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Resilience and Sustainable Development: an Ecological Inquiry

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Preface

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BEFP	Biodiversity-ecosystem functioning paradigm
CEP	Community ecology paradigm
cf.	confer
e.g.	exempli gratia = for example
Et. al.	et alii = and others
GNP	Gross national product
i.e.	id est = that is
IPCC	Intergovernmental Panel on Climate Change
L	Latitude
MEA	Millenium Ecosystem Assessment
P	Panarchy
Pr	Precariousness
R	Resistance
SES	Social-ecological system
SRU	Sachverständigenrat für Umeltfragen der Bundesregierung
$\mathbf{T_{sust}}$	Theory of Sustainable Development
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
WBGU	Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen
WCED	World Commission on Environment and Sustainable Development

1 Introduction

"What is the crisis?" Cabrera *et al.* (2008) recently asked referring to a survey conducted on current academic staff working at Cornell University, USA. The result is a list showing as many as 116 of "the world's most pressing problems", ranging from the shortage of clean water or the rise of fundamentalist religion to insufficient emphasis on sustainability in planning at all levels. Based on this study, Cabrera *et al.* (2008) aim to establish a consensus across academia as to the importance and feasibility of these world-wide issues.

To me, the expression "what is the crisis?" is striking for two reasons. First, there is an awareness among academics of a global crisis.[1] This feeling of being part of a "world risk society" (Beck 2007) gets scientific support by influential publications, such as the *Assessment Reports* published by the "Intergovernmental Panel on Climate Change", the *Development Reports* put forward by the "United Nations Development Programme" or the *Millenium Ecosystem Assessment* and the *Global Environmental Outlook* edited by the "United Nations Environment Programme", to name but a few.[2] "Science" tells us that in 2001 over 1 billion people lived in extreme poverty, i.e. survived on less than 1$ per day of income (UNDP 2003), 60% of nature's services essential for human well-being, including freshwater or fish resources, are being degraded or used unsustainably (UNEP 2005b) and average global surface warming amounts to 1.8 – 4.0° Celsius by the end of the 21st century (IPCC 2007). An intuitive sense of justice is sufficient in order to realize the coming of daunting and potentially overwhelming challenges.[3]

Second, there is an awareness that there is a world-wide crisis but we do not really know what it is about. There is a deep uncertainty about what the important issues are, what we should prioritize and why. Leading personalities in the national and the

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[1] Numerous authors within sustainability science point to the precarious character of the times we live in. For instance, Steffen *et al.* (2007) suggest that our present time should be termed "the Anthropocene", as humans and our societies have become a global geophysical force. Steffen *et al.* predict that "whatever unfolds, the next few decades will surely be a tipping point in the evolution of the Anthropocene" (Steffen *et al.* 2007: 614). Similarly from a Christian perspective, Fisher and van Utt (2007: 930) hold that "we do not have much time to change course and begin to adopt sustainable ways of living".

[2] Recently, the global financial crisis fostered this feeling.

[3] Following Safranski (2003), I believe that *globalism*, i.e. the normative idea of forming globalisation, is from a psychological or existential perspective essentially an excessive demand and overwhelming. Safranski (2003) notes that globalism overboards the individual person with new imperatives and appeals, which noone can stand in the long term. The super-ego Siegmund Freud had in mind, may be harmless compared to the super-ego that is implanted by the responsibilities for the global future. The repressing effects of globalism on the individual person in my view require particular consideration.

international discourse give contradicting recommendations. For instance, while former Secretary-General of the United Nations Kofi Annan considers climate change as the most important global challenge to humanity in the 21st century, Wolfgang Haber, professor emeritus for landscape ecology at the Technische Universität München in Germany, holds:

> "we are preoccupied with fighting climate change and loss of biodiversity; but these are minor problems we could adapt to, albeit painfully, and their solution will fail if we are caught in the interrelated traps of energy, food, and land scarcity. Land and soils, finite and irreproducible resources, are the key issues we have to devote our work to" (Haber 2007: 359).

In such an environment, attempts to achieve orientation in meeting the global challenges find themselves on a sticky wicket.

To me, the unassertive question "what is the crisis?" is interesting, as it hints at the mixed emotions with respect to the discussion on "sustainable development". On the one side, sustainable development is recognized the world over as a key challenge facing 21st century society (WBGU 2005; Komiyama & Takeuchi 2006; Clark 2007). Since the Brundlandt-report (WCED 1987) and subsequent international conferences in Rio de Janeiro and Johannesburg the idea of sustainability exhibits broad appeal in both the political and the scientific realm (Kopfmüller *et al.* 2001; UNEP 2002; Kates *et al.* 2005). The existence of a global, environmental and developmental crisis thus represents a wide consensus. On the other side however, there is a deep conceptual confusion about the meaning of the term "sustainable development" (Ott & Döring 2004). Since the early beginnings of research on sustainability issues, the discourse is characterized by the absence of a clear theoretical framework (Lélé 1991). There is a variety of rather arbitrary positions championed within the national and international discourse, ranging from achieving sustainability via technical progress to concepts of a regional subsistence economy (e.g. Brand & Jochum 2000; Parris & Kates 2003a). Also to the general public, sustainable development appears to be an arbitrary, ideological or illusory approach (Grunwald 2004b). It is thus entirely unclear what the critical challenges and the fundamental objectives of sustainable development are.[4] Put simply, the question “what is sustainable development?” (Kates *et al.* 2005) is far from being answered.

Yet the questions “what is the crisis?” and “what is sustainable development?” are as far I am concerned very important. It is absolutely essential to specify the primary objectives of sustainable development. Yet I do not think that a survey across academic staff, as in Cabrera *et al.* (2008), is the silver bullet necessary for achieving progress in this field. Rather, what is required is a critical re-evaluation of the theoretical foundations of sustainability science as well as theory building. Only the reflection on the theoretical basis allows for the formulation of the critical issues of sustainable development in a well-founded and oriented way and thus removing the arbitrariness within sustainability discourse.[5]

Such a well-founded and clear conception of sustainable development contributes to stimulating sober-minded and intellectually honest action. It avoids the widespread catastrophismus (Bechmann 2006), exaggeratedly pathos and apocalyptical tone (Spanier 2006).[6] It takes into account - as the German sociologist Ulrich Beck notes - that global sustainability issues, such as climate change, function in our modern and widely secular society as a source of meaning among politicians (Müller 2007) (and in my view also among scientists), yet refrains from extending the concept of sustainable development to a doctrine of salvation or an art of lifestyle.[7] It counters the truly frightening and overwhelming character of the most pressing global problems (Safranski 2003: 85ff) with a clear vision for orienting local, national and global policy.[8]

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[4] Owing to the use of the *three-pillar model of sustainability* (i.e. development in each the ecological, the economic and the social dimension), the meaning of the term “sustainable development” has been diluted. It is already hard to find anything that is not related to the term “sustainable” in the general public, e.g. the sustainable stimulation of the job market, the sustainable increase in tax revenue or the sustainable development of routes of transport (cf. Ott & Döring 2004: 36).

[5] With reference to Arnim Wiek (http://www.14aisdrc2008.com/), Ott and Döring (2006), Scholz *et al.* (2006), Cash *et al.* (2003) and Grunwald (2004a, 2004b), I suggest that the theoretical basis of sustainable development may refer to the following *fundamental questions of sustainability science*: what is the underlying normative basis (ethics)?; what empirical issues are addressed (subject realm)?; how are these issues investigated (methodology)?; what kind of knowledge is generated in sustainability (epistemology)?; how is sustainability science institutionalized (institutional theory)?; how to apply sustainability to specific policy fields and how to assess the progress towards sustainability (operationalization)?; and how to bridge the science-policy gap (implementation)?. For descriptions on theory building with respect to sustainable development confer section 5.1.4.1.

[6] In a recent article in the *Süddeutsche Zeitung*, Pàmies (2008) describes climate change as a poetic truth and a literary genre.

[7] This does not mean, however, that well-understood religion, sprituality and theology could not contribute to a sustainable way of living. Since several years, there is considerable collaboration between sustainability science and theology (Gardner 2003; cf. for ChristianityVogt 2004; Fisher & van Utt 2007).

[8] Certainly, the impact science can exert on policy is not linear (i.e. from identifying the problem via doing science to formulating policy); rather the process of influencing policy is messy, iterative and involves many

Such a specified comprehension of sustainable development is based on clear scientific concepts. Scientific investigation of the inherent logic, the underlying assumptions and the practical implications of these concepts is highly valued, as conceptual problem solving may be as relevant as empirical work in order to achieve progress in science in general (Laudan 1977), and in particular progress in potentially poly-paradigmatic disciplines, such as sustainability science (cf. section 2.2).

This thesis focuses on the concept of resilience. *Resilience* can provisionally be defined here as *the capacity of a system to resist disturbance and still maintain a specified state*. Resilience thus represents a measure for the "stability" of a system.[9] In an ecological interpretation, the concept of resilience expresses the societal desire to maintain, to protect or to preserve specific parts of nature[10] that we value for particular reasons (e.g. socio-cultural relevance, ecological importance or economic value). For instance, a coastal community aims to maintain the stock of coral reefs in order to secure the delivery of sea food products, raw materials or recreational possibilities (Moberg & Folke 1999) and "science" tries to grasp the mechanisms underlying the "stability" of coral reefs by means of the concept of resilience. Hence the expression "coral reef resilience" (e.g. Nyström 2006). Apparently, the concept of resilience can be applied to many types of ecosystems, such as boreal forests in Canada, semi-arid savannahs in Africa or shallow lakes in Sweden, and thus promises to be an appropriate tool for use within sustainability science.

The concept of resilience emerged in ecology in the 1960 – 1970s but has been adopted since then by numerous scientific disciplines, e.g. sociology, economics and political science (Folke 2006). During the 1990s, the concept gained high momentum and has been viewed as one of the most important research topics for sustainability science. For instance, Arrow *et al.* argued that economic activities are sustainable only if the life-support ecosystems upon which they depend are resilient" (Arrow *et al.* 1995: 521), while Perrings and Ansuategi held that "the relevant question is not whether economic growth has environmental consequences: it is whether those consequences threaten the resilience of the ecological systems on which economic

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players, different belief- and value-systems and powerful vested interests (Lawton 2007). For instance, from the perspective of general systems theory the political subsystem can only be "irritated" by the scientific subsystem of society (Luhmann 1990; Egner 2007). For enhancing the communication between science and policy, the mode of sustainability science ought to be essentially transdisciplinary.

[9] The term "stability" is a meta-concept that covers various stability concepts, such as resistance, elasticity or constancy (cf. section 4.1.3).

[10] Of course, the term "nature" is highly ambiguous and has been intepreted in many different ways (cf. section 2.3).

activities depend“ (Perrings & Ansuategi 2000: 19).[11] Today, the concept is one of the rising stars within ecology, sustainability science and environmental management (Janssen 2007).[12]

This thesis examines the concept of resilience and its relation to a specific conception of sustainable development. It corresponds to use-inspired basic research within sustainability science *sensu* Clark (2007) that is oriented at the concept of interdisciplinarity *sensu* Lélé and Norgaard (2005) and conducts conceptual research *sensu* Laudan (1977) with the aim to generate societal knowledge (confer for more details on the methodology of this thesis section 2.2). It aims to (i) clarify the “empirical status” of the concept of resilience (i.e. in which empirical situations the assumptions of the concept are valid?), (ii) identify and potentially solve some conceptual problems (e.g. how is resilience being defined?; how should resilience be defined?; how to measure resilience?; can resilience be applied to social or economic systems?) and (iii) describe its potential to nourish sustainability science (e.g. how can resilience be related to natural capital?; how can the management for resilience contribute to poverty eradication in developing countries?) (cf. for more details on the objectives of this thesis section 2.1). This thesis does not represent a further empirical study on the resilience of a particular region (as e.g. in Gunderson & Holling 2002; Gunderson & Pritchard 2002). Rather, it tries to examine the overall theoretical and practical relevance of the concept of resilience for sustainability science. This thesis investigates the scientific possibilities offered by the concept of resilience for meeting some of the challenges arising from the question “what is the crisis?”.

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[11] Yet the concept of resilience is only one possibility to specify the ecological dimension of sustainable development. “Resilience” supersedes previous scientific concepts used at the foremost in the 1980s – 1990s, such as ecosystem integrity (Lemons *et al.* 1997; Rapport 1999) and ecosystem health (Rapport 1989; McShane 2004). Also, it complements alternative concepts used nowadays, such as biodiversity (van der Maarel 1997; UNEP 2005c), ecosystem services (UNEP 2005b; Rodriguez *et al.* 2006) or vulnerability (Schröter *et al.* 2005; Adger 2006). The concept of resilience thus has to be understood in the context of these previous and complementing concepts. Yet it still can be viewed as an “environmental innovation” within sustainability science (confer section 4.1.4).

[12] The recent award of the *Volvo Environment Prize 2008* for C. S. Holling, the “inventor” of the resilience approach, illustrates the relevance that is given to resilience research within ecological science.

2 Preliminaries

This chapter delineates the preliminaries of this thesis. This includes explanations on objectives (section 2.1), methods (section 2.2), terminology (section 2.3), basic assumptions (section 2.4) and overview of the thesis (section 2.5).

2.1 Objectives

This thesis entitled “Resilience and Sustainable Development: an Ecological Inquiry” examines the concept of ecological resilience and its relation to a specific conception of sustainable development. The expression “Ecological Inquiry” in the title appear to be confusing, as the term “ecological” is ambiguous (Trepl 1987: 13ff; 2005: 14ff). Broadly speaking, the term “ecology” refers to (*a*) the biological subdiscipline ecology, (*b*) interdisciplinary research on environmental issues (e.g. sustainability science), or (*c*) to a world view or a political direction (i.e. the “green movement”). Here the expression “Ecological Inquiry” indicates two issues. First, this thesis focuses on the ecological dimension of the concept of resilience. That means, I focus on the concept of *ecological* resilience that is applied at the foremost to ecological systems, rather than on concepts, such as social resilience, economic resilience or livelihood resilience that refer to other types of systems, as my expertise is largely restricted to this dimension.[13]

It does not however mean that this thesis is restricted to *ecological science*.[14] Rather, this thesis follows the second view of “ecology”. The subject realm (‘*Gegenstandsbereich*’) of this thesis is the concept of ecological resilience, yet the methods stem from a variety of scientific disciplines, e.g. ecological science, science & technology studies, philosophy of science or ecological economics. Thus second, the expression “Ecological Inquiry” refers to the interdisciplinary research on environmental issues.[15]

The objectives of this thesis are three-fold.

First, I aim to *describe and analyze the fundamental theoretical framework of resilience research*. According to specific approaches within the philosophy of science, each scientific theory – thus also the “resilience theory” – is formulated within a theoretical framework. This framework (*a*) is comprised of conventional

[13] I did not chose “Ecological Resilience and Sustainable Development: an Inquiry”, as this thesis is not restricted solely to the concept of ecological resilience. I also treat the concept of social-ecological resilience, as it represents the prevalent concept in resilience research (Brand & Jax 2007), and the concept of social resilience and economic resilience as regards to the poverty reduction measures in developing countries.

[14] Nor does it mean that I follow a specific ecological theory, be it individualistic (e.g. Trepl 2005) or holistic (e.g. Odum 1999). This thesis does not presuppose a specific ecological theory, rather it uses the different ecological theories for understanding the concept of resilience. It is beyond my competence to reflect on the issues of ecological resilience from the perspective of solely one ecological theory. This may be a certain weakness of this thesis.

