning (Gaston et al. 2006) and the unpredictability of the effects of increased perturbations caused by climate change (Mitchell and Hulme 1999). Therefore, a paradigm shift in biodiversity conservation planning is needed.

Three main challenges are identified. Firstly, the exclusion of disturbances in current conservation practice ignores the nonequilibrium nature of ecosystems (Holling 1973). Disturbances are increasingly considered on the one hand the base of co-existence of species and therefore of biodiversity. On the other hand disturbances allow ecosystems to adapt to changed environmental conditions and are considered a source of renewal. Preventing disturbances to occur and restoring ecosystems to its original form, if an inadvertent disturbance did occur, results in a loss of biodiversity and adaptive capacity. Disturbances should not be perceived as undesirable, but be incorporated as an integral part of biodiversity conservation planning.

Secondly, species differ in their sensitivity and response to different sources of disturbance. This is illustrated by the various responses to changing weather conditions due to climate change. The differences in the responses of species result in changing species composition of communities. Hence, the very base of current conservation policy that community types have high predictive capacity for the occurrence of target species, erodes away as climate change progresses. Instead of a focus on individual species as conservation targets for protected areas, the presence of functional diversity enabling differential responses to disturbances and enabling the continuation of ecosystem functioning should be the focus of biodiversity conservation planning.

Thirdly, most existing reserves are too small to incorporate long term and large scale dynamics (Bengtsson et al. 2003). Population studies in fragmented ecosystem patterns on the landscape scale have shown dynamic patterns typical for metapopulations (Hanski & Gilpin 1991; Verboom et al. 2001; Vos et al. 2001), implying that the local occurrence of species is often unpredictable and largely dependent on characteristics of ecosystem networks at the regional scale level (Opdam & Wiens 2002; Opdam et al. 2003). Also the consequences of climate change on species ranges cannot be controlled or counteracted by local management actions. Thus the scale of the individual reserve is too small to sustain nature quality targets. Instead of local sites as the object of planning, a landscape and regional approach in biodiversity conservation planning is needed.

Science has to play a key role in the paradigm shift. Science needs to provide evidence to societal partners about the effectiveness of dealing with ecosystems in a more adaptive way. Convincing cases, based on thorough science, of land use planning where biodiversity and ecosystems are considered in a more functional way are pivotal. Huge efforts are demanded before science can provide operational methods for goal setting, design and evaluation for regional planning. For example, a framework for the diagnosis of effect and response diversity should be developed. Ecosystems and ecosystem networks should be analysed for their key structures and processes and feedback mechanisms, based on such an analysis key functional groups are identified. Next the potential variation of functional groups needs to be explored and interpreted to generate a system of reference values for determining an operational framework for goal setting. We need to develop insight in the quantitative relation between the variation of functional groups, the adaptive capacity of ecosystems and ecosystem networks and the spatial characteristics.

The challenge to science is not only to make this information quantitatively applicable in a spatial context, but science should also be more effective in transferring this knowledge into societal decision-making than it has been up to now. Implementing the paradigm shift is a societal learning process. Science should be part of that, and learn from practical applications just as well as practice is learning from science. A key issue in this learning process is how to deal with uncertainty. We will not be able to predict exact levels of adaptive capacity, because the nature, frequency and severity of disturbances that ecosystems will be faced with are unpredictable. Ecosystems are moving targets, with multiple potential futures that are uncertain and unpredictable (Holling and Meffe 1996). What we intend to realize with this approach is that we learn, in the end, to manage ecosystems and landscapes in such a way that they are adapted to the unpredictability and uncertainty that we face.

Climate change DOES NOT enforce a paradigm shift in biodiversity conservation planning

Olly Watts, Climate Change Policy Team, RSPB, UK

Just because we will increasingly need to manage ecosystems and landscapes to be adapted to unpredictability and uncertainty, does not mean that we should be sidetracked into the superficial attractions of a paradigm shift for nature conservation.

As the above contribution shows, huge efforts are needed before science can operationalise and set goals for the paradigm shift they propose. Meanwhile, biodiversity conservation action now and our current suite of measures, such as the Birds and Habitats Directives, are providing a robust, if currently under resourced, framework for protecting Europe's wildlife from a wide manner of impacts alongside climate change.

