HELMHOLTZ
ZENTRUM FÜR
UMWELTFORSCHUNG
UFZ

UFZ-Diskussionspapiere

Department Ökonomie

5/2009

Integrating knowledge from social and natural sciences for biodiversity management: the asymmetric information trap

Frank Wätzold, Helmut Haberl, Hanne Svarstad, Wouter van Reeth, Rehema White

June 2009

Herausgeber:
Helmholtz-Zentrum für Umweltforschung GmbH - UFZ
Permoserstr. 15
04318 Leipzig

ISSN 1436-140X

Integrating knowledge from social and natural sciences for biodiversity management: the asymmetric information trap

Frank Wätzold, Helmut Haberl, Hanne Svarstad, Wouter van Reeth, Rehema White

Abstract

Most problems related to biodiversity management have an ecological as well as a socio-economic dimension. Consequently, there has been a growing recognition that adequate management recommendations directed at such problems can only be developed if knowledge from ecology, economics and various social science disciplines is taken into account in an integrated manner. To respond to the need for integrated research, a number of approaches have been proposed over the last decade or so with the aim of integrating knowledge from the natural and social sciences. These approaches emerged in different contexts and have integrated different disciplines. As the recognition of the need for integrated research is rather recent the approaches that integrate natural and social sciences are still in a phase of development. In order to further this development, a better understanding of how to tackle specific challenges that arise when knowledge from different disciplines is integrated may be helpful. The aim of this paper is to contribute to this task by analysing and comparing how selected approaches cope with one key challenge of integration: ensuring that state-of-the-art knowledge from both disciplines is used in the integrated approach. We selected the following approaches for comparison: Ecological-economic modelling, political ecology, the resilience approach, multi criteria analysis, and methods of material and energy flow accounting (MEFA) of socio-ecological systems. We selected these approaches because there is already a significant amount of literature that can be referred to and because they represent integration of different disciplines. For our analysis we used an economic approach: we consider the incentive structure of researchers and focus on asymmetric information between researchers from different disciplines about the quality of scientific research of the involved disciplines and the worldviews behind scientific approaches. We find that in order to attract high quality researchers the integrated approaches need to be attractive to researchers from both disciplines (I) in terms of generating funding opportunities, (II) of publication opportunities in highly ranked journals accepted in each specific discipline and (III) in helping to solve problems related to conservation policies that are of interest to all involved researchers. Approaches that do not fulfil these conditions have to struggle with the problem that they attract researchers of low scientific quality which they cannot identify. They need to be aware of this trap. A possible solution may be to put particular emphasis on external reviews by independent researchers.

Contact:
Dr. Frank Wätzold, Department of Economics, Helmholtz Centre for Environmental Research - UFZ, frank.waetzold@ufz.de

I. Introduction

Most problems related to the sustainable use and conservation of biodiversity (henceforth refereed to as biodiversity management) have an ecological as well as a socio-economic dimension. Consequently, there has been a growing recognition that adequate management recommendations directed at such problems can only be developed if knowledge from ecology, economics and various social science disciplines is taken into account in an integrated manner (cf. Wätzold et al. 2006). It is not sufficient that scientists work in their own disciplines and combine their knowledge only when it comes to formulating recommendations for biodiversity management. Such an approach does not capture feedback loops between the ecological and the socioeconomic system (e.g. Quaas et al. 2007). In addition, each discipline poses the management problem in its own way and comes up with its own most appropriate solution. These disciplinary solutions, however, are likely to be so different that a combined solution considering aspects of both disciplines cannot be found (Wätzold et al. 2006).

To respond to the need for integrated research, a number of approaches have been proposed over the last decade or so with the aim of integrating knowledge from the natural and social sciences. These approaches emerged in different contexts and have integrated different disciplines. As the recognition of the need for integrated research is rather recent the approaches that integrate natural and social sciences are still in a phase of development. In order to further this development, a better understanding of how to tackle the challenges that arise in the context of integrating disciplines may be helpful.

In this paper we focus on one particular challenge: to ensure that state-of-the-art knowledge from both disciplines is used in the integrated approach. For the analysis we employ an approach which is often used in economic analysis – we consider the incentive structure of researchers for integrated research and focus on asymmetric information between researchers from different disciplines about the quality of scientific research and the worldviews behind scientific approaches.

We selected the following approaches for comparison: Ecological-economic modelling, political ecology, resilience approach, multi criteria analysis, and methods of material and energy flow accounting (MEFA) of socio-ecological systems. We selected these approaches because there is already a significant amount of literature that can be referred to and because they represent integration of different disciplines.

We do not carry out a comprehensive analysis of the approaches, but focus our analysis on the challenges for integration. The framework for analysis is outlined in Section two. Section three contains a description of the approaches. In Section four the approaches are analysed on how they cope with the challenges of integration, and Section five concludes.

2. Focus of analysis

A key task for successful integration is to ensure that state-of-the-art knowledge from both disciplines is used in the integrated approach. In order to understand the challenges that arise for the various integrated research in this respect we employ an economic approach: we consider the incentive structure of researchers and focus on asymmetric information between researchers from different disciplines about the quality of scientific research and the worldviews behind scientific approaches.

Our starting point is the incentive structure for researchers to participate in integrated research. It is reasonable to assume that researchers are potentially motivated to participate in research activities – be it disciplinary or integrated research – by three factors:

(I) The prospect of publications in well established journals. These will often be publications in journals related to ones own discipline as promotion, scientific career and reputation often depend on publication records in such journals.

(II) Third party funding for scientific projects as researchers' reputation, salary and promotion also often depend on it.

(III) A personal interest in preserving biodiversity and improving related policies. Hence researchers may be interested in the policy relevance and applicability of their research.

We do not assume that all researchers are motivated by all three factors equally. We rather assume that individual researchers are influenced by these factors to different degrees and that these factors influence their willingness to participate in the five integrated research concepts analysed in this paper in different ways (as will be outlined below).

Furthermore, we consider that scientists from one discipline only have a very limited knowledge about the content, methods and quality of research from other disciplines. They are also unable to judge to what extent different world views are behind certain approaches. In economic terms, there is asymmetric information between researchers of one discipline and researchers of other disciplines in terms of what represents state-of-the-art knowledge and world views. This means that social (natural) scientists often do not know whether the natural (social) science knowledge their partners use in integrated research represents well established knowledge from this discipline. There are, of course, also often debates and disagreements among scholars from one discipline about what is the best scientific approach to analyse a certain problem and the relevance of certain problems. However, in this case, scientists have a much better understanding of the different arguments for and against different approaches.

The requirements of integration and hence the incentives for the researchers to participate in the various approaches differ depending on the management questions they address, the methods they use and the history of the approaches. In order to better understand requirements for integration and the resulting incentives we address the following questions when describing the approaches in the section 3:

1. What are the discipline and the particular approach within the disciplines from the social science and the natural science respectively in the integrated approach?