[15] In my view, even though this form of research is oriented at the concept of “interdisciplinarity” (Lele & Norgaard 2005), this thesis is still based on my competence that stems largely from biology and ecology.

assumptions on the general constitution of the study objects, which are not empirically justified; (*b*) determines what kind of questions are (not) being posed and in what form these questions are (not) to be answered; and (*c*) that is not in itself prone to falsification, but even to the contrary is protected against falsification. Such a framework is termed “hard core of a research program” by Lakatos (1970) or “styles of scientific thinking” by Crombie (1994).

The analysis of the hard core of resilience theory includes the examination of the empirical validity of the concept in terms of applicability and power of explanation. That means, I consider in what empirical cases resilience theory is valid. The first objective of this thesis thus refers to the following research questions: what does the term “resilience” mean?; what are the fundamental presuppositions of the concept of resilience?; in what empirical cases do the presuppositions and the concept hold?; how can the concept be operationalized?; and what is the relation of the concept to other important concepts, such as “biodiversity” or “disturbance”? The issue of operationalization is especially important for analyzing the concept of resilience and I put a special emphasis on this issue in this thesis. The first objective of this thesis is thus to describe and analyze the hard core of the research program of resilience theory.

Second, based on the first objective I aim to *understand, critizice and question the hard core of resilience theory*. This reflection on the styles of scientific thinking of resilience theory includes several conceptual and theoretical issues. (i) I treat the issue of *linguistic uncertainty* in terms of ambiguity, indeterminacy of theoretical terms, underspecificity and context dependence (Regan *et al.* 2002). The aim is to clarify the meaning, to make decisions about future usage of terms and specify the context of specific concepts. This includes analyzing the ambiguous use of the term “resilience” in fundamentally distinct contexts. (ii) The scientific thought-styles of scientific theories can be understood by their cultural background, i.e. the philosophical or cultural underpinning that is – often unconsciously - being presupposed. That is, I examine the *cultural background*, the “cultural apriori” (Eisel 2004) of resilience theory. (iii) In the relevant literature, resilience theory is extended from ecological systems to other types of systems, such as social, economic or technical systems. I investigate the relation of the concept of resilience to “general systems theory” in order to understand this extension.

Thus, the second objective of this thesis is to answer the following questions: how to understand and handle the ambiguous use of the term “resilience” in entirely different contexts, from the capacity to absorb shocks and still maintain function in ecology (Nyström 2006), via the transition probability between states as a function of the consumption and production activities of decision makers in economics (Brock *et al.* 2002) to a surrogate for well-being in sustainability science (Marschke & Berkes 2006)? Why does the resilience approach assume that the relation of ecological systems and social systems corresponds to a coevolution? Why does the resilience approach assume that social or economic systems follow the same rules as ecological systems? Thus, the second objective of this thesis is to understand, critizice and question the hard core of resilience theory.

Third, it is my aim to *investigate the relevance of resilience theory for achieving sustainable development*. The goal of achieving sustainable development is viewed as a touchstone for the practical relevance of the concept of resilience. The focus is on the operationalization (i.e. indication, estimation and measurement) of resilience and the approach of environmental management for resilience (i.e. adaptive co-management). More precisely, I examine whether the concept of resilience can be applied to the concept of critical natural capital and whether the concept can contribute to fighting poverty in developing countries.

What is the reason for my choosing these topics? The reason for the former concept (i.e. critical natural capital) is that I advocate a conception of “strong sustainability” or at least “intermediate sustainability”. In such a conception, the maintenance of natural capital is an important objective for achieving sustainable development (this is not necessarily the case if a conception of weak sustainability is championed). The concept of critical natural capital becomes especially relevant in the face of intensive population growth and global change, as this term signifies that part of the natural capital that is to be preserved for present and future generations.

The reason for the latter topic (i.e. fighting poverty in developing countries) is that this tenet is a central objective of many if not all sustainability conceptions. If the concept of ecological resilience contributes in tackling this sustainability problem, it will be of high practical relevance for sustainability science.

Thus, the third objective of this thesis is to investigate the relevance of resilience theory for achieving sustainable development. This thesis corresponds to use-inspired basic research (Clark 2007).

Altogether it has become clear that the main objectives of this thesis are: (i) describing and analyzing the fundamental theoretical framework of resilience theory; (ii) understanding, critizicing and questioning the theoretical framework of resilience theory; and (iii) investigating the relevance of resilience theory for achieving sustainable development.

2.2 Methods

This thesis represents a study on the relation between the concept of resilience and a specific conception of sustainable development. Both concepts refer to ecological but also to social, economic, institutional and political domains and are therefore employed by a variety of disciplines. In order to investigate these concepts as well as the resilience-sustainability nexus, it is inappropriate to focus on one sole discipline such as ecological science. Rather, an adequate investigation is geared to the concept of *interdisciplinarity*, understood as all types of crossings between and among disciplines (Lele & Norgaard 2005).[16] Still my competence stems largely from biology and ecology, but this thesis nonetheless attempts to integrate insights from other scientific disciplines as well, such as ecological economics, sustainability science, philosophy of science, sociology or political science.

It is thus more appropriate to subsum this thesis under "social-ecological research" or "sustainability science". According to Becker (2003), *social-ecological research* ('*Sozial-ökologische Forschung*') is the science of the relationships between humans and their specific natural and societal environment. Social-ecological research investigates the forms of these relationships from an interdisciplinary perspective. The aim is to generate knowledge in favour of societal activities for securing society's future reproductivity, opportunities for action and the natural livelihoods (Becker 2003). Similarly, *sustainability science* ('*Nachhaltigkeitsforschung*') represents a problem-oriented science that regulates the man-nature-relationship of society, where facts are uncertain, values in dispute, stakes high and decisions urgent (Bechmann & Frederichs 1996; Nölting *et al.* 2004). This is what Clark (2007) terms *use-inspired basic research*. Correspondingly, this thesis investigates the concepts of resilience and sustainability as well as the resilience-sustainability nexus in order to generate knowledge for societal action. It is essentially use-inspired basic research.

No biological-empirical research is carried out in this thesis and it is therefore a genuine theoretical study. Yet the term "theoretical" is used ambiguously in ecology. Cooper (1998) distinguishes between two different senses of "theory": theory as qua mathematical modelling and theory as an expression of the most fundamental

[16] The concept of interdisciplinarity is extended by the concept of *transdisciplinarity* that requires crossing boundaries both horizontally (across disciplines; as interdisciplinarity) and vertically (across experts, policymakers, practitioners and the public) (e.g. Klein 2004; Hirsch-Hadorn *et al.* 2006; Scholz *et al.* 2006).

principles which structure the phenomena. This thesis does not carry out ecological modelling but focuses on the latter meaning of theory and conducts *conceptual work*.

In contrast to physics, most theories in scientific disciplines such as biology or ecology are based not on laws but on concepts (Mayr 2007: 28). Mayr (2000: 97) holds that most of the recent progress in more complex biological sciences, e.g. ecology, behavioural biology and evolutionary biology, is due to the development of new concepts. In a similar vein, Trepl (2005: 13) notes that progress in "diffuse disciplines" such as ecology is achieved mainly by the clarification of basic terms and concepts, while Paine (2002) considers conceptual evolution to be the driving force in the progress of ecological science.[17] Also, Pickett *et al.* (1994: 57ff and 85ff) argue that conceptual refinement and exactitude is of high relevance for theory maturation. This may be even more true for social-ecological research or sustainability science also employed in this thesis. The message of this is that it might stand to reason that - besides empirical work - conceptual work is an important tool for achieving scientific progress.

The importance of conceptual work for attaining scientific progress is dependent on the specific position one advocates within the philosophy of science. The philosophies of each Popper, Kuhn and Lakatos stress empirical work as the most relevant factor (Poser 2001: 166). On the contrary, the *theory of scientific growth* proposed by Laudan (1977) argues that conceptual work is at least as important in the development of science as empirical problem solving (Laudan 1977: 45).[18] According to Laudan (1977: 66) any theory about the nature of science which finds no role for conceptual problems forfeits any claim to being a theory on how science has actually evolved. Conceptual work solves *conceptual problems* of theories. Laudan (1977) distinguishes between internal conceptual problems and external conceptual problems. *Internal conceptual problems* arise if a theory exhibits certain internal inconsistencies or when its basic categories of analysis are vague and unclear. *External conceptual problems* emerge if a theory is in conflict with another theory or doctrine which proponents believe to be rationally well founded. This thesis follows the philosophy of science proposed by Laudan (1977). It focuses on internal

[17] A good example for this conceptal progress in ecology is the conceptual clarification of the long-standing diversity-stability-debate by Pimm (1984).

[18] The aim of science is then to maximize the scope of solved empirical problems, while minimizing the scope of anomalous and conceptual problems (Laudan 1977: 66).

conceptual problems of resilience theory (cf. chapter 3 and section 4.1) but some external conceptual problems are also treated (cf. section 3.5.2.3 and section 4.2).
Altogether it has become clear that this thesis corresponds to use-inspired basic research within sustainability science *sensu* Clark (2007) that is oriented around the concept of interdisciplinarity *sensu* Lélé and Norgaard (2005) and conducts conceptual research *sensu* Laudan (1977) with the aim to generate societal knowledge.

What types of methods are being applied in this thesis?
First, I have conducted literature research on the variety of topics treated in this thesis using the Web of Science ISI Databases and some further databases managed by the *Resilience Alliance* (http://www.resalliance.org) or the *Forum on Science and Innovation for Sustainable Development* (http://sustsci.aaas.org) In some cases this method results in a (mini-)review of a specific topic, as in section 5.3 on resilience and poverty reduction in developing countries.
Second, I used the framework of Regan *et al.* (2002) to solve and clarify *linguistic uncertainties*. Linguistic uncertainty arises "because much of our natural language, including a great deal of our scientific vocabulary, is underspecific, ambiguous, vague, context dependent, or exhibits theoretical indeterminacies" (Regan *et al.* 2002: 618). According to Regan *et al*. (2002), linguistic uncertainty can be classified into five distinct types. Those are: (i) *vagueness*, i.e. the occurrence of borderline cases in scientific language; (ii) *context dependence*, i.e. the failure to specify the context in which a proposition is to be understood; (iii) *ambiguity*, i.e. a word can have more than one meaning and it is not clear which meaning is intended; (iv) *underspecificity*, i.e. unwanted generality; and (v) *indeterminacy of theoretical terms*, i.e. the potential for ambiguity, as the future usage of theoretical terms is not completely fixed by past usage. Regan *et al.* (2002) suggest that vagueness should be counteracted by sharp delineation, context dependence by specifying the context, ambiguity by clarifying the meaning, underspecificity by providing the narrowest bounds and intedeterminacy in theoretical terms by making decisions about future usage of term when need arises. I have used these strategies to manage linguistic uncertainty whenever appropriate. This is a means to solve internal conceptual problems *sensu* Laudan (1977).

Third, I have used the criteria for the usefulness of concepts, suggested by (Jax 2002: 14). Those are: (i) the meaning of the concept must be definite; (ii) the concept must be consistent; (iii) the concept must be communicable and intersubjectively comprehensible; (iv) the premises of the concept must be well-founded; they should be stated clearly; (v) the concept must contribute to the generation of knowledge about the object of investigation; and (vi) it must be possible to operationalize the concept. The basic terms of this thesis are scrutinized by means of these criteria.

Fourth, I have made use of several fundamental insights taken from ecological science. These include classifications of *stability concepts* proposed by Pimm (1984), Grimm and Wissel (1997) and Hansson and Helgesson (2003) (cf. section 4.1.3), the *ecological situation* proposed by Grimm and Wissel (1997) (cf. section 3.3.3.2) and the methodology of *self-identity* put forward by Jax *et al.* (1998) and Jax (2006) (cf. section 3.4.2.1.2).

Fifth, we[19] have taken the concept of a boundary object used within *Science & Technology Studies* and have applied it to the term “resilience” (cf. section 4.1). A *boundary object* signifies a term that facilitates communication across disciplinary borders by creating shared vocabulary although the parties’ understanding would differ regarding the precise meaning of the term in question (Star & Griesemer 1989), similar to biodiversity (Eser 2002) or sustainability (Brand & Jax 2007).

Sixth, we[20] have used a *cultural position within philosophy of science*. This position has been unfolded in Kirchhoff (2007). Kirchhoff (2007) holds the view that scientific theories have cultural underpinnings, which implies that the “context of justification” of scientific theories is dependent on their “context of discovery”. This cultural background of scientific theories can be made explicit by the identification of structural analogies of cultural/ philosophical theories and scientific theories. We have applied this cultural position to the resilience approach in section 4.2.

[19] This method is the result of the collaboration with Prof. Dr. Kurt Jax, Leipzig and München, in our recent article (Brand & Jax 2007).

[20] This method is the result of the collaboration with Dr. Thomas Kirchhoff, München, and Deborah Hoheisel, München.

2.3 Terminology

Most terms in ecology require quotation marks. They are slippery because there are various comprehensions, definitions or measures for each of them. Yet imprecise definitions of terms impede scientific development in ecology as well as the ability to apply ecological knowledge to relevant fields of society (Jax *et al.* 1992). As Jax (2008: 178) notes, “the problem is not which words are used, but rather what meaning is attached to them, the concepts behind those words, and the intricacies of those concepts”. Jax continues that “there is a lack of understanding of the major epistemic differences of current and historical concepts that travel under the label of particular terms”. Thus, basic terms should be defined as explicitly and precisely as possible (Jax *et al.* 1992).

With respect to the fundamental terms in ecology (especially population, community and ecosystem) a dual approach may be appropriate (Jax 2006). *Generic meanings* of the main concepts should be used only as heuristically useful perspectives, while *specific and operational definitions* of the concepts should be developed, depending on the specific purpose of the study. While more specific and operational definitions of terms and concepts are given in the course of this thesis, this section defines the more frequently used basic and generic terms.

Before I will turn to the specific definitions, some words on the term “definition” as such are in order. Definitions are crucial for every serious discipline (Jax *et al.* 1992; Belnap 1993; Pickett *et al.* 1994). A *definition* is “a statement, declaration or proposal (the “definiens”) establishing the meaning of an expression (the “definiendum”) (Craig 1998: 845f). It is generally distinguished between three types of definitions (Belnap 1993).[21] Those are: (i) *dictionary* or *lexical definitions*, i.e. explaining the existing meaning of an old word, that is, a word already in use in the scientific community but unfamiliar to the person wanting the explanation; (ii) *stipulative definitions*, i.e. explaining a proposed meaning for a new word; and (iii) *analyses or explications*, i.e. relying on an old, existing meaning and attaching a new, proposed meaning by the means of discussing its basic assumptions, implications and theoretical underpinnings. The latter type of definition will be used for several fundamental terms examined within this thesis, such as “ecological resilience”, “disturbance” or

[21] There are different classifications of the types of definitions in the relevant literature. For instance, Audi (1995: 185f) distinguishes between as many as 14 types of definitions.

“sustainability”. In the following I will use lexical definitions and stipulative definitions in order to establish a sort of conceptual fundament for this thesis. To be sure: each of the subsequent lexical definitions is contentious, yet I chose those definitions that are provided by an salient authority in the specific field and that possess the highest degree of conceptual clarity and brevity.

In this thesis, an *ecological unit* will be understood as “all those units that are subject to ecological research and comprise more than one single organism” (Jax 2006: 239). *Units* are defined as “aggregations of objects (particulars), which are chosen and arranged according to such criteria that they can be characterized as new relevant objects of their own” (Jax 2006: 239).

In a generic meaning, *population* signifies “a group of individual organisms of the same species in space and time” (Jax 2006: 240); *community* means “an assemblage of organisms of different types (species, life forms) in space and time” (Jax 2006: 240); while *ecosystem* characterizes “an assemblage of organisms of different types (species, life forms) together with their abiotic environment in space and time” (Jax 2006: 240).