Any nature site manager knows well enough the ebb and flow of species' fortunes, and the Directives allow considerable flexibility in their overarching objectives to contribute to biodiversity through the conservation of natural habitats and wild fauna and flora. To tar current conservation as a static, site based strategy is an over-simplification, and ignores the continually evolving developments to reduce fragmentation, build stronger ecological networks, use agri-environment, forestry planting and other landscape scale schemes to improve the ecological viability of the wider landscape, in which nature sites will continue to provide important, top-quality areas.

Species targets have served us pretty well, and I'd argue will continue to do so. They are not set in stone. They have changed in the past and will continue to do so in the future. National biodiversity target setting provides a spatial scale and dimension that can accommodate range and other shifts, helping to drive actions to accommodate change while also ensuring that we don't suffer death-from-a-thousand-cuts type losses.

And we should not forget that it is species that ultimately provide functional diversity and ecosystem services. And usually these are the more common species, simply because they are common, and therefore 'do more' in whatever we may require of them (apart from rare-species eco-tourism, of course!). Conservation tends to focus on the rare species, because they are rare. Focusing conservation on functionally useful species therefore brings the risk that rare species will fall by the wayside; yet, as is pointed out below, we live in an increasingly dynamic age - in which rare species may become common, and vice versa. Species conservation should therefore give us insurance for the future, as well as an approach that gives us the detail of ecological knowledge most often required to develop effective management - and to continually improve and advance that management, of course!

Conservation needs to remain vital and tangible and inspiring. Accordingly, it must relish new ideas and opportunities, which approaches like functional diversity and ecosystem services can bring. But these can, and should, fit alongside and add to the approach that has served extremely well for decades, and that has a strong legal foundation which provides an essential test for sustainability in the face of threats of a plethora of threats to biodiversity, of which climate change is but one, and in landscapes which most often have to accommodate a variety of human uses and needs.

And let us not forget that climate change is another man-made harm to biodiversity, and that much of the point of conservation is to redress man-made harm to biodiversity. The global community is now recognising the urgent need to get on track to mitigate to stay within ‘safe’ levels of climate change; which, if safe for our own species, will also limit to largely manageable levels its impacts for much other biodiversity.

Dynamic integration of humans and ecosystem service provision

Pam Berry, Environmental Change Institute, Oxford University Centre for the Environment, UK

Ecosystem services is a relatively recent, but rapidly expanding field of science (Vandewalle et al., 2008) which has led to a different perspective on the relationship between humans and their environment and, in particular, the importance of biodiversity. Currently, the quantification of many services provided by ecosystems is often minimal and the quantification that exists is based on purely direct and indirect monetary values. Such an approach often fails to identify the relevant actors involved in the service and their interaction. For example, humans are seen as the beneficiaries of ecosystem services, although they may, through inappropriate management practices, decrease the ability of the ecosystem to provide a (different) service. Humans also change their requirement for different services in response to external drivers (e.g. demand for biofuels versus agriculture), as well as personal factors, such as food or recreational preferences.

The dynamics of these links between human behaviour, species' attributes and supporting habitats in response to environmental pressures, determine whether ecosystem services are provided at an adequate and sustainable level. In order to enhance the concept's potential operational capability, research is needed on these ecosystem responses to human-induced changes in biodiversity dynamics and the actions necessary to maintain delivery of ecosystem function and services in the face of changes in climate, land use and social attitude. This could be achieved by integrating agent-based modelling of human behaviour with models of the species and habitat which provide the service. This could provide a much more robust understanding of how to safeguard ecosystem services for the future.

Buy one get one free?

Richard Harrington, Centre for Bioenergy and Climate Change, Rothamsted Research, UK

Conservation of biodiversity for the sake of ecosystem service delivery, and attempts to value ecosystem services to give them a less woolly status when it comes to competition with alternative land-use options, are approaches described by some as pragmatic and others as cynical. Most, though, see such approaches as complementary, not alternative, to conventional conservation programmes. A research activity that seems somewhat neglected is to investigate what such approaches carry with them in terms of species which may have no value in provision of the service for which the approaches were designed or, indeed, any other currently recognised service. If the answer is 'lots' the cynics may be assuaged.

No-regret strategies, flexible adaptation and monitoring

Rob Bugter, Alterra Landscape Centre, the Netherlands

Our knowledge on how, for instance, land use change and climate change will affect biodiversity and what this will mean for us in the future is still lacking, as is clearly indicated by the research priorities identified in previous contributions. While more research on mechanisms is undoubtedly needed, it is also clear that in the meantime conservation strategies need to be adapted to our present state of knowledge. At the same time, the lack of knowledge on the precise impacts of changing conditions for biodiversity calls for strategies with the highest possible 'no regret' factor. Part of that strategy is certainly flexibility and ease of adaptability to new situations, views and results, which requires close monitoring of the effects of measures.