2. Are both disciplines interested in a common management question?

3. Do both disciplines share similar or common methods?

4. What was the starting point of the approach? Did both disciplines start off together or did one discipline come later? Why did it come later? What were the reasons for integration?

5. Is one discipline dominant? If yes, why and in what sense?

6. What is the expected future development of the integrated approach? May it move towards a new integrated discipline, like biology and chemistry moved into biochemistry?

3. Description of approaches

3.1 Resilience theory

Resilience theory emerged from ecology in the late 1960s and early 1970s from the study of predator-prey interactions and their functional responses in relation to ecological stability theory (see Folke 2006 for review; Janssen and Ostrom 2006, Janssen et al 2006, Walker et al 2006). A seminal paper by Holling (1973) on resilience and stability in ecological systems demonstrated the existence of multiple stability domains and their relation to random events and heterogeneity of temporal and spatial scales. He proposed that "resilience determines the persistence of relationships within a system and is a measure of the ability of these systems to absorb changes of state variables, driving variables and parameters, and still persist" (Holling 1973, p17).

Initially, empirical evidence for this theory was lacking, but ecological studies on boreal forest dynamics (Holling 1978, Ludwig 1978), rangelands (Walker et al 1981) and lakes together with resource management consequences soon provided data for mathematical modelling and further theory development. Resilience theory became the theoretical foundation for work on active adaptive ecosystem management (Holling and Chambers 1973, Holling 1978).

It then began to influence social sciences, including anthropology, ecological economics, human geography, environmental psychology, and resource management research (for reviews see Folke 2006, Scoones 1999, Abel and Stepp 2003, Davidson-Hunt and Berkes 2003). Resilience theory continued to develop in interdisciplinary debate related to social learning, sustainability science and other areas (Folke 2006). There was also a parallel evolution of similar ideas in fields such as psychology (Deveson 2003) and mental health (Walsh 2003).

Ecosystem resilience has been used to explore the capacity of a system to absorb disturbance, and the functions of biodiversity (eg Folke et al 1996, Bellwood et al

2004). The concept of resilience has also been used in relation to social change (Adger 2000), institutional diversity (Ostrom 2005) and when expanded to social-ecological systems acknowledges adaptation, learning and self-organisation (Folke 2006). Social-ecological resilience thus includes the amount of disturbance a system can absorb and remain within the same domain, the degree to which a system is capable of self organisation and the degree to which a system can build and increase its capacity for learning and adaptation (Carpenter et al 2001).

Resilience in this sense provides a framework or approach with which to explore social-ecological systems, particularly how humans affect the resilience of ecosystems (Janssen and Ostrom 2006) and can be combined with other more specific theories to test and predict system responses. It offers a systematic methodology for understanding the dynamics of social-ecological systems, rather than a theory to explain the behaviour of social ecological systems (Anderies et al 2006).

To explore the potential of this new paradigm for interdisciplinary research the Resilience Network was founded. This research programme later developed into the Resilience Alliance, a consortium of research groups and institutes interested in exploring social-ecological concepts (Folke 2006). Subsequent work includes studies on social capital, social learning, knowledge systems, participatory processes, networks and institutions (Folke 2006). This research is still exploratory; challenges include assessment of feedbacks and clarifying responses across scales. However, there is evidence that scholars of resilience are beginning to interact more with other knowledge domains and integrate theories more widely (Janssen et al 2006).

Resilience theory is seen to be a developing framework and has recently been used as a conceptual approach to guide interdisciplinary studies on the dynamics of social-ecological systems (Walker et al 2006, Anderies et al 2006). It can address issues that have been neglected by other theories, such as multiple interacting scales (Anderies et al 2006). The theory has been applied to many case studies, varying across a spectrum from largely ecological to largely social (eg see Anderies et al 2006). As studies have shifted from a focus on ecological elements, there has been a change in approach from mathematical modelling to the collection and analysis of qualitative data (Janssen et al 2006).

Resilience theory offers a conceptual approach for the investigation of social-ecological systems. It can be used to guide research across a broad range of fields and encourages the development of interdisciplinary approaches (Anderies et al 2006) such as e.g. social-ecological network analysis. However, it is currently a framework and substantial research needs to be undertaken to test the propositions that are emerging from recent work (Walker et al 2006) and to develop more specific accepted uses.

Resilience theory has major implications for natural resource management and policy. Its acceptance in mainstream research and policy fields would alter the current emphasis on the management of resources as controllable, steady state, single equilibrium parameters to an acknowledgement that resources form part of complex adaptive systems (Anderies et al 2006). There would be a paradigm shift to management for change and unpredictability. It also offers managers and policy makers an approach to view systems in a more holistic manner than current perspectives of ecological resources, social issues and economic drivers.

There are also particular management applications emerging. For example, critical changes in social-ecological systems tend to be determined by a small set of three to five key variables and identification of these in a particular system would greatly assist understanding of change (Walker et al 2006). Slowly changing variables control ecological resilience but social resilience is controlled by either fast or slow variables (Walker et al 2006). Implications for governance strategy are that resilience can be increased by participation, deliberation, multilayered institutions and accountable authorities (Lebel et al 2006). Learning should be encouraged, since the capacity to effectively combine knowledge from different sources, including tacit and formal knowledge, increases the likelihood that key thresholds will be recognised (Berkes and Folke 1998) and aids response to loss of ecological resilience (Gunderson et al 2006).

The capacity of social-ecological systems to self-organise is also critical for resilience (Berkes and Turner 2006). Since rebuilding can be helped by external capital, subsidisation from external sources can aid systems in recovery after collapse, but theoretically such subsidisation should stop once self-organisation is apparent (Abel et al 2006). Resilience theory also has the potential to aid managers to understand regime shifts within systems by viewing cascading thresholds across different domains (social, economic and ecological) and scales (patch, farm and regional) (Kinzig et al 2006). In some cases, low resilience may derive from a mismatch of governance and ecological scale (Cumming et al 2006). In a resilient social-ecological system, disturbance has the potential to create opportunity for doing new things, for innovation and for development (Folke 2006, Lebel et al 2006).

3.2 Mulitcriteria analysis

Multicriteria analysis (MCA) emerged between the end of the 1960's and the early 1980's, as a new generation of decision-supporting evaluation methods. Its roots are in the field of operations research, a branch of applied mathematics. MCA aims to evaluate a set of alternative solutions to a problem, on a set of pre-defined objectives. From these objectives criteria are derived on which each of the alternative solutions is scored. The scores can be expressed both in quantitative (monetary or other) and in qualitative measures. MCA is typically used for evaluation problems that concern a discrete set of alternatives, such as environmental impact assessments, the location of land consolidation projects or alternative trajectories for transport infrastructure.

Following Janssen (1984) and Van Huylenbroeck (1987) a MCA typically consists of a definition, a research and an evaluation stage. During the definition stage a set of (policy) objectives that is relevant for the problem under investigation is selected. The objectives may consist of different hierarchical layers and are typically collected from policy documents or via consultation procedures from policy makers and/or experts. For each of these objectives one or more criteria are selected upon which the alternatives can be compared. Each criterion represents an indicator that may show the monetary costs and benefits of the alternatives. However, it may also express non-monetary consequences in quantitative or in qualitative terms. In addition to the objectives and criteria also a set of alternative solutions is defined. The definition stage results in an evaluation matrix in which the rows are defined by a hierarchical set of objectives and criteria and the columns are defined by a discrete set of alternatives.