The term “function” is used ambiguously in ecology. At least, four major uses of the term *function* in ecology can be distinguished (Jax 2005). Those are: (i) functions denote what happens between two objects. In this case, the term “function” is used in a similar fashion to terms such as “ecosystem process”, “interaction”, “pathway” or “mechanism”; (ii) function in the sense of the functioning of the whole ecosystem, i.e. how the whole is sustained; (iii) function in the sense of a role within the system, as used for instance in the concept of functional groups or functional types; and finally (iv) functions in terms of practical use, as used in the concept of ecosystem services. In this thesis, I use the terms “ecosystem process”, “functioning of the whole system”, “functional role” and “ecosystem service” to express the four meanings, or I at least make explicit, in which specific meaning the term is used.

Biodiversity (or “biological diversity”) is generally defined as the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (UN 1992; UNEP 2005c). The concept of biodiversity refers to the diversity on the genetic (e.g. population diversity), species (e.g. number of species) and ecosystem level (e.g. heterogeneity).

Ecosystem services can be broadly defined as "the benefits that people obtain from ecosystems" (Bennett *et al.* 2005b).

The term "nature" can be defined following Aristoteles as "everything that achieves form and changes by itself" (own translation, from Trepl *et al.* 2005: 685). The term has been used highly ambiguously for a variety of meanings (Mutschler 2002; Trepl *et al.* 2005).

The concept "landscape" is a meta-category and can be interpreted in many different ways, from the landscape as "an ecosystem" to the landscape as homeland. For a detailed treatment of the ambiguous concept "landscape" confer Haber (2001) and the articles in Kazal *et al.* (2006).

Based on the concept of ecological resilience (confer section 3.1), a research program has been established, which I dub *resilience approach* or *resilience research* in this thesis. The resilience approach is being put forward by the *Resilience Alliance* (http://www.resalliance.org), a research organization comprised of scientists and practitioners from many disciplines who collaborate to explore the dynamics of coupled social-ecological systems[22]. The Resilience Alliance edits an international, peer-reviewed journal termed *Ecology & Society*, which is available on the internet (http://www.ecologyandsociety.org). Key documents of resilience research are listed in Janssen *et al.* (2006) and Janssen (2007).

[22] For the term "social-ecological system" confer section 3.5.2.2.

2.4 Basic Assumptions

This thesis is based on several basic assumptions and is therefore to some degree tentative. This tentative character is due to the fact that any ecological theory must be based on certain terms, concepts and assumptions. These concepts and assumptions are in almost every case contentious, because ecology represents a diffuse and poly-paradigmatic discipline (Trepl 2005: 13ff) characterized by several competing theories on almost any subject (Kirchhoff 2007), and owing to the vague status of ecology within philosophy of science, as considered below. For instance, the concept of functional groups (cf. section 3.3.1.2.2) or the source-sink paradigm on the relation of biodiversity and “ecological stability” (cf. section 3.3.1.2.5) are controversial, yet this thesis assumes their validity or at least usefulness. It almost goes without saying that for the purposes of transparency the contentious character of any controversial concept is made explicit in the course of this thesis.

In addition, this thesis is tentative because there exist fundamental discussions in ecology that cannot be decided upon in this thesis. For example, the issue of ontological realism with respect to ecological units, i.e. the question whether or not a community or an ecosystem is real in an ontological sense, is highly contentious, as both perspectives (the affirmative and the negative) are plausible (Trepl 1988). Also, different comprehensions of ecological units (e.g. individualistic, organic) are championed in ecology and it is elusive to decide upon their validity (Kirchhoff 2007; Voigt 2008). In these rare cases, this thesis refrains from choosing one particular position. The following paragraphs describe some of the basic assumptions made in this thesis.

The first assumption refers to the issue of ontological realism with respect to ecological units mentioned above. Jax *et al.* (1998) distinguish between two fundamental positions. Those are: (a) an *epistemological view* towards the perception of ecological units, i.e. ecological units are viewed as abstractions which an observer creates for specific purposes or to answer certain questions by selecting patterns and processes from the whole of nature; and (b) an *ontological view*, i.e. ecological units are seen as something that is given as such in nature and which therefore have to be found and identified instead of being defined and delimited. This distinction refers to the question whether or not an ecological unit, such as an ecosystem, exists ontologically. Trepl (1988) considers both views as justified and doubts that one view will become dominant or prevailing.[23] In contrast, Jax (2006) holds that a scientific study that applies concepts, such as community or ecosystem, is only possible on the basis of the epistemological view of ecological units. He states that the ontological approach leads to metaphysical speculations, is not apt to aid research and that there is no way for us to know any supraorganismal entity of nature in its reality and totality. This thesis refrains from advocating a specific position. My metaphysical reservation is being complicated by the fact that resilience research advocates an ontological view with respect to ecological units (cf. section 4.2.3.1). Yet I assume that it is *methodologically* useful to examine ecological systems at the ecosystem level and to thus use the term “ecosystem”, but neither do I advocate nor refuse an ontological realism with respect to ecological units in this thesis.

The second basic assumption of this thesis refers to the different system comprehensions existent in the ecological literature. It is about the query of how ecological units, such as communities or ecosystems, are structured and how the elements of an ecological unit interact. For many decades, there has been an ongoing controversy between rather holistic and rather individualistic positions (Trepl 1987; Kirchhoff 2007; Voigt 2008).[24] Put crudely, the holistic system comprehensions assume that ecological units behave similar to an organisms, so that the components of an ecological unit interact strongly and are mutually dependent on each other, whereas the individualistic notions hold that ecological units are not “compact entities” and that the components are not tightly linked to one another. The resilience approach champions a holistic, organic position with respect to ecological units (cf. section 4.2.3.1). Yet this thesis does not advocate a specific position - neither holistic nor individualistic; rather I use the different systems notions to elucidate the underlying presuppositions of the concept of ecological resilience.

The third basic assumption of this thesis refers to the concept of emergent properties. Breckling *et al.* (2005) state that *emergent properties* in ecological systems can be defined by three main characteristics. These are: (1) they do not exist on the level of isolated subsystems; (2) they emerge on higher levels as a result of interactions between the subsystems; and (3) new properties appear at one level of a system and

[23] Trepl (1988) suggests that the issue of the influence societal conditions have on the perception of nature in general and ecosystems in particular is scientifically more interesting than the long-standing debate whether ecosystems are ontologically real or not (cf. also Kirchhoff 2007; Voigt 2008).

[24] This picture of holistic positions on the one side and individualistic positions on the other side is oversimplifying. The holistic and individualistic notions are extreme positions of a continuum and there are many hybrid forms (Kirchhoff 2007; Voigt 2008).

are not deducible from the observation of the lower levels units or compartments.[25] Based on this notion of emergent properties, Reuter *et al.* (2005) distinguish between emergent properties and collective properties. *Collective properties* are properties on higher levels, which result from a simple arithmetic operation regarding lower level entities. No new quality on higher levels is involved, they are only extended quantitatively. Examples include the total weight of a population. In contrast, emergent properties, as stated above, include all properties that cannot be linearly reduced to properties on lower levels or are newly defined on a particular level. According to Reuter *et al.* (2005), emergent properties can further be subdivided into aggregational properties and connective properties. *Aggregational properties* can be specified from the perspective of the observer. They primarily denote statistical properties, e.g. the spatial distribution or the spatial structure of a population. *Connective properties* are based on the interactions of organisms or their independently modelled components. These properties necessarily depend on interactions and feedback processes between the represented organisms and between these organisms and their environment. Examples include space exploitation by plant species, connectedness within a metacommunity or interpatch fluxes (Reuter *et al.* 2005; Lidicker 2008). Within this classification, ecological resilience represents an emergent and connective property of ecological systems resulting from the interaction of many different properties and mechanisms occurring at lower levels of organization (cf. section 3.3.3.3). This thesis assumes that the concept of emergent properties is at least methodologically useful.

The fourth basic assumption refers to recent insights gained within the philosophy of science in ecology. First, it is questionable in how far statements made in ecology can be generalized. This query has stimulated a debate on whether or not there are ecological laws (Lawton 1999; Hansson 2003; Lange 2005). Among the popular candidates for ecological lawhood have been the Malthusian “law” of exponential growth, the allometries of macroecology, the rules of stoichometry, the generalization that ecological succession will occur if an open site becomes available, the competitive exclusion principle, the impossibility of a population’s growth without bound and the species-area relationship (Lange 2005). Apparently, this lawhood depends on the particular conception of law that is used, ranging from “generalized formulation based on a series of events or processes observed to recur regularly

[25] Note that there are alternative comprehensions of emergent properties in ecological systems (e.g. Lidicker 2008).

under certain conditions; a widely observed tendency" (Lawton 1999: 177) to "p is an ecological law only if p would still have been reliable under any counterfactual circumstances that is logically consistent with p together with the (other) ecological laws" (Lange 2005: 400). If the latter conception of law is used, few of the relationships mentioned above will achieve the status of a law.[26]

Indeed, many authors hold that ecological phenomena are too complex and locally variable, temporally and spatially, to be covered by general ecological laws (McIntosh 1987; Peters 1991; Hansson 2003).[27] Regarding statements on "ecological stability", Grimm and Wissel (1997) argue that stability statements are only valid for a particular ecological situation, characterized by e.g. the variables considered, the level of decription, the type of disturbance regime or scale (cf. also section 3.3.1.1). This thesis takes a careful and tentative position with respect to this debate. Any statement on ecological resilience is at first only valid for a particular ecological situation and it must be meticulously assessed whether or not the statement can be generalized across other ecological situations.

The second insight from the philosophy of science in ecology is related to the issue of causation. According to Holling (1992b), there are two scientific cultures in ecology. The first analytical stream is essentially experimental, reductionist and disciplinary, whereas the second integrative stream is interdisciplinary and combines historical, comparative and experimental approaches at scales appropriate to the issue. This thesis focusses on the latter integrative "culture". The important point for this section is that this scientific stream focusses not on single causation but on *multiple causation*. It is about the assumption that within ecology the search for a single necessary cause frequently fails and that instead causes in ecological systems are multiple and overlapping and are not entirely separable (Hilborn & Stearns 1982; Holling 1992b; Pickett *et al.* 1994; Holling & Allen 2002). The message of this is that this thesis assumes that it is not useful to search for the single cause for the emergence of ecological resilience, but rather that a variety of causes must be identified (cf. section 3.3.3).

[26] This interesting debate is widely unsettled, but it appears that ecological laws would look quite different than physical laws, for instance (Lange 2005).

[27] According to Kingsland (1995), a tension between a theoretical pluralism that values historical case studies, on the one hand, and an aspiration to build general, unifying models, on the other hand, permeates the history of ecology.

2.5 Overview of the Thesis

This section offers an overview of this thesis. The main chapters of this thesis are: description of the concept of resilience (chapter 3), theoretical reflections on the concept of resilience (chapter 4), the relation of the concept of resilience to sustainable development (chapter 5) and the conclusion (chapter 6).

Chapter 3 unfolds the "theory" on the concept of resilience. This theory can be located on different layers, which include descriptions on the term (section 3.1), the basic assumptions of the theory, i.e. its background theory (section 3.2), the resilience mechanisms (section 3.3), the operationalization, i.e. estimation, indication or measurement (section 3.4) and the management approach (section 3.5). The five layers together build what I dub the "anatomy of ecological resilience". The fundamental question of chapter 3 is: what does resilience theory mean in the relevant literature? Chapter 3 thus aims to *describe* the scientific the hard core (Lakatos 1970) of resilience theory.

Chapter 3 examines the concept of resilience *within* a paradigmatic perspective of theory building. The concept is examined and critiziced from an *internal* perspective only. That means, I only use empirical critique in terms of the power of explanation and applicability of resilience theory.[28] At first, chapter 3 does not question the fundamental theoretical framework of resilience research. This is the topic of chapter 4, in particular section 4.2. Yet the internal critique includes questioning the specific assumptions of the concept, such as the notion of system dynamics or the theory on alternative stable states. For instance, I evaluate the theory on alternative stable states with respect to its empirical applicability, i.e. for which empirical situations this theory is valid. This is also the delimiting criterion and rationale for for the whole chapter 3: the use of internal, empirical critique. Chapter 3 thus aims to *analyze* the hard core of resilience research.

In section 3.1 at first I provisionally define the term "resilience", and in particular the term "ecological resilience", as the latter concept will be the central topic of this thesis.

Subsequently, section 3.2 describes and critizices the background theory, i.e. the underlying assumptions of the concept of ecological resilience. The concept of

[28] When I use external critique in chapter 3 this is only for complementary purposes.

ecological resilience presupposes several related concepts, namely the concepts of adaptive cycle and panarchy, the concept of key variables and the concept of alternative stable states. Section 3.2.1 investigates a notion for the structure and dynamics of ecosystems (termed “adaptive cycle” and “panarchy”) and shows that this notion represents a meta-model for ecosystem dynamics among others that fails to be universally applicable. Section 3.2.2 examines the concept of key variables and the ‘rule of hand’. I argue that this concept corresponds to a top-down approach to ecological modelling. Such an approach is although partial, as it competes with bottom-up approaches to ecological modelling, e.g. individual-based models. Finally, section 3.2.3 explores the theory on alternative stable states and ecological thresholds. My findings suggest that the theory on alternative stable states is applicable to many ecosystem types but not all. Based on a typology for ecological thresholds, I suggest that there are some interesting methods for predicting the location of ecological thresholds that have not been of much impact in the relevant literature.

Section 3.3 investigates the concept of resilience mechanisms. I examine which properties and mechanisms are conceived to be conducive for the ecological resilience of specific ecosystems. When the issue of stability mechanisms is at stake, at least two fundamental debates within ecological science are, in my view, relevant: the diversity-stability-discussion and the biodiversity-ecosystem functioning-debate. The study of the relevant literature on *resilience* mechanisms suggests, however, that this literature is widely detached from the two other relevant debates within ecological science. Thus, in a first step section 3.3.1 delineates the state-of-the-art on the discussion on the diversity-stability-discussion, the biodiversity-ecosystem functioning research, the hypothesis on ecological “stability” that have been proposed in the relevant literature and the critics of biodiversity-ecosystem functioning research. Section 3.3.1 concludes with recommendations for future studies and suggests a “conceptual volume” that integrates the lessons learned and the tools gained from the debates mentioned above (section 3.3.1.6). In a second step, section 3.3.2 applies these lessons and tools to the discussion on resilience mechanisms. It turns out that the “ability” of ecosystems to show ecological resilience is dependent on a variety of resilience-conducive properties and mechanisms. Finally, section 3.3.3 concludes with a list of resilience-conducive properties and mechanisms and with some further remarks. In sum, section 3.3 aims to unify the state-of-the-art in

research on 'biodiversity and ecosystem functioning' with the work on resilience mechanisms.
Subsequently, section 3.4 describes and evaluates the methods to operationalize, i.e. to indicate, estimate or measure the concept of ecological resilience. After clarifying the term "operationalization" in section 3.4.1, section 3.4.2 presents a comprehensive procedure for conducting a resilience analysis. A resilience analysis at first requires descriptions on what exactly should be resilient (the of-what part in section 3.4.2.1) against which disturbances (the to-what part in section 3.4.2.2). Based on these descriptions, section 3.4.2.3 proposes several methods for the empirical indication of ecological resilience. Hereby, section 3.4.2.3.1 describes an approach that I term "threshold approach", which assumes several concepts taken from the background theory, such as the concept of key variables, the theory on alternative stable states or the concept of ecological thresholds; section 3.4.2.3.2 applies the methodology of self-identity proposed by Jax *et al.* (1998) and Jax (2006) to the field of ecological resilience (what I coin "self-identity approach"); and section 3.4.2.3.3 uses the concept of resilience mechanisms as a measure of ecological resilience (dubbed "resilience mechanisms approach"). Subsequently, section 3.4.2.4 explores the possibilities to set up disturbance scenarios for generalizing statements about the degree of ecological resilience of a specific ecosystem.
Finally, section 3.5 describes the environmental management approach that is being proposed within resilience research, namely the *adaptive co-management approach*. At first, section 3.5.1 presents a short sketch of the conventional resource management. This conventional resource management approach (termed *command-and-control management*) is viewed as the antithesis to the adaptive co-management approach proposed within resilience research. Subsequently, section 3.5.2 examines the characteristics of the adaptive co-management approach. After delineating the basics of adaptive management and co-management in section 3.5.2.1, section 3.5.2.2 analyzes its focus on social-ecological systems; section 3.5.2.3 hints to the widespread extension of ecological theory to other types of systems and the relation between resilience research and general systems theory, and section 3.5.2.4 explores the concrete management measures that are being proposed in the relevant literature.