Three research priorities follow on from this: 1) how to translate scientific uncertainty into the best no-regret strategies, 2) how to monitor the effects of measures in an adequate way, and 3) how to communicate the need for flexible, adaptable strategies and get them implemented.

The need for a common indication approach across ecosystems

José Paulo Sousa, IMAR-CIC, Department of Zoology, University of Coimbra, Portugal

To reverse the trend of biodiversity loss involves not only the need to identify indicators for the different components of biodiversity, and their use to indicate service provision and/or underlying ecological processes (stressed earlier by Christian Feld), but also the need to develop concerted indication concepts and approaches that could be used across ecosystems.

Specifically, this implies the need to develop schemes and guidelines to process information gathered from regular biodiversity monitoring programmes, which can help to assess and understand the effects of the major drivers affecting biodiversity.

Several of these schemes are common in the aquatic arena, with their development or implementation driven by the Water Framework Directive. Similar concepts have already been adapted for soils, and are already in practice in some EU countries and surely more research will be done with the (long awaited) approval and implementation of the Soil Framework Directive.

However, the discussion of a common indication concept and the delineation of a scheme that could be used across several ecosystems should be encouraged. Of course each ecosystem has its own specificities attached to the scheme (e.g., particular organisms or services, the type of biodiversity indicators to include etc.), but all have common "problems" to solve, that future frameworks need to take into account. These are to define reference conditions/sites, threshold values (apart from those which the status of the indicator can be considered at some level of risk), the scale of application, including structural and/or functional indicators, etc.

The discussion of these and other topics are very advanced in some ecosystems, but are only just beginning in others. To find a full, uniform approach applicable across ecosystems could be difficult, but a dialogue based on common concepts should be possible at the very least.

In summary, and to add some words on the type of indicators that could be included in such schemes, I would like to pick up on Francesco de Bello's contribution. I believe that the inclusion of traits in such an approach can help to find common response trends to particular stressors across ecosystems, or across regions within an ecosystem type.

This is, of course, widely applicable and known in plant ecology, but is only just beginning to be recognized in other groups. Interesting and promising approaches have been introduced in soil ecology (as indicators of soil quality) and in freshwater ecology and eco-toxicology (in this last case trying to depict common responses to similar chemicals when analysing freshwater invertebrate communities in different biogeographical regions). So, as pinpointed by Francesco, more research on traits (e.g., how to measure them) is necessary, and I believe the effort will be very compensatory!

Linking quantitative changes in service providers with impacts on service provision

Gary Luck, School of Environmental Sciences, Charles Sturt University, Australia

A critical research priority is to understand how quantitative changes in service providing units (organisms and their characteristics) impact on the delivery of services. For example, how does the population density or distribution of pollinators affect the delivery of pollination services? Is there a threshold population density above which a landholder obtains their entire pollination service from local pollinators (and below which pollination is substantially disrupted). A recent paper on this by Ricketts et al. (2008) identifies a common distance effect whereby pollination services are greatly disrupted when the supporting habitat of pollinators is at a certain distance from crops. Obviously this varies with context, but many more syntheses like Ricketts et al. (2008) are needed to identify generalities and exceptions.

Making biodiversity loss count in climate change damage cost assessment

Dominic Moran, Land Economy and Environment Research Group, Scottish Agricultural College, UK

The combined discussions have raised some interesting ideas that further the research agenda on defining goods and services and the validity of different ways to value them. Here I want to perhaps cut through some of the angst about precision and offer a question on how well we can use what we do know in a way that combines efforts to deal with biodiversity loss and climate change. Specifically I want to signpost a direction for getting what we know about biodiversity damage costs into important climate change calculations that influence how much we spend now on greenhouse gas mitigation. I will suggest that while we focus on biodiversity loss and the need to refine definitions of valuing ecosystem goods and services, we also need to keep an eye on how we feed what we know into this bigger agenda. Climate change discourses are currently trumping biodiversity debates, but here's a way to get stuck in (please pile in if I am wrong).