During the research stage information is gathered to obtain a quantitative or qualitative score for all criteria for each alternative. If MCA is used as part of an ex ante-evaluation the indicators (i.e. scores of the alternatives) need to be estimated via a modelling or forecasting method. If MCA is part of an ex-post evaluation data on the indicators are collected from direct or indirect field monitoring. The scores can be expressed in a quantitative (ratio or interval scales) or qualitative (ordinal scale) units. During the research stage information on the weights or relative importance of each criterion and objective also needs to be collected. The relative importance of the objectives needs to reflect the preferences and values of the stakeholders involved. The weights of the criteria are typically assessed by experts from the field or discipline to which a criterion relates. The criteria and objectives can be given a quantitative weight. They also can be given a qualitative weight, for instance by ranking them as more, less or least important. The output of the research stage is the evaluation matrix with criterion scores for each alternative, together with weight vectors for the criteria and objectives.

During the evaluation stage the alternatives are first compared and ranked for each of the criteria, per objective. If the objectives are strongly conflicting the evaluation may end here. In that case the rankings of the alternatives are aggregated across the criteria, based on the criterion weights, for each objective. Alternatively the evaluator may also wish to aggregate the evaluation across the objectives to present an overall ranking of the alternatives, based on the weights of the different objectives. In that case the MCA may result in a general recommendation identifying the best alternative, a partial ranking of the alternatives, or a set of alternatives that meets pre-defined requirements. It is this third evaluation stage that constitutes the actual 'core' of MCA. Depending on whether the scores and the weights are expressed in quantitative or qualitative measures the data can be aggregated via quantitative or qualitative MCA-methods.

MCA reduces the complexity of decision-making by structuring the selection and comparison of alternatives. By making the trade-offs between conflicting objectives and criteria more transparent it also tries to reduce the ambiguity of the decision-making process. In this way it supports the selection of that alternative (or those alternatives) that best meet pre-defined objectives. Prior to MCA, cost-benefit analysis (CBA) was a standard approach for the comparative evaluation of discrete alternatives. Contrary to CBA, however, MCA is not limited to comparing alternatives on monetary criteria alone. The analysis on multiple criteria and varying weights is better suited to incorporate the increasingly complex and conflicting nature of policy making. The rising participation of pressure groups and stakeholders in various policy domains increases the demand for evaluation methods in which the importance of alternative criteria can be assessed. At the same time MCA allows some 'degrees of freedom' to policy makers by including a range of acceptable or defendable solutions rather than presenting them with one 'take it or leave it' advice (Janssen & Nijkamp, 1998).

In densely populated and increasingly industrialized Europe nature and biodiversity have increasingly been entangled in struggles for space with other sectors as agriculture, fisheries and urban planning. MCA's tendency to look at spatial choice problem from a variety of angles and stakeholders' interest has made it an increasingly used tool for this aspect of nature policy. It therefore appears to be most relevant for spatially sensitive policy choices.

MCA's highly systematic approach, in which seemingly unquantifyable problems are given a quantitative or at least highly analytical appearance, has given it certain appeal as hallmark of objectivity and validity. The application of MCA nevertheless involves several choices which have an impact on the results of the evaluation, including the ranking of the alternatives. Proponents of MCA tend to argue that MCA doesn't exclude political choices but instead makes them more transparent. More critical observers argue that MCA's quantitative techniques mask rather than clarify the implicit choices and assumptions, especially for the majority of non-specialized users.

MCA can be and has been used as a vehicle to bring together knowledge from various disciplines and interests from competing sectors. In a time where interdisciplinary research and multi-actor government are advocated, MCA offers the potential to help researchers, policy makers and other stakeholders to interact and find a common discourse. This is considered a necessary albeit insufficient requirement for good governance.

3.3 Ecological-economic modelling

In ecological-economic modelling knowledge from two disciplines, ecology and economics, is combined in mathematical or computer-based models. Ecological-economic modelling is mainly applied in two fields: management of renewable resources and design of conservation policies and strategies. Especially in renewable resource management it is also often referred to as bioeconomic modelling. For the purpose of simplification in the following we only speak of ecological-economic modelling.

In designing conservation policies a prominent field of applying ecological- economic modelling is the optimal selection of reserve sites. Early analysis (Ando et al. 1998) started from the observation that in conservation biology typical approaches to selecting reserves neglect economic aspects of the problem, in particular, that there are often large cost differences among sites. Ando et al. took into account these cost differences and solved a budget-constrained reserve site selection problem using data on the locations of endangered species and average land value by county for the United States. They found that the costs of achieving a given level of species coverage were far lower with the budget-constrained approach than with traditional ecological approaches.

Since then ecological-economic models have been applied to analyse different aspects of the reserve site selection problem (e.g. Polasky and Solow 2001 and Drechsler 2005). Next to addressing the reserve site selection problem ecological-economic modelling has been applied to analyse how compensation payments for biodiversity-enhancing land-use measures should be designed to be ecologically effective and cost-effective (e.g. Johst et al. 2002, Groeneveld 2004, Wätzold and Drechsler 2005, Drechsler et al. 2007a). Other research does not focus on a particular policy instrument but addresses issues such as conflicts between conservation and human activities (e.g. hunting, see Skonhoft et al. 2002) or compares different conservation approaches such as state-dependent and static conservation management (Drechsler et al. 2006).

Ecological-economic modelling has played a prominent role in understanding the factors that affect renewable resource exploitation and to develop recommendations to achieve a sustainable resource yield. This field of application of ecological-economic modelling is much older and broader than the design of conservation policies. A prominent example for early work is Clark (1976) and an example for research in fishery is Pezzey et al. (2000), in forestry Sankhayan et al. (2003) and in grassland management Quaas et al. (2007).

Still, the overwhelming work in these areas is probably disciplinary and for many models it is not easy to say whether they should be considered a disciplinary model or an ecological-economic model. E.g. most economic models about renewable resource management include some ecological knowledge but this knowledge often does not represent the state-of-the-art in ecology. For example, Sanchirico and Wilen (1999) state that most of the fisheries economics literature still uses the Shaefer model in which biomass dynamics are characterised by the intrinsic growth rate and the carrying capacity of the environment. However, modern ecology emphasises much more spatial aspects which is not covered by the Shaefer model. Such criticism has been taken into account and there are now a number of fishery models that include spatial aspects (see Armstrong 2007 for an overview).

Most authors applying ecological-economic modelling are not explicit about whether their analysis is meant in a normative or positive way. Many models can be used for normative and positive analysis. However, the purpose of some models is more towards understanding, e.g., in cases where the impact of open access on harvesting of fish and fish stocks is analysed. Other research has a stronger normative component. E.g., developing recommendations for designing conservation policies has a strong normative element.