Chapter 4 offers several theoretical reflections on the concept of resilience. The aim of this chapter is not to evaluate whether and in how far the concept applies to empirical reality or not. This is the objective of chapter 3. Rather, chapter 4 aims to understand, critizice and question the hard core of resilience theory. More specifically, I examine the concept of resilience with respect to its conceptual clarity and linguistic (un-)certainty (section 4.1) and the cultural and philosophical background of the concept (section 4.2).

Section 4.1 examines the conceptual development of resilience. After some introductory words in section 4.1.1 , section 4.1.2 offers a typology of the variety of definitions of resilience proposed in the relevant literature. The results show that there are at least ten types of definitions for the concept of resilience that are characterized by a different degree of normativity. Subsequently, section 4.1.3 focusses on the descriptive definition of ecological resilience suggested in ecological science, while section 4.1.4 shows in how far the concept of resilience has been adopted by other scientific disciplines. We[29] argue that resilience has become a “boundary object”, i.e. a term that facilitates understanding between different scientific disciplines or social groups and point to some merits and threats of boundary objects. Finally, in section 4.1.5 we list some conceptual difficulties and linguistic uncertainties of the concept of resilience that are related to the use as a boundary object. The results of section 4.1 have been published in the international peer-reviewed journal *Ecology and Society* (Brand & Jax 2007).

Section 4.2 examines the cultural bias of the resilience approach. This section thus questions the “hard core of the research program” (Lakatos 1970). After some introductory words in section 4.2.1, section 4.2.2 jusitifies our[30] method used for the analysis within the philosophy of science. We hold the view that scientific theories have cultural underpinnings, which implies that the “context of justification” of scientific theories is dependent on their “context of discovery”. This cultural background of scientific theories can be made explicit by the identification of structural analogies of cultural/ philosophical theories and scientific theories. Correspondingly, section 4.2.3 shows the structural analogy of the organic, holistic systems notion within resilience research and the monadology proposed by G. W. Leibniz, while section 4.2.4 elucidates the analogy of the man-nature relationship

[29] This section is the result of the cooperation with my supervisor Kurt Jax.

[30] This section is the result of the cooperation with two colleagues working at the chair of landscape ecology at the Technische Universität München, namely Thomas Kirchhoff and Deborah Hoheisel.

within the adaptive co-management approach as proposed within resilience research, and the counter-enlightenment, cultural theory of J. G. Herder. In section 4.2.5 we conclude that the resilience approach can be interpreted as a scientific, utilitarian reformulation of a conservative theory on the man-nature-relationship. The resilience approach is partial, as it is dependent on specific cultural theories, and cannot claim universal validity. Some of the results of section 4.2 have been handed in to the journal *Ecology and Society* but the article is still in the review process (Kirchhoff *et al.* in review).

Chapter 5 examines the relation between the resilience approach and the discussion on sustainable development. The objective of this chapter is thus to explore the practical relevance of resilience research. As the scope of the discourse on sustainable development is very wide, it is beyond this thesis to examine all the relations and connections between resilience research and sustainable development. Rather, I focus on two central topics within sustainability science: (i) the concept of critical natural capital and (ii) poverty reduction measures in developing countries. I consider these two issues as highly relevant for achieving sustainable development, and they may be regarded as a touchstone for the practical relevance of resilience research. At first, I clarify the concept of sustainable development (section 5.1), before I examine the relation between the concept of ecological resilience and the concept of critical natural capital (section 5.2) as well as poverty eradication (section 5.3).

Section 5.1 clarifies the understanding of the concept of sustainable development. The terms “sustainability” and “sustainable development” are highly unclear and lack a profound theoretical framework. After the introduction in section 5.1.1, section 5.1.2 pictures the international and the German discourse on sustainability. We[31] propose that the relevant literature suggest that there is a need for a comprehensive theoretical foundation for sustainable development, what we term a “Theory of Sustainable Development”. Section 5.1.3 provides several reasons why we do need such a theory after all. Subsequently, section 5.1.4 delineates two of the salient approaches to set up such a theory, namely the Greifswald-approach and the HGF (‘*Helmholtz-Gemeinschaft Deutscher Forschungszentren*’)-approach. We conclude

[31] This section is the result of the cooperation with the following colleagues: Julia Schultz (Greifswald, Berlin), Konrad Ott (Greifswald) and Jürgen Kopfmüller (Karlsruhe).

with the findings in section 5.1.5. Section 5.1 has been published in the *International Journal of Environment and Sustainable Development* (Schultz *et al.* 2008).
Section 5.2 examines the relation of the concept of ecological resilience and the concept of critical natural capital. After the introduction in section 5.2.1, section 5.2.2 delineates the concept of ecological resilience. Section 5.2.3 revisits the concept of critical natural capital and formulates an own approach to “criticality”. The criticality of natural capital is a multi-facet concept that is comprised of cultural, economic and ecological components, for instance. Subsequently, section 5.2.4 investigates the relation of ecological resilience and critical natural capital. Critical natural capital is conceptionalized by two components: importance and degree of threat. I propose that the degree of ecological resilience of a specific ecosystem can be a measure of the degree of threat of specific parts of the natural capital. Section 5.2.5 concludes with the findings and with some thoughts about their relevance for ecological economics and sustainability science. Section 5.2 has been published in the journal *Ecological Economics* (Brand 2009).
Section 5.3 examines the possibilities the concept of ecological resilience offers for poverty reduction measures in developing countries. After some introductory words in section 5.3.1, section 5.3.2 pictures the environment-poverty nexus. Subsequently, section 5.3.3 provides a mini-review of the literature on the relation between resilience research and poverty eradication in developing countries, before section 5.3.4 gives some recommendations for future work on the resilience-poverty link. Section 5.3.5 concludes with the findings of this section.

Finally, chapter 6 presents the conclusions of this thesis and a tentative prospect on research on resilience and sustainable development.

3 The Anatomy of Ecological Resilience

This chapter investigates the fundamental theoretical framework of the resilience approach. Such a theoretical framework has been termed the “hard core of a research program” by Lakatos (1970). This chapter thus describes the hard core of resilience research. The analysis of the hard core of resilience theory includes the examination of the empirical validity of the concept in terms of applicability and power of explanation. That means, I consider in what empirical cases resilience theory is valid.

This theory[32] on resilience is nothing but extensive. Various definitions have been formulated, mechanisms proposed and concepts debated (Anderies *et al.* 2006; Folke 2006; Walker *et al.* 2006; Brand & Jax 2007). It is therefore useful to distinguish between several levels where the discussion on “resilience” takes place (confer *Table 1*).

Table 1: Anatomy of ecological resilience

No. Level	Description of Level	No. Section
1	*Term*	3.1
2	*Background Theory*	3.2
3	*Resilience Mechanisms*	3.3
4	*Operationalization*	3.4
5	*Environmental Management Approach*	3.5

Thus, in what follows this chapter traces out the *anatomy of ecosystem resilience*, which is comprised of five levels.[33] Those are: term (section 3.1), background theory (section 3.2), resilience mechanisms (section 3.3), operationalization (section 3.4) and finally environmental management approach (section 3.5). Higher levels of the anatomy do not determine lower levels entirely, rather they give some orientation.

[32] The term „theory“ can be defined as a “system of conceptual constructs” (Pickett *et al.* 1994: 57). In such a sense of the word, the resilience approach can be dubbed a theory.
[33] I borrow the term “anatomy” from Smith *et al.* (2000).

For example, if “resilience” is interpreted as descriptive stability concept in ecological science at level 1, then level 2 - 4 will also be based in ecological science. If, however, “resilience” is viewed as an approach to address whole social-ecological systems at level 1, then level 2 - 4 will comprise insights of other sciences, such as economy, sociology or geography. Level 5 generally reflects the societal context of conservation or resource management efforts and is interdisciplinary by definition. Yet the *object* of management of level 5 is dependent on the specification of higher levels. If at level 1 an ecological-descriptive concept is favoured, the ecological dimension will be strong at level 5 whereas this dimension potentially is about to vanish if level 1 is filled with an interdisciplinary concept.

3.1 Term

Conceptual confusion on the term “resilience” is immense. The term “resilience” is interpreted in divergent ways by numerous scientific disciplines, e.g. ecology, economics, sociology, political science and engineering sciences (Folke 2006; Brand & Jax 2007). The levels and aspects of the meaning of the term “resilience” are numerous and require a sophisticated analysis. Yet the understanding of these different levels and aspects requires to grasp many of the concepts introduced in the subsequent sections. Therefore, this section merely sketches a couple of basic distinctions as regards to the term “resilience” and aims to provisionally clarify the meaning of the central subject of this thesis: the term “ecological resilience”. For a more comprehensive analysis of the term “ecological resilience” confer also section 4.1.

First of all, two basically distinct meanings of “resilience” must be distinguished. The first one refers to dynamics close to equilibrium[34] and is defined as the time required for a system to return to an equilibrium point following a disturbance event (Pimm 1984). It has been coined “engineering resilience” (Holling 1996) and is largely identical to the stability property “elasticity” (Grimm & Wissel 1997).[35] The second meaning of resilience refers to dynamics far from any equilibrium steady state and is defined as “the capacity of a system to experience shocks while retaining essentially the same function, structure, feedbacks, and therefore identity” (Walker *et al.* 2006: 2). This has been termed “ecological resilience” and is comprised of three characteristics: (1) the amount of change a system can undergo and still remain within the same basin of attraction[36], i. e. to retain the same controls on structure and processes; (2) the degree to which the system is capable of self-organization; and (3) the degree to which the system expresses capacity for learning and adaptation (Carpenter *et al.* 2001; Walker *et al.* 2002; Folke 2006). It is this second meaning of “resilience” that is the primary subject of this thesis.[37]

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[34] The term “equilibrium” is highly ambiguous. For specifying the concept of ecological resilience I will not use “equilibrium” but rather the terms “state” or “regime”. Confer for a detailed treatment of the term “equilibrium” Weil (1999).

[35] This first meaning corresponds to the etymological meaning of resilience, which was formed in English on the model of Latin *resilientem*, which means “to rebound” or “to recoil”.

[36] The resilience approach assumes that ecosystems can exhibit “alternative stable states” or “alternative basins of attraction”. For more information confer section 3.2.3.

[37] In the relevant literature, the term “ecological resilience” is applied almost interchangeable with the words “ecosystem resilience” (Gunderson & Holling 2002) or “resilience” (Holling 1973; Arrow *et al.* 1995; Perrings

The intension of the term "ecological resilience" has also been interpreted in numerous ways (Folke 2006; Brand & Jax 2007). What is of particular relevance for the subsequent sections is to distinguish a genuine descriptive concept of ecological resilience applied to ecological systems, i.e. *ecological resilience*, from a much broader and vaguer concept of resilience that is used as a perspective to analyze coupled social-ecological systems, i.e. *social-ecological resilience* (cf. section 4.1) (Brand & Jax 2007). In this chapter I will mainly focus on the concept of ecological resilience that is applied to ecological systems only.

I define *ecological resilience* here as *the capacity of an ecosystem to resist disturbance and still maintain a specified state.* I consider this definition to be superior to the one proposed by Walker *et al.* (2006) quoted above for two reasons. First, it avoids ambiguous and unclear terms as "function" (Jax 2005) or "identity" (Jax *et al.* 1998).[38] Second, it does not presuppose the existence of alternative stable states. This presupposition may be indeed problematic (confer section 3.2.3).

My definition may have the following limitations. First, it may underestimate other important characteristics of ecological resilience, such as the capacities for renewal, reorganization and development, as interpreted in Folke (2006). Yet the capacities for renewal, reorganization and development represent, in my view, additional descriptive statements, rather than defining criteria of ecological resilience (Jax 2007). That is, the capacities may empirically be linked to the concept of ecological resilience, but this must at first be proven empirically. The link of the concept of ecological resilience to these other aspects should not be taken for granted. Rather this link should be the subject of empirical research.

Second, my definition presupposes at least three other highly ambiguous concepts that have to be clarified, i.e. "ecosystem", "disturbance" and "state". The term "ecosystem" can be defined in a generic meaning as "an assemblage of organisms of different types (species, life forms) together with their abiotic environment in space and time" (Jax 2006: 240) (cf. section 2.3). Yet the term must be further specified for concrete ecological situations (Jax 2007). In addition, the resilience approach advocates an ontological realism with respect to ecological units. That is, resilience scholars assume that "ecosystems" do exist ontologically, that ecosystems can be

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et al. 1995; Carpenter & Cottingham 1997; Carpenter *et al.* 2001; Folke *et al.* 2004; Walker *et al.* 2004; Carpenter & Folke 2006). In the following I will use the expression "ecological resilience" for this meaning.

[38] My definition may thus be interpreted as a *precising definition*, as it intends to reduce the vagueness of previous defnitions (Audi 1995: 186).

found and identified out there in nature (Jax *et al.* 1998). In addition, resilience research champions a holistic and organic systems notion with respect to ecosystems, i.e. ecosystems are considered to behave similar to organisms (cf. section 4.2.3.1). This thesis refrains from advocating a specific position. I assume that it is *methodologically* useful to examine ecological systems at the ecosystem level. Yet I advocate neither an ontological realism with respect to ecological units nor an specific systems notion, be it individualistic or holistic (cf. section 2.4).

In ecology, the concept of *disturbance* can be understood in two fundamentally different ways (White & Jentsch 2001). The *relative definition* of disturbance seeks to define disturbances as *causing deviation from the "normal" dynamics of an ecosystem* (White & Jentsch 2001). In contrast, the *absolute definition* of disturbance is based on any physical and measurable changes in variables or in the disposal of resources, whether or not these changes are recurrent, expected or normal. The resilience approach employs a relative definition of disturbance and I follow their approach in this thesis (confer for a detailed treatment of the topic 'ecological resilience and disturbance' section 3.3.2.3.1).

The term "state" may be disadvantageous, as it has static connotations. Yet in fact, populations, communities and ecosystems are always changing and fluctuations in abundances and distributions are the rule. I still use "state" in my definition, because of its wide-spread use and acceptance.

In this thesis, the term "ecological resilience" thus signifies a descriptive stability concept for ecological systems that is based on other fundamental concepts in ecology, such as "ecosystem", "disturbance" or "state". The following sections describe the background theory of this very concept.

3.2 Background Theory

This section explores the background theory of the resilience approach. At first section 3.2.1 delineates the comprehension of ecosystem structure and dynamics termed “adaptive cycle” and “panarchy”, before section 3.2.2 describes the concept of key variables and the “rule of hand”. Subsequently, section 3.2.3 examines the theory on alternative stable states, which includes the concept of ecological thresholds.