Most recently Rob Bugter wrote: "Our knowledge on how, for instance, land use change and climate change will affect biodiversity and what this will mean for us in the future is still lacking". I would like to raise a point and offer a challenge that is a pressing research priority. There are three stages to my issue. The initial question is: do our combined insights provide us with a good picture of what climate change is doing to global biodiversity? The answer is probably yes. The picture is less than perfect, but frankly, the way things are changing, we cannot be too precious about getting the best picture we have into the global policy area.

A way to barge into the wider policy debate is to ask (as a next step): can the economists help us translate this climate related damage (or damage from any other source) into a monetary equivalent? Let us leave aside for now the normative question of whether we should do this, and possibly be more pragmatic about what can drive policy. The answer is also probably yes. The big figures will not be perfect, but....

The final and more complicated step is have we fed these damage costs back into the debate on mitigating climate change? More specifically, the social (or shadow) cost of carbon calculation (SCC), which is likely to become more relevant in judging climate change policy in many countries? The answer is no. The SCC that emerges from the best forecast of the future damage costs is from emissions of a marginal tonne of carbon equivalent gas. If we translate these damage costs to a present value, then policy has an evidence base to judge how much to spend now in relation to future damages. But the SCC is only as good as the damage cost information that is fed into it, and currently the biodiversity damage picture is not that good. This is our own fault.

Put simply, if biodiversity damages have not been inserted into this calculation, then the cost of the carbon signal that will increasingly be used by policy, is not doing the job. This in turn potentially leads to fewer mitigating polices, which in turn leads to emissions and so on. It sounds simplistic, and it may well be, but SCC offers something for us to focus on pragmatically. Going forward, we can use it as a framework for collecting what we know now.

So, in a nutshell, the research priority is to collect an inventory of the best forecasts of climate/warming related biodiversity damages and to do the best job we can at valuing them, however we can. We then need to get these to figure in the integrated assessment models that currently drive SCC calculations. The European Commission is currently cooking up a "Stern" review on biodiversity loss or cost of inaction. This might be an interesting compendium of what we know. But the short timescale leads me to doubt it, and in any case, this was really a job for the research community, if only we come out of our own silos.

This is my view and I offer it as an aside to many insightful contributions in the sessions. Others may have different perspectives on carbon trading and surrogate carbon valuation. Some think these are ways for reluctant governments to cook the mitigation books. For an alternative perspective see anything recent by Paul Ekins (in The Guardian mostly!), or www.timesonline.co.uk/tol/news/uk/science/article1751509.ece. For the UK use of SCC see www.defra.gov.uk/environment/climatechange/research/carboncost/pdf/background.pdf. For insights into SCC and integrated modelling see www.ucl.ac.uk/~uctpa15/SOCIAL_COST_OF_CARBON.pdf

Ecological corridors and invertebrate conservation

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Research priorities need to reflect gaps in present scientific understanding in combination with the practicalities of acceptance and implementation by stakeholders. This underlies much of the thinking behind RUBICODE. With this in mind I draw attention to a couple of major gaps in present knowledge that directly affect all major ecosystem types and all scales of research, from local to pan-European and global.

1) The appropriateness and function of ecological corridors in dynamic ecosystems. The idea that "ecological corridors" should enhance the dispersion of organisms across landscapes, comes from theories in biogeography and population biology, and is now inherent in conservation biology. Corridors are generally considered to be "good" in conservation, but the truth is that there is little scientific research to back this up. Indeed linear structures in landscapes at any given scale can have a variety of influences, positive and negative. They can function as a conduit, implying directional flow (the standard interpretation of a corridor), or they can serve as a habitat, filter, barrier, source or sink (Samways, 2005). Thus what may serve as a true corridor for some organisms, or at certain spatial scales, may have an entirely different influence on others. Given the heavy reliance already being placed on corridors and networks in conservation management, we need to know much more about their functional dynamics.

2) Research to enhance the successful conservation of invertebrates and their ecosystem services. There is an urgent need to place a new and much greater emphasis on understanding the importance of invertebrate animals to conservation strategies and policy. Invertebrates form not only by far the greatest portion of biodiversity, but they also have a wide range of essential roles in ecosystem function and provide an equally wide range of ecosystem services, the majority of which have not yet been considered. Invertebrate communities often function at spatial scales different to those of normal human perception, so that here conservation needs to be considered from the "organism point of view". This includes the issues of networks and corridors referred to above. The importance and urgency of undertaking new conservation-related research on these animals has now been highlighted by the Council of Europe under the Bern Convention in the recently published European Strategy for the Conservation of Invertebrates, which also identifies further, necessary key actions and research priorities.