Fundamental similarities exist in the problems addressed by economics and ecology which makes it less difficult to combine their knowledge in models. For example, both disciplines are interested in the optimal use of limited resources. Ecologists explore how plant and animal species maximise their reproductive success and survival being confronted with limited food and other resources, whereas economists examine how humans maximise their well-being given a budget constraint (Shogren et al. 2003). Another question addressed by both disciplines is the study of stability properties of a system. Economists investigate the equilibria of systems related to the economy, whether those equilibria are stable and in which direction the system's state changes when certain constraints and parameters are altered. Ecologists are similarly concerned with stability, except they usually do not assume that their system is in equilibrium (be it static or dynamic), and they also allow for complex system behaviour such as cycles, chaos, or a variation of key state variables within certain boundaries (e.g. Grimm and Wissel 1997).

In ecological-economic modelling a variety of approaches are applied. There exist more conceptual models which are based on mathematical equations and which can be solved either analytically (e.g. Baumgärtner 2004) or numerically (e.g. Wätzold and Drechsler 2005). More applied models are frequently rule-based models or at least their ecological sub-model is of that type (e.g. Johst et al. 2002). An analysis of the type of modelling that is predominantly used in ecological, economic and ecological-economic models dealing with conservation is given in Drechsler et al. (2007b).

3.4 Political ecology

Political ecology is a research tradition that focuses on issues regarding the management of natural resources and the environment. Often particular conflicts provide the point of departure for studies. Being an inter-disciplinary tradition, political ecology integrates social and natural science elements. In contrast to much inter-disciplinary research in this field, political ecology is a tradition in which qualitatively oriented social sciences play an important role. The "ecology" part of political ecology" implies a broad focus on bio-physical environments. The "political" part of the term has origins linked to the tradition of "political economy", and it also signals an emphasis on power dimensions. It does not provide one specific political view, and deviates substantially from the expression of strong political opinions without analysis. Watts (2000:257) holds the goal of political ecology to be to explain environmental conflict especially in terms of struggles over "knowledge, power and practice" and "politics, justice and governance". He defines the purpose of political ecology "to understand the complex relations between nature and society through a careful analysis of what one might call the forms of access and control over resources and their implications for environmental health and sustainable livelihoods" (Watts 2000:257).

Political ecology is a relatively new research tradition. Work by the geographers Blaikie (1987) and Blaikie and Brookfield (1987) constitute some of the most important early contributions. Today, political ecology holds a strong position among social science disciplines such as human geography and anthropology, and there are also substantial contributions to political ecology from natural sciences such as natural geography and biology. Walker holds that political ecology has today "become firmly established as a dominant field of human-environmental research in geography" replacing its predecessor cultural ecology (Walker 2005: 73). Most political ecology so far has concentrated on subsistence producers in rural areas of poor countries. However, during the last few years scholars have started to apply political ecology to frame studies also in North America and Europe (McCarthy 2005, Schroeder et al. 2006, Benjaminsen and Svarstad forthcoming).

Political ecology has evolved as a tradition in which the necessity has been much stressed of exceeding disciplinary borders and particularly conducting research that exceeds the division between social and natural sciences. In a recent textbook in political ecology, the ambition of political ecology is described as arising "from its efforts to link social and physical sciences to address environmental changes, conflicts, and problems" (Paulson *et al.* 2005: 17). Likewise, another recent book on political ecology "seeks to advance debates about integrating social and biophysical explanations of environment” (Forsyth 2003). However, many contributions to the political ecology literature come from social scientists who do not themselves contribute to exceed this division.[1] Nevertheless, a number of studies go all the way in this ambition (e.g. Blaikie and Brookfield eds. 1987, Fairhead and Leach 1996, Sullivan 1998, Zimmerer and Bassett eds. 2003, Forsyth 2003, Benjaminsen et al. 2006).

Looking at the normative frames of political ecology, there is often an emphasis on the value of local and indigenous knowledge and especially the ability of local people to act in ecological sustainable ways, if given the opportunity. Furthermore, aspects of power and politics are often identified to obstruct good local solutions and cause harm to marginalised people. Moreover, many political ecology studies have contested claims of irreversible degradations (e.g. Rosin 1993 [ref. by Robbins:13], Stott and Sullivan eds. 2000, Paulson et al. 2004, Robbins 2004, Neumann 2005).

The social science research within political ecology emphasises qualitative examinations aiming at obtaining a deep understanding of the focused issues in terms of perceptions held by actors and their discursive frames as well as structural features related to the management of natural resources. Political ecology has incorporated a social constructivist perspective, and often more specifically with a Foucaultian orientation. This enables the researchers to take into account the ways knowledge on natural resources and the environment are structured by discursive “rules” that divides between knowledge empowered as relevant on the one hand and silenced aspects and “taken-for-grantednesses” on the other. This critical examination of discursive power in the way issues are approached by various actors constitutes a strength of the political ecology tradition (Peet and Watts 1996, Stott and Sullivan eds. 2000, Adger et al. 2001, Martinez-Alier 2002). Nevertheless, in many political ecology contributions there is a lack of self-reference to own positions. Thus, a strong discursive position often constitutes the point of departure, while this framework is itself not subject to critical examination.[2]

A strength of political ecology is that the social constructivist position is applied not only on social dimensions but also on natural science knowledge on the biophysical reality. In other words, natural science elements are subject to deconstruction and discursive contextualisation. Blaikie and Brookfield, for instance, argued the necessity “to examine critically the political, social and economic content of seemingly physical and ‘apolitical’ measures such as the Universal Soil Loss Equation, the ‘T’ factor and erodibility” (1987:xix). Similarly, in Forsyth’s recent contribution to political ecology, he seeks “to establish the political forces behind different accounts of “ecology” as a representation of biophysical reality” (2003:4). Sullivan advocates conceptual exchanges between a biophysical science focusing on both structure and change in living complexes, and an actor-oriented applied social science grappling with conflicts between local dynamics and national or global structures, and with an emphasis on hegemonic environmental discourses (Sullivan 2000).

[1] In a paper by two human ecologists, political ecology was argued to be based on political arguments and not on ecology (Vayda and Walters 1999), and an often lacking weight on ecology within contributions to political ecology is also criticised by Walker 2005.
[2] The “liberation ecology” by Peet and Watts (1996) can be seen as a strand of political ecology towards which this criticism is particular relevant.

Establishment of a historical understanding is seen as important within political ecology. This encompasses history of a conflict and its social aspects as well as history of the natural conditions. Changes often play an important role for the conflict, whether these are changes in practices, changes in the natural conditions or combinations. Studies based on non-equilibrium ecology have provided important contributions by identifying environmental changes as non-linear, non-cyclical and chaotic (Neumann 2005).