3.2.1 Ecosystem Structure and Dynamics

The first critical assumption of "resilience theory" is the proposition of a metamodel[39] (Cumming & Collier 2005) of ecosystem dynamics, the so-called "adaptive cycle" and "Panarchy". These metamodels imply a specific approach to the understanding of complexity in ecological systems. Holling (2001) distinguishes between two alternative approaches to complexity. *View 1* sees complexity as anything we do not understand, because there are apparently a large number of interacting elements. The appropriate scientific method is to embrace the complexity and resulting uncertainty and analyze different subsets of interactions, each of which seem relevant from a number of different operational or scientific perspectives. *View 2* suggests that the complexity of living systems of people and nature emerges not from a random association of a large number of interacting factors, but rather from a small number of controlling processes or variables. These variables establish a persistent template upon which a host of other variables exercise their influence. Such "subsidiary variables" or factors can be interesting, relevant, and important for scientific inquiries, but they exist at the whim of the critical controlling processes and variables (Holling 2001). *View 2* corresponds to the resilience approach to complexity and is outlined in several key publications (Holling 2001; Gunderson & Holling 2002).

It is beyond the scope of this thesis to describe the whole discussion as regards to the different approaches to complexity. Rather, in the following I will explore the position of the resilience approach. This section draws heavily on an earlier publication (Brand 2005: 48ff) and represents merely an updated version of this work. However, I must repeat some of the work here because this descriptions are necessary for understanding further aspects of ecological resilience in the subsequent sections of this thesis.

The overall goal of this approach to complexity proposed by resilience research is to rationalize the interplay between change and persistence, between the predictable and the unpredictable processes occurring in complex systems. The resilience approach suggests that there is a level of simplicity behind the complexity that, if identified, can lead to an understanding of the structure and dynamic of ecosystems.

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[39] Cumming and Collier specify the term *metamodel* as „a broader class of model that encapsulates the key dynamics of numerous other models" (Cumming & Collier 2005: 6).

It might be possible to identify the points at which a system is capable of accepting desirable change and the points where it is vulnerable (Holling 2001).

3.2.1.1 Adaptive Cycle

The concept of "adaptive cycle" is based on three requirements (Holling & Gunderson 2002). Those are: (1) the system must be productive, i.e. must acquire resources and accumulate them; (2) there must be some sort of shifting balance between stabilizing and destabilizing forces reflecting the degree and intensity of internal controls and the degree of influence of external variability (i.e. the disturbance regime); and (3) the resilience of the system must be a dynamic and changing quantity that generates and sustains both options and novelty, providing a shifting balance between vulnerability and persistence *sensu* Grimm and Wissel (1997).

Gunderson and Holling (2002) suggest that a framework for "adaptive change" ("adaptive" in the sense of reacting to changing conditions) exhibits generality if it contains three ecosystem properties shaping the future responses: (1) the *potential for change*, which determines the range of options possible, (2) the *degree of connectedness* between internal controlling variables and processes, i.e. a measure that reflects the degree of flexibility or rigidity of such controls, and (3) the *ecological resilience* of the system. As Holling and Gunderson (2002) point out, these three properties together shape a dynamic of change:

> "Potential sets limits to what is possible – it determines the number of the alternative options for the future. Connectedness determines the degree to which a system can control its own destiny (…). Resilience determines how vulnerable the system is to unexpected disturbances and surprises that can exceed or break that control" (Holling & Gunderson 2002: 51).

These properties are not static characteristics but exhibit dynamic behaviour, i.e. they expand and contract as ecosystem succession proceeds. They show a regular behaviour on a patch scale[40] and pass through four distinct phases, which together

[40] The adaptive cycle refers to the local or patch scale in the first place. In this thesis the *patch scale* corresponds to areas from 1 – 10 km^2 and is defined as locality capable of holding a local and clearly specified community (Leibold *et al.* 2004). Section 3.2.1.2 explores the "Panarchy" that includes adaptive cycles at multiple scales and larger scales.

constitute the *adaptive cycle* (Holling 2001; Gunderson & Holling 2002). This general model of systemic change is illustrated in *Figure 1*.

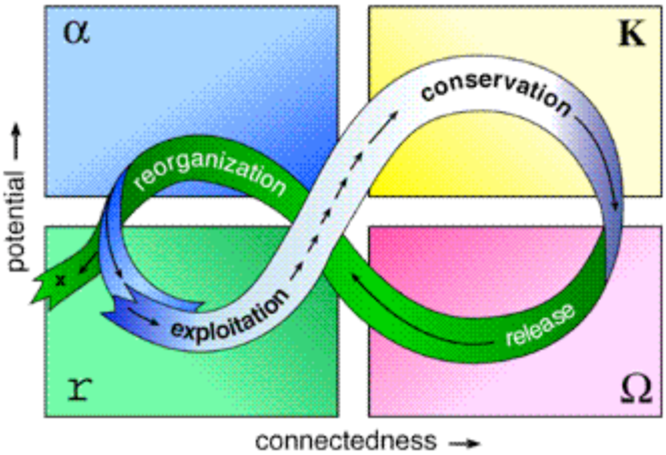

Figure 1: Adaptive cycle: the heuristic of ecosystem dynamics for two ecosystem characteristics: potential and connectedness
from URL: http://www.resalliance.org, resource library

The *traditional view* pictured ecosystem succession as being controlled by two functions: *exploitation*, in which rapid colonisation of recently disturbed areas is emphasised and *conservation*, in which slow accumulation and storage of energy and material is emphasised (Begon *et al.* 2006: 486). The species in the exploitation phase have been characterized as r-strategists and those in the conservation phase as K-strategists.[41] For this reason, the exploitation phase of the adaptive cycle is dubbed *r phase*, while the conservation phase is termed *K phase* (Holling 2001).
According to Holling and Gunderson (2002), two additional functions are needed in order to properly understand ecosystem dynamics. The first revision is that of *release*, or *creative destruction* - the *omega* (Ω) *phase*. The tightly bound accumulation of biomass and nutrients becomes increasingly fragile or over-connected until suddenly released by agents of disturbance such as forest fires, drought, insect pests or intense pulse of grazing. The second additional function is one of *reorganization* – the *alpha* (α) *phase* – in which soil processes minimize

[41] The names *r-* and *K-strategists* are drawn from the traditional designation of parameters of the logistic equation (r represents the instantaneous rate of growth of a population, and K the sustained plateau or maximum population that is attained) (Krebs 2001).

nutrient loss and reorganize nutrients in such a manner that they become available for the next phase of exploitation. Part of this reorganization involves the transient appearance or expansion of organisms that begin to capture opportunity, i.e. the pioneer species.[42]

If the omega phase represents the end, then it is immediately followed by the alpha phase, the beginning, to complete the cycle. During this cycle, biological time flows unevenly. The progression in the ecosystem cycle proceeds from the exploitation (r) phase slowly to the conservation (K) phase, very rapidly to release (Ω) phase, rapidly to reorganization (α) phase and rapidly back to exploitation (Holling *et al.* 1995; Holling & Gunderson 2002).

As illustrated in *Figure 1*, the flow of a system through the adaptive cycle is plotted as changes in two properties, i.e. connectedness and potential, which expand and contract as proceeding the cycle. During the slow sequence from exploitation to conservation (K phase), connectedness and constancy increase and a “capital” of nutrients and biomass is slowly accumulated (Holling *et al.* 1995). Competitive processes lead to a few species becoming dominant, with diversity retained in residual pockets preserved in a patchy landscape.

While the accumulated capital is sequestered for the growing, maturing ecosystem, it also represents a gradual increase in potential for other kinds of ecosystem states and futures (Holling & Gunderson 2002). Hence, connectedness and potential are low in r phase and high in K phase. As the progression to the K phase proceeds, the accumulating nutrient and biomass resources become more and more tightly bound within existing vegetation, preventing other competitors from utilizing them. That means, the system’s connectedness increases, eventually to become over-connected and increasingly rigid in its control.

The actual change in ecosystem dynamics is triggered by agents of disturbance such as wind, fire, disease, insect outbreak and drought or a combination of these. The resources sequestered in vegetation and soil are then suddenly released and the tight organization is lost (Holling & Gunderson 2002). Hence, potential is low and connectedness contracts within Ω phase, whereas potential is relatively higher and connectedness is low in α phase. As the system shifts from α to r phase, some of the potential leaks away because of the collapse of organization. Some of the accumulated resources literally leave the system (Holling & Gunderson 2002). In

[42] Pioneer species are generally conceived as being typical for the exploitative phase, yet the resilience approach holds that pioneer species are already relevant in the reorganization phase.

addition, new entrants, those that survived to the α phase, and the *biotic legacies* [43] of past cycles (Franklin & MacMahon 2000; Elmqvist *et al.* 2002) begin to sequester and organize resources in a process that leads to the r species establishing "founding rights" over the remaining capital (Holling & Gunderson 2002). Hence, the potential and connectedness is low but expands during r phase until both are high in K phase to complete the cycle.

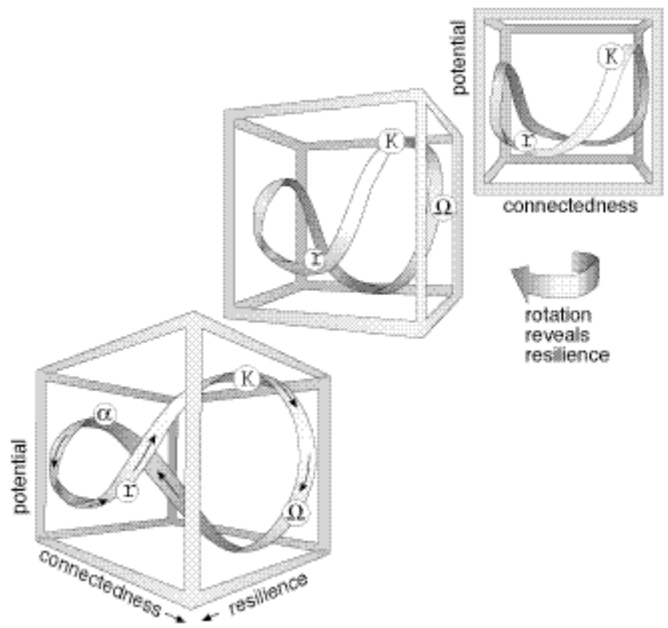

Figure 2: Three-dimensional adaptive cycle for three ecosystem characteristics: potential, connectedness and ecological resilience
from URL: http://www.resalliance.org, resource library

Yet there is another dimension that is included in an adaptive cycle. *Figure 2* adds the third dimension of an adaptive cycle, namely ecological resilience. It is recognized that conditions are needed that occasionally foster novelty and experiment. Those become possible during periods when connectedness is low and ecological resilience is high. The low connectedness permits novel re-assortments of elements that previously were tightly connected to one another. The high ecological

[43] *Biotic legacies* (Franklin & MacMahon 2000) comprise organisms that survive a disturbance event as well as biological structures that serve as foci for regeneration and allow species to colonize (e.g. tree stumps after fire) and represents a component of the concept of *ecological memory* discussed in section 3.3.2.2.

resilience allows tests of those novel combinations because system-wide costs of failure are low (Holling & Gunderson 2002).

How does the adaptive cycle proceed in terms of ecological resilience? During the α phase ecological resilience is high since there is a wide basin of attraction with weak regulation around the attractor, low connectedness among variables, and high potential for future development. It is a welcoming environment for experiments, for the appearance and initial establishment of species that otherwise would be out-competed. The α phase is the phase where novel re-assortments of species in ecosystems generate new possibilities that are later tested (Holling & Gunderson 2002). Ecological resilience remains high in the r phase since the pioneer species are adapted to high variability of microclimate and extremes of soil conditions.

During K phase the abundance of K-strategists and the connectedness among them increases resulting in conditions that are more predictable. Engineering resilience (i.e. elasticity) is high since small changes in regime configuration are removed quickly whereas ecological resilience contracts as the system becomes more vulnerable to surprise. In forests, for instance, fuel for fires and food for insect defoliators reach critical levels as processes that inhibit fire propagation and insect population growth (e.g. avian predation) are homogenized and diluted. The accumulated nutrient and biomass resources become more and more tightly bound within existing vegetation, hence, connectedness increases, eventually becoming over-connected and increasingly rigid in its control but ecological resilience is low. The system becomes an “accident waiting to happen” (Holling 2001).

In Ω phase strong destabilizing positive feedbacks between the destructing element (e.g. insect pest, fire, drought, grazing pressure) and established aggregates (e.g. trees in the mature forest, palatable plants in the savannah) result in a creative destruction or release of the established elements. Ecological resilience expands again during the late Ω and the early α phase since species have loose connections to others and function in a wide, loosely regulated basin of attraction (Holling & Gunderson 2002).

It has become clear that the adaptive cycle exhibits two major phases. A slow, incremental *front loop* of growth and accumulation and a fast *back loop* of reorganization and renewal. The first stage is predictable with higher degrees of certainty, whereas the outcomes of the second stage in the back loop can be highly unpredictable (Holling 2001). In this perspective, uncertainty and surprise[44] is inevitable (Berkes & Folke 1998; Holling 2001; Gunderson & Holling 2002). As Holling and Gunderson point out:

> "It is as if two separate objectives are functioning, but in sequence. The first maximizes production and accumulation; the second maximizes invention and re-assortment. The two objectives cannot be maximized simultaneously, they can occur only sequentially. And the success in achieving one tends to set the stage for its opposite" (Holling & Gunderson 2002: 47).

According to Gunderson and Holling (2002), this metaphorical descriptions suggest that attempting to optimize around a single objective is fundamentally impossible for adaptive cycles, although optimizing the context that allows such a dynamic might be possible.[45] Achieving both objectives needs a clear understanding of when it is appropriate to try to increase production efficiency, and when (and where) it is appropriate to try to ensure ecological resilience (Walker *et al.* 2002).

Following Holling (2001) four key features thus characterize an adaptive cycle and can be distinguished (Holling 2001). Those are: (1) the potential increases incrementally in conjunction with increased efficiency but also in conjunction with increased rigidity (the front loop from r to K); (2) as potential increases slow changes gradually expose increasing vulnerability (decreased ecological resilience) to such threats as fire or insect outbreak. A break can trigger the release of accumulated potential in a creative destruction (from K to Ω); (3) novel recombination can form where low connectedness allows unexpected combinations of previously isolated or constrained species (α phase). (4) Those innovations are then tested (r phase), some fail, but others survive and adapt in a succeeding phase from r to K.

Due to its generality this framework more or less represents a metaphor (Carpenter *et al.* 2001). It should not be read as a rigid, predetermined path and trajectory proposing that ecosystem succession necessarily follows this procedure. Systems can move back from K toward r, or from r directly into Ω, or back from α to Ω (Walker *et al.* 2004). Rather, the adaptive cycle is considered as a tool for thought, a heuristic

[44] "Surprise denotes the condition when perceived reality departs *qualitatively* from expectation" (Berkes & Folke 1998: 6).

[45] The resilience approach assumes that optimizing the context that allows the occurrence of the adaptive cycle at a small spatial and temporal scale is desirable in many cases, as it is impossible to surpress the dynamics of the adaptive cycle (cf. section 3.3.2.3.1).

that stimulates thoughts and hypothesis, which can be tested empirically (Gunderson & Holling 2002).

The argument that the adaptive cycle is a heuristic rather than a predetermined path has also been made by Cumming and Collier (2005). These authors consider the adaptive cycle to be merely one metamodel of ecosystem dynamics among others. Although the adaptive cycle offers a persuasive approach to characterizing and understanding system dynamics, it is only one of a set of possible metamodels that might explain or clarify different aspects of system behaviour. Alternative metamodels are represented by *replacement*, e.g. lotic ecosystems or volcanic eruptions on oceanic islands, *succession*[46] or *dynamic limitation*, e.g. the long-term dynamics of a boreal forest with respect to the movement of a glacier (cf. for more information Cumming & Collier 2005).

The important point is that the adaptive cycle does not apply to all situations and is not an useful metaphor for all system dynamics. For instance, there may be no release phase, as for instance the transition from a bog to a forest, or no conservation phase if external disturbances are intensive and early in the development stage. For these systems it makes no sense to speak of an adaptive cycle (Walker *et al.* 2006).