Analysis of the dependency of quality of life on ESS

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From the MA we know that there is a strong connection between ecosystem services (ESS) and human well-being. It is obvious that ESS contribute towards fulfilling basic needs such as nutrition, warmth or shelter. But well-being is more than the fulfilment of basic needs. Besides this objective component, it consists of a subjective component, which is related to how the basic needs are met (strategies) and also to which needs, beyond the basic ones, should be met. These two vary greatly among individuals. This research priority now asks (1) for a thorough analysis of which ESS are contributing what, to which needs and (2) to study the impacts of reduced biodiversity or disturbed ecosystems on quality of life. This research will preferably be carried out on a conceptual and empirical basis (action research) in different regions throughout Europe and the rest of the world.

Standardizing indicators, monitoring and data quality: do we need another directive?

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As the RUBICODE indicators review shows, indicators come in various shapes and, more importantly, mass. However, in the context of making and comparing biodiversity assessments over large spatial scales, there is no need to have a large number of options but instead, a limited set of standardized indicators used universally. Over the last few decades, this need has led to several indicator projects and initiatives (e.g. EEA 2001, 2005, 2007) attempting to create some sort of order. The resulting indicator frameworks mostly use indirect indicators (i.e. indicating for biodiversity conditions rather than biodiversity itself). Apart from the fact that indirect indicators are often a lot more cost effective, a very important reason is also the lack of availability of standardized measurements and monitoring schemes for biodiversity itself against the more available standardized indirect data from sources like CORINE. However, one problem with this approach is that the indicative value of the indirect measures for biodiversity is very tenuous, especially at coarse spatial resolution. Where this approach might work reasonably for, e.g. forest surface and fragmentation, work on projects concerned with the agricultural landscape like EURURALIS (www.eururalis.eu) are, even at a very broad scale, seriously hampered by the lack of data, because agricultural biodiversity mainly depends on the amount and configuration of small landscape elements and on Land Use Intensity. Since agricultural landscapes cover over 50% of the EU surface area, they are therefore the dominant type connecting all other types and are the most prone to changes due to socio-economic developments. This is a serious problem.

Ideally, biodiversity as well as the conditions needed for it should be monitored and linked together in a standardized way everywhere in Europe or, even more ideally, in the whole world. Indirect and direct measurements (and the indicators for them) could then easily be generalized for larger spatial scales, resulting in a much better base for assessments and predictions on every level. Since we are currently still very far from this ideal situation, the questions can be asked: can we get there, and if yes, how do we get there?

Monitoring and linking biodiversity and biodiversity conditions in a standardized way in the first place requires standardized data collection. Assuming reaching consensus on what data to collect and how to do it is not beyond us, the perspectives on getting a standard system off the ground still does not seem likely for a number of reasons: 1) Although countries are now under the obligation to monitor their Natura 2000 areas, political climate does not seem to allow the EU to stretch that obligation as far as using the same indicators and monitoring methods. 2) For the agricultural landscape, we linked biodiversity to biodiversity conditions for temperate Europe’s agricultural landscapes in the GREENVEINS project (Billeter et al. 2008), but collecting fine resolution data in a standardised way with sample points covering large spatial areas is very rare and does not have any priority. On the other hand, combining virtually incompatible data from existing sources has been carried out, apparently due to the misguided belief that this will be more cost-effective. 3) Despite initiatives from e.g. EEA resulting in new data sources like CORINE, we are still a long way off from good quality data sources about structure and intensity of landscape use. However, standardizing the incorporation of landscape information in all countries topographic maps should be accomplishable, provided the willingness is there.

While defining a standard indicator and monitoring framework, including the standard data needed for it, seems accomplishable, getting it implemented will require political willingness. However, biodiversity conservation and environment were apparently of urgent enough concern to create the willingness to implement the Habitats, Birds, Water Framework and the coming Soils directives. Since there will be an enormous gain for these existing directives as well as for climate change adaptation and mitigation policies etc. from good quality biodiversity assessments and predictions, the question can be asked: should we not give priority to research demonstrating that gain, and increase awareness of that fact? Or in other words: should we be aiming research at convincing policy makers that we need a Biodiversity Monitoring and Assessment Directive (B-MAD)?

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