Furthermore, understanding that exceeds scales constitutes an important feature of the political ecology tradition. This implies the view that an issue cannot be understood satisfactory only by, for instance, research at the local level. Instead, contextualisation beyond scales is seen as prerequisite (Paulson and Gezon eds. 2004). In Blaikie's seminal work on political ecology, he suggests to first focusing on a local land manager, his relationship to the land and what the effects of his practices leads to on the land itself[3]. Thereafter, influences on the land manager should be studied according to "chains of explanations" on a gradually extending scale (Blaikie 1985, Blaikie and Brookfield 1987). However, political ecology studies may also start on a global level, for instance with leading discourses on an issue as a point of departure (Svarstad 2004).

3.5 The Material and Energy Flow Accounting (MEFA) approach

In the last two decades, the analysis of materials and energy flows related to socio-economic processes has gained importance as one approach to analyze socio-ecological systems, i.e. systems that emerge through the interaction of societies with their natural environment (Berkes and Folke 1994, Fischer-Kowalski and Haberl 1997, Sieferle 1997). Basically, the idea is that societies depend on a continuous flow of energy and materials from their natural environment, which are then transformed in economic production and consumption processes and returned to the environment as wastes and emissions. Obviously, this process, often denoted as "socio-economic metabolism" (Ayres and Simonis, 1994, Fischer-Kowalski, 1998, Fischer-Kowalski and Hüttler, 1998, Martinez-Alier, 1987) depends on socio-economic settings such as production systems, economic structure and growth, political system, institutions, infrastructure, etc. and is ecologically highly relevant.

While accounts of economic energy flows are traditionally reported as integral part of national and international statistics (e.g., UN, 1997, IEA, 2001), material flow accounts (MFA) are only recently being incorporated in national and international statistical databases (e.g., Eurostat, 2002, Weisz et al., 2005). Complementary methods of Energy Flow Accounting (EFA) that use the same system boundaries as MFA does are currently discussed and applied in the scientific literature (Haberl, 2001a, Haberl, 2001b, Haberl et al., 2006, Krausmann and Haberl, 2002). Some of these resource flows are obviously relevant for biodiversity, above all fossil fuels – because their combustion results in greenhouse gas emissions and induce climate change, a major driver of biodiversity loss (Sala et al., 2000) – and biomass, the provision of which requires land use, a major driver of biodiversity change as well (Sala et al., 2000).

[3] Thematically, Blaikie has concentrated his studies on land degradation.

Impacts of fossil fuel use on biodiversity largely act on a global level: Emission of CO_2 and other fossil-fuel derived greenhouse gases results in global climate change, and this process in turn affects biodiversity. This implies that fossil fuel use need not result in local biodiversity changes. In contrast, land use directly alters biotic communities at the very locality at which it takes place. An aggregate measure of land-use intensity is the human appropriation of net primary production (HANPP). In contrast to MFA and EFA which only account for socio-economic flows, HANPP is based on the measurement of socio-ecological material (biomass) flows, i.e. flows through the larger system including both socio-economic and ecological compartments. Basically, HANPP measures changes in the availability of net primary production (NPP), i.e. the biomass generated in an ecosystem per year through photosynthesis, for ecosystem processes.

To understand HANPP, it is necessary to remember that, in using the land, humans alter the production ecology of ecosystems in two interrelated ways: (1) by changing the productivity (NPP per unit area) of ecosystems and (2) by harvesting parts of the NPP. Both processes result in an alteration of the amount of NPP available in ecosystems as compared to their original status. HANPP is an indicator for land-use intensity based on the measurement of changes in the availability of trophic (biomass) energy in terrestrial ecosystems induced through land-use induced changes in productivity and harvest.

HANPP may be expressed as an absolute amount of dry matter biomass (kg dry matter), carbon contained in biomass (kgC), energy equivalent of biomass (J). It is possible to assess HANPP in great spatial detail by combining statistical data with land-cover data derived from remote sensing (Haberl et al., 2001). In principle, HANPP could be linked consistently to the System of National Accounts (SNA), thus facilitating integrated economic-ecological models of pressures on biodiversity, but actually achieving this goal will require substantial improvements in methods.

HANPP is a measure of the human domination (Vitousek et al., 1997) or colonization (Fischer-Kowalski and Haberl, 1997) of ecosystems. HANPP indicates how intensively a defined area of land is being used in terms of flows of trophic energy in ecosystems (Haberl et al., 2004). HANPP is a measure of how strongly human use of a defined land area affects its primary productivity, and how much of the NPP is diverted to human uses and consequently is not available for processes within the ecosystem.

Trophic energy is one of the most important factors that determine patterns and processes in ecosystems. NPP is the sole energy input of all heterotroph food chains. Many aspects of ecosystem functioning, e.g., nutrient cycling, build-up of organic material in soils or in the aboveground compartment of ecosystems, vitally depend on this energy flow. HANPP demonstrates the impact of human activities on these important ecosystem processes, and thus also on ecosystems services such as carbon sequestration or buffering capacity. Theoretical considerations indicate that a sufficient amount of energy remaining in the ecosystem is necessary for ecosystems to be resilient (Kay et al., 1999). HANPP might impede ecosystem services and thus sustainability: “to the extent that (…) natural systems, species and populations provide goods or services that are essential to the sustainability of human systems, their shrunken base of operations must be a cause of concern" (Vitousek and Lubchenco, 1995, p. 60).

It is plausible that HANPP may be an important driver of biodiversity loss. The theoretical background behind this notion is the so-called species-energy hypothesis (Brown, 1981, Hutchinson, 1959, Wright, 1983) which holds that species numbers in ecosystems depend on the availability of trophic energy. If humans remove energy from ecosystems and lower NPP_t, species numbers would therefore be bound to decline (Wright, 1987, Wright, 1990). On an abstract level this seems obvious. Biomass is the mass of living or dead organisms present in a system. The very idea of trophic-dynamic process in ecosystems (Lindeman, 1942) is an abstract notion for organisms coming into being, growing, and dieing. This process is fuelled by various metabolic processes taking place within organisms. Energy enters organisms above all through two processes: photosynthesis and ingestion of dead or living organisms or parts thereof. Human-induced changes in this process affect patterns (including biodiversity), processes, functions, and services of ecosystems almost by definition.

4. Comparison of approaches

4.1 Resilience approach

The starting point of the resilience approach came from the natural science. The approach suggested a specific view on ecosystems (multiple equilibria, focus on change rather than on optimum, danger of irreversible change etc.). This created demand for new management strategies with an emphasis on learning and adaptive management which is able to cope with change. A deeper analysis of such management strategies was only possible through cooperation with social scientists. There existed already social science approaches that focus on societal adaptation and change such as policy learning and institutional analysis.

There are fairly strong incentives for researchers from both disciplines to cooperate with each other. In order to generate useful policy recommendations natural scientists have to co-operate with social scientists. For social scientists focusing on change in their analysis, the resilience approach delivers arguments for the relevance of their research.

Given that both approaches focus on the same management problem and are complementary there should be a good chance of preparing coherent integrated research proposals and receiving funding. Journals close to the resilience alliance such as "Ecology and Society" have a good reputation in the natural as well as the social science community and allow publication of natural and social science work.