From what has been said in the previous paragraphs we can draw an important conclusion. If we take into account the metamodel of 'adaptive cycle', ecological resilience represents a dynamic system quantity that changes quantitatively on the patch scale as the cycle proceeds. There are times when the patch exhibits high ecological resilience (α and r phases), and there are times when the patch has low ecological resilience (K and Ω phases), i.e. when the system is either more or less vulnerable to internal and external fluctuations (Peterson 2000). This comprehension of ecological resilience has been dubbed the *systemic-heuristic definition* by Brand and Jax (2007).[47]

3.2.1.2 Panarchy

Based on the heuristic of the adaptive cycle described in the previous section, this section delineates the metamodel "Panarchy" for understanding ecosystem

[46] For a detailed review of the concept of *succession* confer Weidemann and Koehler (1997)
[47] For an overview of the various definitions of ecological resilience including the systemic-heuristic definition confer section 4.1.2.

dynamics. This notion of ecosystem dynamics is based on hierarchy theories (e.g. Allen & Starr 1982) and on the concept of self-organization, understood as processes that are generated, maintained and modificated by the system itself (Müller 1997).[48] This self-organization produces hierarchical structures – with smaller and faster distinct from larger and slower levels - *because* these structures increase the ecological resilience and thus maintain the whole system, whereas non-hierarchical do not (Peterson 2000).[49] Within resilience research, these hierarchical structures are not conceptualized in the sense of a top-down sequence of authoritative control. Rather, hierarchical structures are viewed to be semi-autonomous levels that are formed from the interactions among a set of variables that share similar speeds and spatial attributes (Holling 2001).[50] As long as the transfer from one level to the other is maintained, the interactions within the levels themselves could be transformed, or the variables changed, without the whole system losing it ecological resilience (Peterson 2000; Holling 2001).

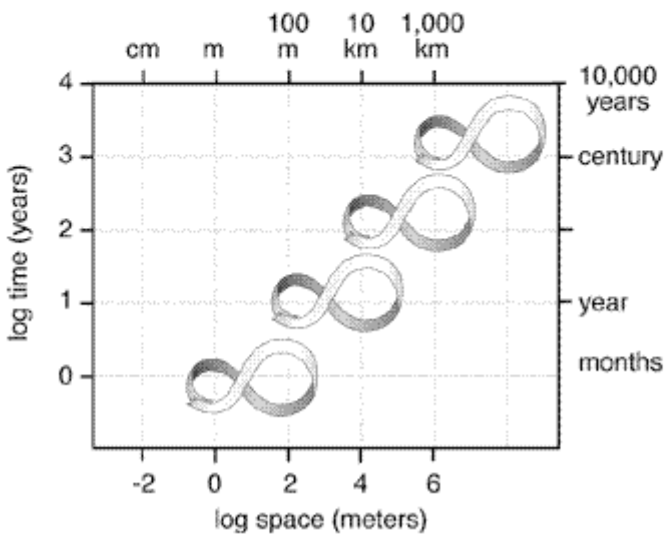

Figure 3: Panarchy: nested sets of adaptive cycles over various scales
from URL: http://www.resalliance.org, resource library

[48] The term *self-organization* is used ambiguously in ecology (for a detailed treatment cf. Müller 1997). It is interesting to note that the discussion on self-organized systems can be interpreted as a revitalization of the debate on cybernetic systems (Jax 2007).

[49] This notion includes facets of teleological thinking, which can be uncovered in several of the subsequent descriptions. Natural sciences are by definition free from teleological thinking and restricted to causal thinking. Yet teleological thinking is typical for the biological sciences and might be inevitable, just *because* from an ontological perspective there is indeed a telos in nature (Mutschler 2002).

[50] There is a wealth of hierarchy theories in ecology, which can be conveived as purposeful abstractions or models. For a review of hierarchy theories in ecology confer Wiegleb (1996).

Each of the levels of a dynamic hierarchy is seen to serve two functions, one is to conserve and stabilize conditions for the faster and smaller levels and the other is to generate and test innovations by experiments occurring within a level (Holling 2001), which is reflected by the r and K phases and the Ω and α phases, respectively, of the adaptive cycle explored in section 3.2.1.1.

The adaptive cycle is at first being proposed for the patch scale (Holling & Gunderson 2002). Yet there cannot be one single scale that covers the whole range of dynamics of complex systems (Levin 1998; Holling *et al.* 2002b). Dynamics occur at every scale of the hierarchy. Therefore, each spatial and temporal level – from plant to patch, to stand, to ecosystem, to landscape – follows its own adaptive cycle (Holling *et al.* 1995). As illustrated in *Figure 3*, there are nested sets of such four-phase cycles that interact with each other. These interacting hierarchies are dubbed *Panarchy* (Gunderson & Holling 2002).

A critical feature of this new notion of hierarchy are asymmetric interactions between levels. Slower and larger levels set the conditions within faster and smaller ones perform (Holling *et al.* 2002b). But at the two-phase transitions between gradual and rapid change and *vice versa*, the large and slow entities become sensitive to change from the small and fast ones. This results in dynamic, adaptive entities, rather than in fixed static structures (Holling 2001). There are, of course, multiple connections between the levels of a *Panarchy*. But two of these *cross-scale interactions* are seen to be particularly significant to the search for the meaning of sustainability[51] (Holling 2001). They are labelled *Revolt* and *Remember* (Holling 2001; Gunderson & Holling 2002).

When a level in the *Panarchy* enters its Ω phase of creative destruction and experiences a collapse, that collapse can cascade up to the next larger and slower level by triggering a crisis, particularly if that level is at the K phase, where ecological resilience is low. The “Revolt” arrow in *Figure 4* suggests this effect, where fast and small events overwhelm slow and large ones (Holling *et al.* 2002b). A good example is given by a ground fire that spreads to the crown of a tree, then to a patch in the forest, and then to a whole stand of trees. Each step in that cascade moves the transformation to a larger and slower level.

[51] It is highly unclear what Holling (2001) actually means when referring to the term “sustainability”. For a detailed treatment of the term “sustainability” confer section 5.1.

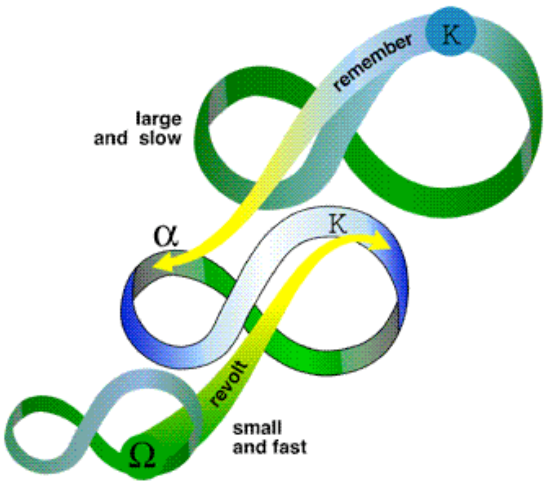

Figure 4: Interactions between scales: Revolt and Remember
from [online] URL: http://www.resalliance.org, resource library

The arrow labelled "Remember" in *Figure 4* indicates a second-type of cross-scale interaction. Once a catastrophe is triggered at one level, the opportunities for, or constraints against, the renewal of the cycle are strongly influenced by the K phase of the next slower and larger level. An opened patch in a forest due to a storm disturbance, for instance, is re-colonized by surviving species, seed-banks or seeds and living organisms from higher levels (e.g. shrubs and trees) of neighbouring patches or by invasions from distanced patches.

According to Holling (2001), a healthy[52] social-ecological system is thus a system where each level of the hierarchy is allowed to operated at its own pace, protected from above by slower, larger levels but invigorated from below by faster, smaller cycles of innovation. The whole Panarchy is therefore both creative and conserving, learning and continuity are combined.[53]

Note that the metamodels "adaptive cycle" and "Panarchy" can be critized as being not universally valid and limited, as they reflect a holistic and organic systems notion. This topic will be dealt with comprehensively in section 4.2.3.

[52] A critical discussion of the term "health of ecosystems" is provided by Steiner and Wiggering (2000).

[53] It is important to note that the resilience approach considers both the adaptive cycle and the *Panarchy* as being useful to describe not only ecological systems but also social-ecological systems. In this and the subsequent sections, however, my main focus is on dynamics in ecological systems.

3.2.2 Key Variables and the 'Rule of Hand'

The second assumption of „resilience theory“ (Anderies *et al.* 2006) is a concept coined "rule of hand". This "rule" or pattern proposed in some key publications (e.g. (Yorque *et al.* 2002; Walker *et al.* 2006) represents a specific approach for understanding and reducing the complexity in ecosystem dynamics. Before putting the rule of hand in a wider picture of "ecosystem functioning", this section delineates the basics and the empirical evidence of the concept.

According to the resilience approach, the hierarchical structure of ecosystems is primarily regulated by a small set of plant, animal, and abiotic processes. These processes can be identified as the *essential variables* or *key variables* (Holling 1992a; Holling *et al.* 1995; Holling 2001; Gunderson & Holling 2002). Holling (1992a) terms this comprehension of ecosystem dynamics the *extended keystone hypothesis*. Each of the key variables operates at characteristic periodicities, namely fastest, slower, and slowest temporal scales and specific small, meso- or large spatial scales. These variables establish a persistent template upon which a host of other variables exercise their influence. Such *subsidiary variables* can be interesting, relevant, and important for scientific questions, but they exist at the whim of the critical controlling processes and variables (Holling 2001).

Holling *et al.* (2002b) hold that three decades of studies of regional ecosystems from northern forest, southern wetlands, dry grasslands, lakes and marine ecosystems show that the key features of ecosystems are established by the interaction between these fast and slow essential variables. At the patch scale, biophysical processes dominate dynamics, such as plant physiology, inter-specific plant competition processes and meso-scale disturbance processes of fire, storm, or herbivory, for instance. In contrast, at the larger landscape or regional scale climate, geo-morphological, and bio-geographical processes alter ecological structure (Holling *et al.* 2002b). Following the resilience approach, the complexity of adaptive systems can therefore be traced to interactions among three to five sets of variables, each operating at a qualitatively distinct speed and scale. Hence the expression "rule of hand", with reference to the five fingers of a hand.[54] Note that Peterson (2002a)

[54] The number of three to five variables is in my view an educated guess in order to reduce complexity. This top-down approach to ecological modelling can be critiziced by individual-based modellers (cf. Grimm 1999 and section 3.4.2.3.1.1).

suggests that this extended keystone hypothesis is most likely to be demonstrated in ecosystems that exhibit strong *ecological memory* [55].

In this sense, the behaviour of complex systems both is generated and can be understood by three to five sets of variables, the nonlinear relationships between them and stochastic processes (Holling *et al.* 2002b). Thus, according to the rule of hand important changes in ecosystem dynamics can be understood by analyzing a few, typically no more than five, key variables (Yorque *et al.* 2002; Walker *et al.* 2006). These key variables may be mainly driven by *keystone process species*, while the remaining species exist in the niches formed by these keystone process species (Folke *et al.* 1996).

The key variables are in turn separated into *fastest variables*, *slower variables* and *slowest variables* as regards to their characteristic periodicities in space and time. For simplicity, the key variables are often summarized in the relevant literature in *fast variables* and *slow variables*. Holling *et al.* (2002b) suggest representative key variables for some ecosystem types, which are listed in *Table 2*.

Table 2: Variables on disjunct time scales (fastest, slower, slowest) for different types of systems
from Holling *et al.* 2002b

	Variables		
System	**Fastest**	**Slower**	**Slowest**
Forest-pest dynamics	Insect	Foliage	Tree
Forest-fire dynamics	Intensity	Fuel	Trees
Savannah	Annual grasses	Perennial grasses	Shrubs and grazers
Shallow lakes and seas	Phytoplankton and turbidity	Sea grasses	Grazers
Deep lakes	Phytoplankton	Zooplankton	Fish and habitat; phosphate in mud
Wetlands	Periphyton	Saw grasses	Tree island; peat accreation

[55] Ecological memory is a concept to comprehend the relation between biodiversity and ecological resilience at the landscape scale. For more details confer section 3.3.2.2.

Note that the slowest variables in *Table 2* refer to a community (e.g. fish community) or population (e.g. tree population) but also to a nutrient storage (e.g. phosphate in mud in lakes). There are a couple of examples in relevant literature for the identification of slowest variables. Jansson and Jansson (2002) suggest various slow and fast variables for the Baltic Sea. Nutrient storages such as phosphorus or the nitrogen/phosphorus ratio represent slow driving variables whereas phytoplankton, annual seaweeds, the microbes, and pelagic, particulate matter are identified as fast variables. In the Everglades, interactions occur between fast variables, for instance salinity variation in sea-grass beds or ignition sources in fires, and slow variables such as fuel loads in fires or nutrient levels or biomass of sea-grass beds (Gunderson & Walters 2002). Walker *et al.* (2006) list soil, sediment concentrations and long-lived organisms as the slow variables of ecosystems. These authors also distinguish between two *types* of slow variables. One is characterized by a low *rate* of change, the other by a low *frequency* of change.

Slow variables do not perform isolated within an ecosystem; they are rather connected to faster variables by mechanisms such as *Revolt* and *Remember*. An attempt to understand these interactions is to try to model them on disjunct time scales. Rinaldi and Scheffer (2000) for instance, use models dubbed *singular perturbation approach* or *slow-fast-analysis.*[56] These authors state that the interaction of slow and fast variables can be analyzed relatively easily. It appears to be difficult however, to include more than three state variables or a higher number of time scales which is, in turn, needed to explain the overall functioning of ecosystems. According to Carpenter and Turner (2000) these interactions between fast and slow variables are among the most important topics in ecology for achieving a wise environmental policy and with reference to Carpenter *et al.* (1999) the most important scientific information for sustainable management. As Holling *et al.* point out: "[a]nalysis should focus on the interaction between slow phenomena and fast ones, and monitoring should focus on long-term, slow changes in structural variables" (Holling *et al.* 1998: 354).

What is the empirical evidence for these small sets of key variables? Ecosystem dynamics may be understood as a discontinuous template in space and time that

[56] Again it is far beyond the scope of this thesis to point out the whole discussion on this topic. The purpose of this section is to throw some light on the interaction of fast and slow variables since these interactions and slow variables are seen to be crucial for the "threshold approach" to estimate ecological resilience. Confer for the estimation of ecological resilience the chapter about operationalization 3.4.

entrains attributes of variables into a number of distinct *lumps*. Some authors within resilience research state that the attributes of each size, speed, and ecosystem processes are distributed in a lumpy manner (Holling 1992a; Allen & Holling 2002). Ecosystem dynamics thus create landscape structures with scale-specific pattern. This is termed the *textural discontinuity hypothesis* (Allen & Holling 2002). The discontinuous spatial patterns can be produced by simple interactions between contagious disturbance processes and vegetative dynamics, if the landscape holds a memory (Peterson 2002a). Ecological processes that are strongly "remembered" by an ecosystem have the potential to become key processes that generate ecological structure at specific scales and thus have the potential to entrain other processes and ecological attributes (Allen & Holling 2002; Peterson 2002a).
This lumpy ecosystem structure is echoed by a discontinuous distribution of species body masses showing body mass lumps and gaps. This pattern is dubbed the *world-is-lumpy-hypothesis* (Holling 1992a).[57] This entrainment reflects adaptations to a discontinuous pattern of resource distribution acting on animal community assembly and evolution both by sorting species and by providing a specific set of evolutionary opportunities and constraints. Animals within a particular body-mass aggregation perceive and exploit the environment at the same range of scale (Allen & Holling 2002). Based on this empirical evidence, Holling *et al.* (2002b) suggest a strong correlation between complexity of lump structure and productivity (or other correlates of net energy flux through terrestrial ecosystems). Boreal landscapes - as an example for more complex and productive ecosystems - show about eight lumps in body mass, whereas simple marine landscapes have three to four body mass lumps. These lumps are conceived as constant (Hansson & Helgesson 2003), yet populations within them are not (Forys & Allen 2002).
In addition, Allen *et al.* (1999) suggest that this discontinuous pattern has predictive power. Both the invasions or extinctions of species in landscapes subject to human transformation (Allen *et al.* 1999) and the emergence of nomadic birds (Allen & Saunders 2002) tend to be located at the edge of body-mass lumps. This, in turn, makes the existence of the body-mass lumps more likely. The distribution of lumps and gaps is a kind of bioassay of the structure of the 'Panarchy' and is viewed as evidence for both the hierarchical structure of ecosystems and the existence of essential variables operating at distinct speeds and scales (Allen & Holling 2002;

[57] There are sceptical views that doubt whether such lumps are real (Manly 1996).