Problems of asymmetric information may arise to a certain extent. Social scientists are unable to assess whether the view of the natural scientists about ecological problems (flips of systems, etc.) reflects an adequate approach for a general analysis of ecological systems. Natural scientists in turn do not know whether the approaches of social scientists they co-operate with represent state-of-the-art knowledge. The problems are mitigated, however, by the fact that there is a substantial body of debate about the resilience approach in ecology and about approaches to analyse policy learning and institutional change in the social sciences. We can reasonably assume that with more debate it is easier to identify good quality scientists and approaches.

The resilience approach seems to be a good framework to structure integrated research. Furthermore, the analysis of the natural sciences with its demand for adaptive management provides a good starting point for the social sciences and applying social science focusing on learning and institutional change seems to be a logical consequence of natural science analysis. However, it seems unlikely that the natural or the social sciences become an integral part of the other concept or that methods applied in one discipline are being transferred to the other.

4.2 MCA

The starting point for MCA is the requirement from society of how to evaluate decisions that affect society as well as nature (e.g. impact of land use change on a species and on rural employment). MCA aims to gather relevant knowledge from natural and social science and structure it in a way that it can be used for decision-making. Furthermore, it is important to note that it is a method that has been developed in a different context and applied to biodiversity conservation.

Incentives for researchers from both the natural and social sciences to participate in MCA are likely to be higher from funding and policy improvement than from publications. There may be some methodological advantages but most research will probably be an application of MCA to a certain biodiversity management problem. Such applied work is certainly policy relevant and may have good chances of funding from some (policy oriented) funding agencies. However, as the scientific novelty of applying an existing method is limited it is probably difficult to publish such type of research in journals with a high scientific reputation.

Regarding asymmetric information one should note that scientists from both disciplines are probably unable to assess each others quality. They may, however, see whether the partner from the other discipline has experience in MCA and relevant publications. World views may not play a big role, because if one accepts the choice of MCA, the work of conducting an MCA is rather technical.

4.3 Ecological-economic modelling

In ecological-economic modelling both natural and social scientists have the common aim of developing recommendations for biodiversity management. They both look at similar management issues and also apply similar methods such as e.g. mathematical optimisation. However, their knowledge is complementary.

This setting provides very strong incentives for researchers to co-operate or acquire knowledge from the other discipline. An integrated approach is not only beneficial for funding or improvement of policy, it also helps with publications (see e.g. the substantial number of recent papers in journals with a good reputation that explain how to integrate costs into conservation management (e.g. Ando et al. 1998, Naidoo et al. 2006).

There are some problems in terms of asymmetric information. Regarding quality control there is a certain danger that scientists from one discipline acquire selected knowledge from the other discipline and then apply it wrongly or believe that they have captured the full richness of the other discipline (Wätzold et al. 2006). However, because of the mutual long-term interest in co-operation one may assume that scientific discussion and review processes will overcome such problems. Furthermore, similar to the resilience approach the ecological respectively economic approach provide a certain worldview (compared to resilience approach rather static, believe in optimisation, etc.). Here again, the ecologists (respectively economists) are unable to assess whether the view of the economic (respectively ecological) system that their co-operators have is the appropriate one.

4.4. Political ecology

The starting point of political ecology comes from the social scientists which recognise that in order to improve their research they need to include natural science knowledge. There is, however, no similar approach in the natural science which fits to the questions and methods political ecology looks at and requires knowledge from the social sciences.

The incentives for researchers to co-operate differ between the natural and social sciences. Given the emphasis of political ecology on integrating natural science knowledge the incentives for social scientists to co-operate are high. Co-operation is beneficial for publication, funding and policy recommendations. This is different for the natural scientists. They may get no benefits in terms of publications and not too much in terms of funding. The only strong motive for co-operation may be an interest of the natural scientists in developing better recommendations for policy improvement.

There is asymmetric information regarding the quality of research. The fact that there are no similarities in questions and methods means that the two disciplines are rather apart and the problem of asymmetric information might be more severe than in the other approaches. In addition, natural scientists might be rather attracted by political analysis which corresponds to their worldviews than by analysis which is scientifically convincing as they are unable to assess the quality of research.

4.5 MEFA

The early ideas for MEFA come from scientists (like Robert Ayres) who are close to the school of thought of Ecological Economics which is, however, very diverse in itself. The MEFA approach has been then further developed by social scientists taking up ideas by ecologists like the species-energy hypothesis. In both disciplines the approach is conceptually very different from other approaches.

There are some incentives for researchers to cooperate. The approach has been successfully published in refereed journals which, however, are more highly regarded by social scientists than be ecologists (e.g. Ecological Economics, Land Use Policy) implying that incentives for cooperation are higher for social scientists than for natural scientists.

Again, there is asymmetric information with respect to the quality of research. Social scientists cannot judge to what extent approaches like the species-energy-hypothesis are valid. Similarly, natural scientists are unable to assess the quality of work of the social scientists as they are very remote from their own work.

5. Conclusions

All analysed approaches face the challenge of overcoming asymmetric information between different disciplines regarding the quality of research. There are, however, aspects that differ between approaches and that reduce the asymmetric information problem. One important aspect in that respect is that approaches which provide incentives for long-term co-operation probably have less quality problems because long-term co-operations allow better quality checks. Another aspect is that the more researchers are involved from both disciplines and the more incentives there are to participate in the integrated approach the more competition there is and the more quality checks there are.

What can scientists do to overcome the problem of asymmetric information? The first thing is to be aware of the problem of asymmetric information. It is as simple as that that once you are aware of a problem it already helps, this is the first step to take action. A second recommendation is to improve quality controls, e.g. if there is an integrated paper or project it should be reviewed by researchers from both disciplines. Another possibility is to stimulate open discussion among members from the other discipline who have varying backgrounds so that one can observe the debate and arguments.

Finally, one needs to emphasise the importance of overcoming the problem of asymmetric information and ensuring that integrated research is high quality research. Otherwise there is a risk that it comes to what economists call adverse selection and what in economics is summarised with the sentence “the bad apples drive out the good ones”. IIf the outcome of integrated research is low-quality research this may lead to a bad reputation of integrated research driving away interested “good” researchers from integrated research in order to avoid getting a bad reputation. Given that improving biodiversity management requires integrated research this certainly needs to be avoided.

References

Adger, W. N., T. A. Benjaminsen, K. Brown and H. Svarstad (2001): Advancing a political ecology of global environmental discourses. Development and Change. 32 (4): 681-715.

Armstrong, C.W. (2007): A note on the ecological-economic modelling of marine reserves in fisheries, Ecological Economics, 62, 242-250.

Ayres, R. U. and Simonis, U. E., (1994) Industrial Metabolism: Restructuring for Sustainable Development. Tokyo, New York, Paris, United Nations University Press.

Baumgärtner, S. (2004): Optimal Investment in Multi-Species Protection: Interacting Species and Ecosystem Health, EcoHealth 1, 101-110.