Holling *et al.* 2002b). There is thus some empirical evidence for the lumpy organisation of ecosystems that is proposed in order to validate the rule of hand-analogy: important changes in ecosystems can be understood by analyzing three to five key variables.

Yet this notion of key variables proposed by resilience research is limited. There are at least two important points. First, the notion of key variables assumes a top-down approach to ecological modelling, which is partial and should be complemented by individual-based models (Grimm 1999) (cf. section 3.4.2.3.1.1).

Second, the notion of key variables can be interpreted as a specific approach to the "functioning" of ecosystems, in the sense of how the whole is sustained (Jax 2005).[58] The resilience approach holds that as long as the slow variables (or specific keystone process species) are maintained at a certain value the functioning of the regime is maintained. Yet this criterion is non-empirical and the key variables-approach represents only one among a variety of approaches to the functioning of ecosystems.

For instance, Breymeyer (1981) considers the existence or non-existence of basic ecosystem processes as the pivotal criterion for the functioning of ecosystems. *Basic ecosystem processes* are considered to be the production of organic matter, its consumption and incorporation into the bodies of successive consumers and its break down by decomposers. Breymeyer (1981) states that it is principally possible to observe the disappearance of the basic ecosystem processes, and if any of them disappears, the ecosystem, and thus also the functioning of the ecosystem, no longer exists.

Alternatively, some authors consider the provision of ecosystem services as the decisive criterion for the functioning of ecosystems. If desirable ecosystem services, such as food production or water purification, are maintained by a certain state of the ecosystem, the functioning of the ecosystem is also maintained (de Groot *et al.* 2002; UNEP 2005b).

A still other approach refers to the specified identity or self-identity of an ecosystem (Jax *et al.* 1998; Cumming & Collier 2005; Jax 2006). For example, Jax *et al.* (1998) argue that the self-identity of an ecosystem should be specified by referring to four characteristics: (1) if the boundary of the regime is defined topographically or functionally; (2) the expected internal degree of relationships; (3) the set of selected elements; and (4) the degree of component resolution (cf. for more details section

[58] Often, the meaning or conceptualization of the "functioning" of ecosystems is not explicitly specified in the relevant literature. For the different meanings of the term "function" confer section 2.3.

3.4.2.1.2). The scientist specifies the self-identity of the ecosystem she studies at two points in time and if the self-identity of the ecosystem stayed the same, the functioning of the ecosystem has been maintained. Apparently, the self-identity method refrains from an ontological view on ecological units (cf. section 2.4).

Altogether it has become clear that the key variables-approach put forward by resilience research is only one among a variety of approaches to the functioning of ecosystems. The decisive question is then whether or not the specific approach is useful to approach a particular research question.

3.2.3 Theory of Alternative Stable States

The third assumption of the resilience approach is the theory of alternative stable states. At first section 3.2.3.1 describes the theory of alternative basins of attraction, before section 3.2.3.2 delineates the nature of regime shifts between these alternative basins of attraction. Subsequently, section 3.2.3.3 explores the concept of ecological thresholds including threshold progressions and the possibilities to predict a threshold position.

3.2.3.1 Alternative Basins of Attraction

Different paradigms in ecological science have profound implications on how to perceive nature. Put simply, there are two opposing paradigms that have emerged in the history of ecology. First, the *equilibrium paradigm* states that for any system there is only one equilibrium or steady state and that any system returns to this equilibrium after disturbance (e.g. Patten 1975; Pimm 1984). The equilibrium paradigm has been highly influential in the last decades within ecology as well as in other sciences. Ecological concepts ranging from succession, island biography to carrying capacity are all dominated by equilibrium assumptions. Notions of *global stability* or *balance of nature* are still prevalent in the heads of the general public.

Second, the *non-equilibrium paradigm* holds that ecosystems can exhibit *alternative stable states* or *alternative basins of attraction* (e.g. Gunderson 2000; Walker *et al.* 2004). There is only *local stability* and when disturbed a system may shift to another basin of attraction, which is characterized by a different structure and different processes.[59]

The opposing views constitute what is called the *alternative-stable-state-controversy*, which is at the heart of the two different views of resilience (i.e. “engineering resilience” *vs.* “ecological resilience”, as considered in section 3.1) (Holling 1996). The existence of alternative stable states is regarded as the key distinction between these two views (Gunderson 2000). This section focuses on the latter view and is mainly concerned with the theory on ecological resilience.

[59] In addition to the assumption of alternative stable states, the non-equilibrium paradigm includes several other characteristics (Wallington *et al.* 2005). Also, both the concepts of equilibrium and non-equilibrium have been used differently in relevant literature. Confer for a review on this issue Rohde (2005).

Before turning to this body of work in detail some words on the terminology used are in order. The terms "equilibrium" and "state" have been criticized for their static connotations (Scheffer & Carpenter 2003). In fact, populations, communities and ecosystems are always changing and fluctuations in abundances and distributions are the rule. In the following, I will use the terms *regime* or *configuration*, defined as a dynamic state of nature, in order to express the dynamic character of nature. In this I follow the terminology used by Scheffer and Carpenter (2003) as well as Walker *et al.* (2002).[60] Moreover, the terms *alternative stable regimes* and *multiple stable regimes* are employed interchangeable in the relevant literature and I will also use them as synonyms.[61]

There are different ways in which an ecosystem can respond to change in conditions, e.g. temperature, land use pressure or nutrient load, as shown in *Figure 5* (Scheffer & Carpenter 2003).

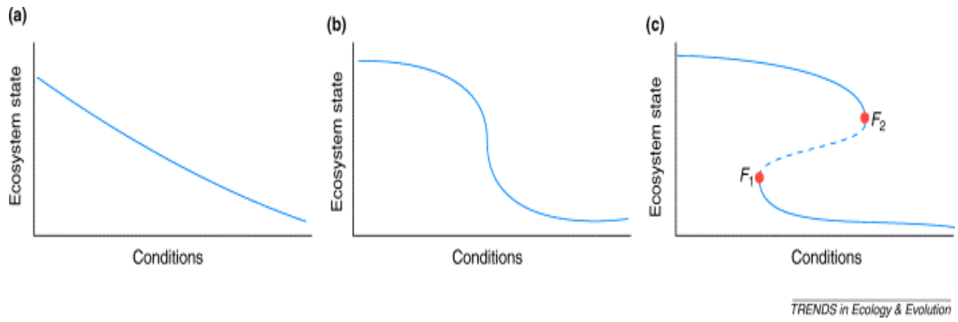

Figure 5: The responses of an ecosystem to a change in conditions
from Scheffer and Carpenter (2003: 649)

Although systems can respond smoothly to change in external conditions (*a*), they can sometimes change profoundly when conditions approach a critical level (*b*) or have more than one stable regime over a range of conditions (*c*). Cases (*b)* and (*c)* in *Figure 5* represent a case of *non-linear dynamics*, where even a tiny incremental change in conditions can trigger a large shift in an ecological unit. In particular, case (*c*) shows the occurrence of alternative stable regimes or basins of attraction. This

[60] The term "equilibrium" is also highly ambiguous. For a detailed treatment of "equilibrium" confer Weil (1999).

[61] Some authors use the term "quasi-alternate states" to describe systems that have regimes that are not stable but in fact slowly changing and only appear as stable on a relatively short time-scale. Semiarid grazing systems are an example. It might not be easy to separate *quasi-alternate states* from *alternative stable regimes* clearly.

implies that occasional catastrophic regime shifts may occur in response to gradual environmental change. In what follows this section takes a closer look at the theory on alternative stable regimes or alternative basins of attraction in ecological[62] systems.

Perceived from a model perspective, single alternative stable regimes (assumed as a more or less extended point in state space) are located *within* different basins of attraction (a larger area in state space). A *basin of attraction* is defined as a region in state space, in which the system tends to remain (Walker *et al.* 2004). For example, a shallow to intermediate depth lake appears to exhibit two regimes with respect to nutrient load. In the oligotrophic, or clear-water regime, the water has a low biomass of phytoplankton and low recycling rates of nutrients from sediment to water. In the eutrophic, or turbid-water regime, phytoplankton biomass is high, often forming noxious blooms, and recycling of nutrients from sediment to water is rapid (Dent *et al.* 2002). These two regimes of lakes correspond to the occurrence of alternative basins of attraction within state space.[63]

Formally a system exhibits alternative basins of attraction when its state variable (e.g. abundance of dominant species, productivity) responds to environmental change by a backwards folding curve, as shown in *Figure 6* (Scheffer *et al.* 2001; Scheffer & Carpenter 2003; Schröder *et al.* 2005). Because of the backward fold, two stable basins overlap, separated by an unstable one over a given range of environmental parameters. When the system is in a state on the upper branch of the folded curve in *Figure 6* (e.g. the clear-water regime of the lake considered above), it can not pass to the lower branch (e.g. the turbid-water regime) smoothly. Instead, when conditions change sufficiently to pass a critical value (e.g. a specific nutrient level in lake water) a catastrophic transition to the lower branch occurs, either caused by only an incremental change in conditions or due to a bigger disturbance. To induce a switch back to the upper branch it is not sufficient to restore the environmental conditions of before the collapse. Instead, one needs to go back further, beyond the other switch point, where the system recovers by shifting back to the upper branch – a pattern known as *hysteresis* (Scheffer *et al.* 2001).

[62] It is important to note that resilience research examines regime shifts not only in ecological systems, but also in social, economic and social-ecological systems (Walker & Meyers 2004; Kinzig *et al.* 2006). Yet in this thesis I focus on ecological systems; but confer section 3.5.2.3.

[63] If the dynamics of ecosystems are determined by two attracting regimes, this is also termed “bistability” by some authors (e.g. Rietkerk *et al.* 2004).

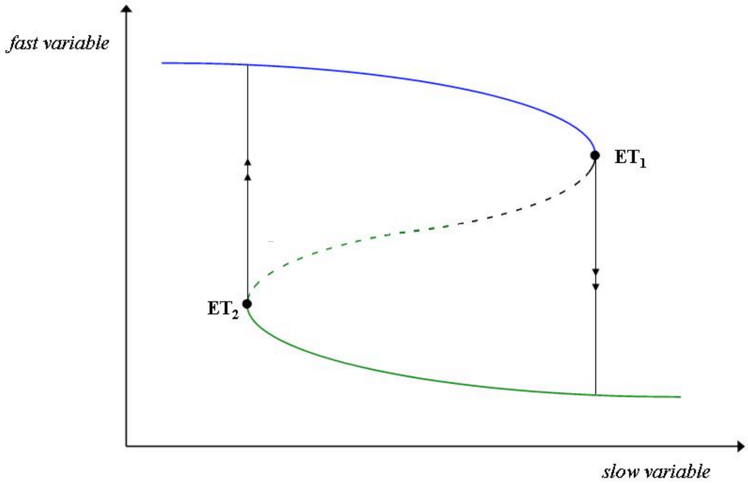

Figure 6: Bifurcation diagram of a system described by a fast variable and a slow variable: the stable regimes are given by the blue and green solid lines and the boundary of the basins of attraction (unstable state) by the dashed line. ET_1 and ET_2 represent ecological threshold points.
from Brand and Jax (2007); modified from Scheffer and Carpenter (2003)

The empirical detection of hysteresis-driven regimes or alternative stable regimes is considered to be elusive yet principally possible (Carpenter 2001). Such behaviour can be revealed, for example, by non-manipulative empirical approaches including time series analyses, between-site comparisons and separate regression analyses (Scheffer & Carpenter 2003). Mounting observational and theoretical evidence suggests that alternative basins of attraction are common for many ecosystem types, such as temperate lakes, tropical lakes, wetlands, estuaries and coastal seas, coral reefs, kelp forests, pelagic marine fisheries, savannas, woodlands, deserts, forests, arctic tundra and oceans. For the evidence on alternative stable states confer Done (1992), Knowlton (1992) and McClanahan *et al.* (2002) for coral reefs, Dent *et al.* (2002) for rivers and lakes, Hsieh *et al.* (2005) for marine systems, Rietkerk *et al.* (2004) for arid, savanna and peatland ecosystems and Carpenter (2001), Scheffer *et al.* (2001) and Folke *et al.* (2004) for general overviews.
Yet some authors suggest that while results based on observational approaches may be in line with predictions, they do not rule out alternative, often more parsimonious explanations like an abrupt, but still continuous response to driving parameters or cycles with a long periodicity. Hence, only experimental evidence might demonstrate

the existence of alternative stable regimes (Schröder *et al.* 2005).[64] Hereby, conclusive experimental approaches involve (1) the existence of different parameter thresholds for back- and forward shifts (*test for discontinuity*); (2) state transitions after disturbances (*test for non-recovery*); and (3) sensitivity of the stable end regime to initial conditions (*test for divergence*) (Schröder *et al.* 2005).

Experimental studies range from single-species laboratory-cultures to manipulated field communities but contain a strong bias towards laboratory experiments. According to this body of work, the existence of alternative stable regimes cannot be generalized over all ecosystem types. Only 13 out of the 21 conclusive experiments (62%) found evidence for alternative stable regimes (Schröder *et al.* 2005). In addition, some *observational* studies show the lack of alternative stable regimes. For example, Sim *et al.* (2006) argue that a concept of alternative stable regimes is not appropriate to understand the dynamics of seasonal drying wetlands in South-Western Australia.[65] This indicates that the concept of alternative stable regimes is just one possibility among others to describe the behaviour of ecological systems.

The important point is that the existence of alternative stable regimes is no ecological law or rule. Since the concept of alternative stable regimes is one of the basic assumptions of resilience theory this insight has large implications on all levels of the anatomy of resilience theory. Some systems may behave in accordance with other types of non-equilibrium theory or equilibrium theory, thus, may not exhibit alternative stable regimes and will require a different method of operationalization and a different management approach.

However, beyond the case-by-case arguments due to context-dependency, are there any generalisable predictions that can be made about which systems are more likely to exhibit alternative regimes? It is possible to perceive different ecosystem types along an axis from strong environmental adversity (e.g. deserts, intertidal mud flats, savannas) to strong competitive adversity (e.g. coral reefs, tropical forests). In a review of the empirical evidence Didham *et al.* (2005) hold that the overwhelming majority of cases in which alternative stable regimes have been detected comes from systems that were historically subject to moderate to extreme abiotic regimes, for

[64] The strongest arguments for alternative stable regimes would be based on multiple lines of evidence, blending long-term observation, experimentation, comparisons of contrasting ecosystems, and modelling tied closely to data (Carpenter 2001).