Benjaminsen, T.A., S.S. Dhillion, J.B. Aune (2006): A critical political ecology of soil fertility in the Malian cotton zone. Geoforum.

Benjaminsen, T.A. and H. Svarstad (forthcoming): Mushers or Moose? Political Ecology of a Norwegian Mountain Conflict. Submittet paper.

Berkes, F. and Folke, C. (1994): Linking Social and Ecological Systems for Resilience and Sustainability. Stockholm.

Bhaskar, R. (1975): A Realist Theory of Science. Leeds: Leeds Books.

Blaikie, P. (1985): The Political Economy of Soil Erosion in Developing Countries. New York: Longman Scientific and Technical.

Blaikie, P. and H. Brookfield (1987): Land Degradation and Society. London: Methuen.

Brown, J. H., (1981): Two Decades of Homage to Santa Rosalia: Toward a General Theory of Diversity. American Zoologist 21, 877-888.

Bryant, R.L. and S. Bailey (1997): Third World Political Ecology. New York: Routledge.

Clark, C.W. (1976): Mathematical bioeconomics: the optimal management of renewable resources. New York, Wiley.

Drechsler, M. (2005): Probabilistic approaches to scheduling reserve selection, Biological Conservation, 122, 253-262.

Drechsler, M., Grimm, V., Mysiak, J., Wätzold, F. (2007a): Differences and similarities between economic and ecological models for biodiversity conservation, Ecological Economics, 62/2, 232-241.

Drechsler, M., Johst, K., Wätzold, F., Westphal, M. (2006): Integrating Economic Costs into the Analysis of Flexible Conservation Management, Ecological Applications, 16, 1959-1966.

Drechsler, M., Wätzold, F., Johst, K., Bergmann, H., Settele, J. (2007b): A model-based approach for designing cost-effective compensation payments for conservation of endangered species in real landscapes, Biological Conservation, 140, 174-186.

Eurostat (2002): Material use in the European Union 1980-2000. Indicators and Analysis. Luxembourg, Eurostat, Office for Official Publications of the European Communities, prepared by Weisz, H., Fischer-Kowalski, M., Amann, C., Eisenmenger, N., Hubacek, K., and Krausmann, F.

Fairhead, J. and M. Leach (1996): Misreading the African Landscape: Society and Ecology in a Forest-Savanna Mosaic. Cambridge: Cambridge University Press.

Fischer-Kowalski, M., (1998): Society's Metabolism. The Intellectual History of Material Flow Analysis, Part I: 1860-1970. Journal of Industrial Ecology 2(1), 61-78.

Fischer-Kowalski, M. and Hüttler, W., (1998): Society's Metabolism. The Intellectual History of Material Flow Analysis, Part II: 1970-1998. Journal of Industrial Ecology 2(4), 107-137.

Fischer-Kowalski, M. and Haberl, H., (1997): Tons, Joules and Money: Modes of Production and their Sustainability Problems. Society and Natural Resources 10(1), 61-85.

Forsyth, T. (2003): Critical Political Ecology. The politics of environmental science. London and New York: Routhledge, Taylor & Francis Group.

Grimm, V., and C. Wissel. (1997): Babel, or the ecological stability discussions: An inventory and analysis of terminology and a guide for avoiding confusion. Oecologia 109:323-334.

Groeneveld, R. (2004): Biodiversity Conservation in Agricultural Landscapes: A Spatially Explicit Economic Analysis. PhD thesis. Wageningen University, Wageningen, The Netherlands.

Haberl, H., (2001a): The Energetic Metabolism of Societies, Part I: Accounting Concepts. Journal of Industrial Ecology 5(1), 11-33.

Haberl, H., (2001b): The Energetic Metabolism of Societies, Part II: Empirical Examples. Journal of Industrial Ecology 5(2), 71-88.

Haberl, H., Erb, K.-H., Krausmann, F., Loibl, W., Schulz, N. B., Weisz, H., (2001): Changes in Ecosystem Processes Induced by Land Use: Human Appropriation of Net Primary Production and Its Influence on Standing Crop in Austria. Global Biogeochemical Cycles 15(4), 929-942.

Haberl, H., Wackernagel, M., Krausmann, F., Erb, K.-H., Monfreda, C., (2004): Ecological footprints and human appropriation of net primary production: A comparison. Land Use Policy 21(3), 279-288.

Haberl, H., Weisz, H., Amann, C., Bondeau, A., Eisenmenger, N., Erb, K.-H., Fischer-Kowalski, M., Krausmann, F., (2006): The energetic metabolism of the EU-15 and the USA. Decadal energy input time-series with an emphasis on biomass. Journal of Industrial Ecology 10(4).

Holling, C.S. (1973): Resilience and Stability of Ecological Systems, Annual Review of Ecology and Systematics, 4: 1-23.

Hutchinson, G. E., (1959): Homage to Santa Rosalia, or why are there so many kinds of animals? The American Naturalist 93, 145-159.

IEA (2001): Energy Statistics of OECD Countries 1998-1999. Paris, International Energy Agency (IEA), Organisation for Economic Co-Operation and Development (OECD).

Janssen R., (1984):. Gevolgen van ruilverkaveling voor het landschap. 11: Beoordeling van de gevolgen van ruilverkaveling met behulp van multi criteria-analyse. Wageningen: Rijksinstituut voor onderzoek in de bos- en landschapsbouw “De Dorschkamp”.

Janssen R. en P. Nijkamp, (1998): Evaluatie van beleids- en evaluatiemethoden. in Hellendoorn J.C. et al., 1998. Gewikt en gewogen: vijfentwintig jaar ‘Beleidsanalyse’. Den Haag: SDU Uitgevers, pp. 209-224.

Johst, K., M. Drechsler, and F. Wätzold. (2002): An ecological-economic modelling procedure to design effective and efficient compensation payments for the protection of species. Ecological Economics 41:37-49.

Kay, J. J., Regier, H. A., Boyle, M., Francis, G., (1999): An Ecosystem Approach for Sustainability: Addressing the Challenge of Complexity. Futures 31(7), 721-742.

Krausmann, F. and Haberl, H., (2002): The process of industrialization from the perspective of energetic metabolism. Socioeconomic energy flows in Austria 1830-1995. Ecological Economics 41(2), 177-201.

Lindeman, R. L., (1942): The Trophic-Dynamic Aspect of Ecology. Ecology 23(4), 399-417.

Martinez-Alier, J., (1987): Ecological Economics. Energy, Environment and Society. Oxford, Blackwell.

Martinez-Alier, J. (2002): The Environmentalism of the Poor: A Study of Ecological Conflicts and Valuation. Northampton, MA: Elgar.

McCarty, J. (2005): First World political ecology: directions and challenges. Environment and Planning A 37: 953-958.

Neumann, R.P. (2005): Making Political Ecology. London: Hodder Arnold.

Paulson, S. and L. Gezon eds. (2004): Political Ecology Across Spaces, Scales and Social Groups. Rutgers University Press.

Paulson, S., L. Gezon and M. Watts (2004): Politics, Ecologies, Genealogies. In: Paulson, S. and L. Gezon eds. (2004): Political Ecology Across Spaces, Scales and Social Groups. Rutgers University Press.

Peet. R., and M. Watts (eds). (1996): Liberation Ecologies. Environment, development, social movement. London and New York: Routledge.

Pezzey, J. C. V., Roberts, C.M., Urdal, B.T. (2000): A simple bioeconomic model of a marine reserve, Ecological Economics, 33/1, 77-91.

Polasky, S., Solow, A. R. (2001): The value of information in reserve site selection. Biodiversity and Conservation, 10/7, 1051-1058.

Quaas, M., Baumgärtner, S., Becker, C., Frank, K., Müller, B. (2007): Uncertainty and sustainability in the management of rangelands, Ecological Economics, 62/2, 251-266.

Robbins, P. (2004): Political Ecology. A Critical Introduction. Malden MA, Oxford UK and Carlton Australia: Blackwell Publishing.

Rosin, R.T. (1993): The Tradition of Groundwater Irrigation in Northwestern India. Human Ecology 21(1):51-86.

Sala, O. E., Chapin III, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sannwald, E., Huennecke, L. F., Jackson, R. B., Kinzig, A., Leemans, R., Lodge, D. M., Mooney, H. A., Oesterheld, M., Poff, N. L., Sykes, M. T., Walker, B., Walker, M., Wall, D. H., (2000): Global biodiversity scenarios for the year 2100. Science 287, 1770-1774.

Sanchirico, J.N., Wilen, J.E. (1999): Bioeconomics of Spatial Exploitation in a Patchy Environment, Journal of Environmental Economics and Management, 37, 129-150.

Sankhayan, P.L., Gurunga, N., Sitaulab, B.K., Hofstada, O. (2003): Bio-economic modeling of land use and forest degradation at watershed level in Nepal. Agriculture, Ecosystems & Environment, 94/1, 105-116.

Schroeder, Richard A., Kevin St. Martin and Katherine E. Albert. (2006): Political ecology in North America: discovering the Third World within? Geoforum 37:163-168.

Settle, C., T. D. Crocker, and J. F. Shogren. (2002): On the joint determination of biological and economic systems. Ecological Economics 42:301-312.

Shogren, J. F., Parkhurst, G.M. and C. Settle (2003): Integrating economics and ecology to protect nature on private lands: models, methods and mindsets. Environmental Science and Policy 6:233-242.

Sieferle, R. P., (1997): Kulturelle Evolution des Gesellschaft-Natur-Verhältnisses. In: Fischer-Kowalski, M., Haberl, H., Hüttler, W., Payer, H., Schandl, H., Winiwarter, V., Zangerl-Weisz, H. (Eds.), Gesellschaftlicher Stoffwechsel und Kolonisierung von Natur. Ein Versuch in Sozialer Ökologie. Amsterdam, Gordon & Breach Fakultas, pp. 37-53.

Skonhoft, A, N. Yoccoz, Stenseth, N. (2002): Management of chamois (rupicapra rupicapra) moving between a protected core area and a hunting area. Ecological Applications 12:1199-1211

Starfield, A. M., K. A. Smith, and A. L. Bleloch. (1990): How to model it: problem solving for the computer age. McGraw-Hill, New York.

Stott, P. and Sian Sullivan (eds). (2000): Political ecology. Science, myth and power. London: Arnold.

Sullivan, S. (1998): People, plants and practice in drylands: socio-political and ecological dimensions of resource-use by Damara farmers in north-west Namibia. Unpublished Ph.D thesis, University College London.

Sullivan, S. (2000): Getting the science right, or introducing science in the first place? Local ‘facts’, global discourse – ‘desertification’ in north-west Namibia. In: Stott, P. and Sian Sullivan (eds). 2000. Political ecology. Science, myth and power. London: Arnold.

Svarstad, H. (2004): A global political ecology of bioprospecting. In: Paulson, S. & L. Gezon eds.: Political Ecology Across Spaces, Scales and Social Groups. Rutgers University Press.

UN (1997): Energy Statistics Yearbook 1995. New York, United Nations, Department for Economic and Social Information and Policy Analysis, Statistics Division.

Van Huylenbroek G., (1986): Evaluatie van ruilverkavelingen met behulp van multicriteria-analyse. doctoral dissertation. Gent: Rijksuniversiteit Gent, Faculteit van de Landbouwwetenschappen.

Vayda and Walters (1999): Against political ecology. Human Ecology 27(1): 167-79.

Vitousek, P. M. and Lubchenco, J., (1995): Limits to Sustainable Use of Resources: From Local Effects to Global Change. In: Munasinghe, M. and Shearer, W. (Eds.), Defining and Measuring Sustainability, The Biogeophysical Foundations. Washington, D.C., United Nations University and The World Bank, pp. 57-64.

Vitousek, P. M., Mooney, H. A., Lubchenco, J., Melillo, J. M., (1997): Human Domination of Earth`s Ecosystems. Science 277, 494-499.

Walker, B., C. S. Holling, S. R. Carpenter, and A. Kinzig. (2004): Resilience, adaptability and transformability in social-ecological systems. Ecology and Society 9: 5.[online] UFL: http://www.ecologyandsociety.org/vol9/iss2/art5

Walker, Peter A. (2005): Political ecology: where is the ecology? Progress in Human Geography 29 (1): 73-82.

Watts, MJ (2000): Political Ecology. In E. Sheppard and T. Barnes (eds): A Companion to Economic Geography. Malden MA: Blackwell Publishers.

Wätzold, F., and M. Drechsler. (2005): Spatially uniform versus spatially differentiated compensation payments for biodiversity-enhancing land-use measures. Environmental and Resource Economics 31:73-93.

Wätzold, F., Drechsler, M., Armstrong, C.W., Baumgärtner, S., Grimm, V., Huth, A., Perrings, C., Possingham, H.P., Shogren, J.F., Skonhoft, A., Verboom-Vasiljev, J., Wissel, C. (2006): Ecological-economic modeling for biodiversity management: Potential, pitfalls, prospects, in: Conservation Biology, 20/4, S. 1034-1041.

Weisz, H., Krausmann, F., Eisenmenger, N., Amann, C., Hubacek, K. (2005): Development of Material Use in the European Union 1970-2001. Material composition, cross-country comparison, and material flow indicators. Luxembourg, Eurostat, Office for Official Publications of the European Communities.

Wright, D. H., (1983): Species-energy theorie: An extension of the species-area theory. Oikos 41, 495-506.

Wright, D. H., (1987): Estimating human effects to global extinction. International Journal of Biometeorology 31(4), 293-299.

Wright, D. H., (1990): Human Impacts on the Energy Flow Through Natural Ecosystems, and Implications for Species Endangerment. Ambio 19(4), 189-194.

Zimmerer, K.S. and T.J. Bassett (eds.) (2003): Political Ecology. An Integrative Approach to Geography and Environment-Development Studies. New York/London: The Guilford Press.