[65] Likewise, in my view the concept of alternative stable regimes does not apply in a smooth fashion to many ecosystem types characteristic for central Europe, such as deciduous forests or agricultural landscapes. This may be one reason for the weak development of resilience research in Germany.

example wetlands, streams, deserts, arid grasslands, rangeland, woodland, savannas, salt marshes or intertidal mud flats. This does *not* suggest that *all* abiotically-controlled systems will necessarily exhibit alternative stable regimes (there are for example grasslands and wetlands that do not), or that systems with weak abiotic gradients or disturbance regimes *never* exhibit alternative regimes (as is clear from empirical examples of coral reefs, lakes or tropical forests). Didham *et al.* (2005) rather raise the possibility that systems with strong underlying abiotic regimes may (*a*) be more prone to enter alternative regimes, (*b*) switch more readily to an alternative regime following a lower level of disturbance, or (*c*) are more difficult to restore, than systems which are weakly structured by environmental adversity. Didham *et al.* (2005) explain this pattern by three major processes controlling community assembly, namely *propagule limitation*, *stochastic priority effects*[66] and the *alteration of the regional species pool* through extinction and invasions.

This view is seriously challenged by Fukami and Lee (2006).[67] These authors propose exactly the opposite in suggesting that competitively-structured communities will be more likely to exhibit alternative stable regimes than are environmentally-structured communities. In a reply to Fukami and Lee, Didham and Norton (2006) insists on the original proposition made by Didham *et al.* (2005) described above. Indeed, the weight of empirical evidence shows that the *relative frequency* of the occurrence of alternative stable regimes across systems is higher for systems controlled by environmental adversity, such as deserts, arctic tundra, intertidal mud flats or savanna (Didham & Norton 2006).

It is important to apply the ecological checklist, considered in section 3.3.1.1, to the definition of alternative stable regimes. For example, the alternative stable regimes just considered are observed at a certain level of description (i.e. the ecosystem level) and at a certain scale (local or regional scale). Yet alternative stable regimes or states occur likewise at other levels of description and other scales. For example, the non-equilibrium nature of local population dynamics is often stressed (i.e. level of description is the population level). Alternative stable regimes can extend even to global levels. Flips in global climate regimes seem to have occurred in the past over very short time scales (Levin 2000; Foley *et al.* 2003). Moreover according to

[66] *Stochastic priority effects* correspond to variation in community assembly that arises from species-specific effects of early colonists on subsequent succession, regardless of whether the sequence of colonization was determined by a significant difference in propagule pressure between species, or by chance events.

[67] A recent contribution complements the critique of Fukami and William (2006). Mason *et al.* (2007) also argue that the argumentation of Didham and Watts (2005) is not valid.

Scheffer and Carpenter (2003), alternative stable regimes can coexist side-by-side. For instance, a clear and a turbid regime can coexist even in a single lake and landscapes often comprise a mosaic of patches with different alternative stable vegetation types.

3.2.3.2 Regime Shifts

The existence of alternative stable regimes are common for some - or even many - ecosystem types, such as intertidal mud flats, tundra ecosystems, savannahs, coral reefs or shallow lakes. These systems may thus experience a *regime shift*, i.e. the system shifts from one alternative stable regime to another. How do these regime shifts occur?

As empirical evidence shows a regime shift may be smooth (i.e. *gradual, linear*) or abrupt (*non-gradual, non-linear*), this depends on the nature of the system and the type of disturbance (Frelich & Reich 1998; Scheffer & Carpenter 2003). There is evidence for abrupt and surprising regime shifts in response to gradual environmental change (i.e. for *non-linear regime shifts*) with respect to models (van Nes & Scheffer 2005), observations (Carpenter 2001; Scheffer *et al.* 2001; Folke *et al.* 2004) and experiments (Schröder *et al.* 2005). Although disrupting disturbances seem an obvious explanation for such shifts, evidence for large disturbances is not always apparent. An alternative explanation is that systems might shift between alternative *attractors*, i.e. that systems can exhibit alternative stable regimes as described in detail above. A regime may become increasingly brittle to the point that even a tiny incremental change in conditions can trigger a large shift to an alternative regime (Carpenter 2001; Scheffer & Carpenter 2003; van Nes & Scheffer 2005).

The characteristic of a regime shift is dependent on the scale of concern. The response of the ecosystem viewed at a landscape scale may be experienced as gradual (i.e. the average of many asynchronous small shifts), while the response regarded at a the patch scale is catastrophic. According to van Nes and Scheffer (2005), spatial heterogeneity at the landscape scale may weaken the tendency for large-scale catastrophic regime shifts (if dispersion is unimportant or if local environmental characteristics vary along a smooth gradient). Thus, large-scale regime shifts may be restricted to systems that are relatively homogeneous, characterized by relatively intense dispersion of matter or organisms or prone to large-scale disturbances, such as volcanic eruptions.[68]

The driving force of a regime shift comes either from outside or from within the system. In the former case a large natural disturbance (e.g. volcanic eruption, hurricane) or human impact (e.g. intensity of use, habitat fragmentation, addition of pollutants or new species) is responsible, whereas in the latter case internal changes (e.g. fog precipitation, resource concentration mechanisms) represent the cause for a regime shift to an alternative stable regime (Walker & Meyers 2004). In some areas, the susceptibility of regions to regime shifts may be enhanced or even "created" by changes in climate or orbit and incoming solar radiation. For instance, sometime between 6000 years ago and today, the Earth's orbital forcing caused the climate-vegetation system over the Sahara to change from a system with only one stable regime (the "green Sahara") to a system with two possible alternative stable regimes (a "green Sahara" or a "desert Sahara" as we know it today) (Foley *et al.* 2003).

In a model perspective, regime shifts are caused by either (*a*) changes in state variables (e.g. population size, competition) or (*b*) changes in parameters (e.g. abiotic factors, ecosystem drivers). This distinction corresponds to two different contexts in which the term "alternative stable states" is used in the ecological literature (Beisner *et al.* 2003). The *community perspective* holds that different states exist simultaneously under the same set of conditions in state space and that the community be conveyed from one state to another by a sufficiently large perturbation (Schröder *et al.* 2005). In contrast, the *ecosystem perspective* has focussed more on the effects of changing parameters (or environmental drivers) within the community. The parameters correspond to slowly changing quantities in a system model, such as long-lived organisms or nutrient storages (Scheffer & Carpenter 2003; Folke *et al.* 2004; Walker *et al.* 2004).

The two views can be visualized by a *ball-in-cup analogy* outlined in *Figure 7.* All conceivable states of the system can be represented by a surface or landscape, with the actual state of the community (e.g. the abundances of all populations) as a point or a ball residing on this surface. The movement of the ball can be anticipated from the nature of the landscape. In the simplest representation of alternative stable regimes, the surface has two basins, with the ball residing in one of them. Valleys or dips in the surface represent domains of attraction for a regime (balls always roll into

[68] But confer the predictive theory of regime shifts especially for *heterogeneous* ecosystem types (Rietkerk *et al.* 2004), as described in this section below .

that regime once in the domain). There are two ways the ball can move from one basin to the other. Either the ball moves, i.e. the state variables change (*community perspective*), or the landscape itself changes, i.e. the parameters of the model change (*ecosystem perspective*) (Beisner *et al.* 2003).

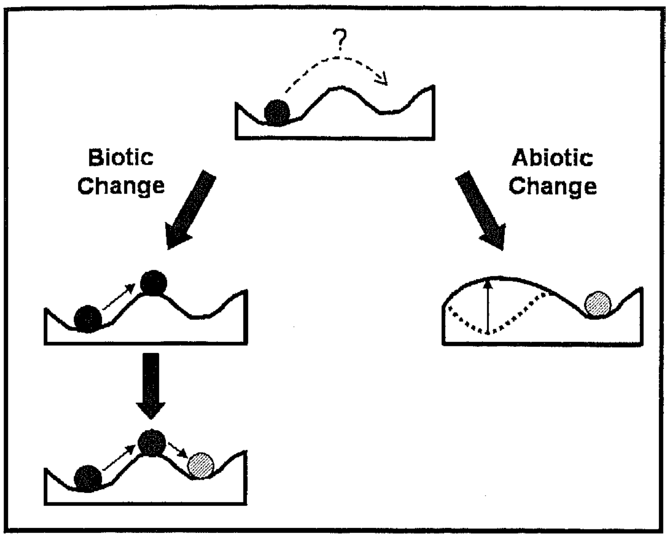

Figure 7: Ball-in-a-cup analogy of an ecosystem experiencing alternative stable states
from (Briske *et al.* 2006: 227), modified from (Beisner *et al.* 2003: 377)

A stable regime (for example the black dot in *Figure 7*) is reinforced by negative *feedbacks* or *resilience mechanisms* (Briske *et al.* 2006). Thus, the *trigger* of a regime shift may be a single event, e.g. a strong fire or storm, but the *cause* of a regime shift is often related to the long-term loss of ecological resilience in terms of resilience mechanisms, such as functional redundancy, response diversity, source habitats or the suppression of small-scale disturbances (Folke *et al.* 2004; Briske *et al.* 2006; Nyström 2006). I will consider the issue of resilience mechanisms in much more detail in section 3.3.

Yet why care for regime shifts after all? Viewed from a societal perspective, regime shifts can have profound effects for humans in terms of the loss of ecosystem services, such as food production, renewable resources or aesthetic values. Thus, the prediction of regime shifts would be of high value for conservation and environmental management.

But is it possible to predict regime shifts? Foley *et al.* (2003) state that while the exact timing and circumstances of regime shifts may be unpredictable, the *existence* of alternative stable states can be – and has been – predicted. It might be possible to forecast the future of certain regions in terms of *probabilistic assessments of the conditions* that favour one stable regime or the other. For example, the Sahel region is predisposed toward having at least two alternative stable regimes – the “wet Sahel” and the “dry Sahel”. A shift from the “wet Sahel” to the “dry Sahel” was caused by strong feedbacks between the climate and the vegetation cover and may have been triggered by slow changes in either land degradation or sea-surface temperatures. By analyzing the slow changes in these underlying environmental conditions (climate, vegetation cover, sea-surface temperature) it might be possible to assess the probability for the occurrence of either the “dry Sahel” or the “wet Sahel” and thus to predict a regime shift (Foley *et al.* 2003).

Another example comes from studies of spatially extensive, heterogeneous ecosystems, such as arid systems (deserts, arid bushlands), savannahs or peatland (Rietkerk *et al.* 2004). At larger scales, all these system exhibit either a stable regime that is characterized by *self-organized patchiness*, i.e. resources and biomass are distributed in a spatially concentrated manner caused by several resource concentration mechanisms (e.g. arid “tiger bush and “leopard bush” systems or spatial patterning in bogs) or an alternative stable regime that is characterized by a homogeneous distribution of resources and biomass. Arid ecosystems, for instance, shift abruptly from a heterogeneous regime containing spotted plant biomass to a bare homogeneous regime if rainfall is decreased beyond a certain threshold point. Rietkerk *et al.* (2004) suggest that *all* ecosystems with self-organized patchiness resulting from a resource concentration mechanism are prone to catastrophic shifts to a homogeneous state. This may be channelled in a *predictive theory of catastrophic shifts* from which early warning systems can be developed on the basis of spatial explicit time-series data (Rietkerk *et al.* 2004). Thus, while it might not be possible to predict the exact timing[69] of a regime shift between alternative stable states, it may be possible to predict which ecosystems of the world are susceptible to regime shifts and which are not.

[69] But see the prediction of the position of ecological thresholds considered in section 3.2.3.3.4.2.

3.2.3.3 Ecological Thresholds

This section explores the concept of ecological thresholds. At first, a threshold typology and analysis is being put forward (section 3.2.3.3.1). Subsequently, this section describes the three threshold types in more detail, i.e. critical loads (section 3.2.3.3.2), extrinsic factor thresholds (section 3.2.3.3.3) and dynamic regime thresholds (section 3.2.3.3.4). The latter section describes stability landscapes and threshold progressions (section 3.2.3.3.4.1) as well as the methods to predict the threshold position (section 3.2.3.3.4.2).

3.2.3.3.1 Threshold Typology and Analysis

Consider a shallow lake in Southern Bavaria in Germany exhibiting a clear-water regime. The surrounding area experiences agricultural practices comprising the use of fertilizers to enhance plant growth. Thus, increased run-off on agricultural lands increases the input of nutrients (due to the fertilizers) to the lake causing an increase in biomass of phytoplankton. The important point is that *at a critical breakpoint in state space* the system shifts from the clear-water state to a turbid water state, which is characterized by high biomass of phytoplankton and rapid recycling of nutrients from sediment to water. That is, a regime shift alters feedback mechanisms (Walker & Meyers 2004; Briske *et al.* 2006) (i.e. resilience mechanisms[70]) causing the ecosystem to reorganize around another set of controlling variables and processes. The critical breakpoint in state space between two regimes of a system, i.e. the critical values of the variables around which the system shifts from one stable regime to the other, is termed *ecological threshold* (Muradian 2001; Walker & Meyers 2004).[71]

A number of definitions have been put forward. In a generic sense, *ecological thresholds* can be defined as *the point or zone within state space at which there is an abrupt change in quality, property or phenomenon or where small changes in a driver may produce large responses in an ecological unit* (Muradian 2001; Huggett 2005; Groffman *et al.* 2006). In the life and natural sciences, thresholds have been investigated since the late 18th century (cf. for literature Huggett 2005). In ecology,

[70] For a detailed description of the topic of "resilience mechanisms" confer section 3.3.

[71] Note that ecological thresholds are the result of several potentially interacting components, rather than simple boundaries in time and space, as shown below.

interest in the concept arose from the idea that ecosystems often exhibit alternative regimes depending on environmental conditions (Holling 1973). Early critics mentioned the rather arbitrary specification of threshold locations (Woodwell 1974). Since then research has been largely theoretical in nature (Huggett 2005), although an increasing number of empirical studies are examining the concept at the level of populations and communities (Radford & Bennett 2004; van der Ree *et al.* 2004; Lindenmayer & Luck 2005; Suorsa *et al.* 2005) as well as ecosystems (Scheffer & Carpenter 2003; Walker & Meyers 2004; Briske *et al.* 2006).

It seems reasonable to expect thresholds to occur for some individual populations, particularly in relation to key environmental (eco-physiological, aut-ecological) factors, such as temperature, light and rainfall, while identification of thresholds at the community and ecosystem level may be more problematic (Lindenmayer & Luck 2005). However, in the following this section suggests that ecological thresholds may be very common for whole ecosystems as well.

There are three main ways that threshold concepts have been applied in ecology (Groffman *et al.* 2006). Those are: (1) the determination of *critical loads*, which represent the amount of pollutant that an ecological unit can safely absorb before there is a change in a particular ecosystem process; (2) the analysis of *extrinsic factor thresholds*, where changes in a variable at a larger scale (e.g. habitat loss and fragmentation) alter variables at a small scale (e.g. species richness); and (3) the analysis of dramatic and *surprising shifts of an ecological unit between alternative stable regimes*, where a small change in a driver causes a marked change in the condition of the ecological unit. Note that although there are fundamental differences in the three major types of thresholds, there is much overlap and interaction among them.[72]

These three major types of ecological thresholds can be related to two further sub-distinctions. The first sub-distinction highlights the *level of description* of the threshold. The detection of ecological thresholds may occur at the ecosystem level, the community level or for individual populations (Luck 2005). Moreover, ecological thresholds may be *point-type*, i.e. shifts between different states are abrupt or sudden, or *zone-type*, i.e. shifts occur in a rather gradual or smooth fashion (Huggett

[72] Even though there is much overlap among the three types of ecological thresholds, I regard this classification as being useful, as it refers to three "classical" environmental problems, namely pollution, habitat fragmentation and habitat conversion. In the course of this thesis the latter two threshold types and environmental problems will be of particular relevance.

2005). Taken together the major types of thresholds and the two sub-distinctions result in a fairly rough *threshold typology* (cf. *Table 3*). Within this typology, thresholds exist with respect to (1) (*a*) critical loads, (*b*) extrinsic factors or (*c*) regime shifts, occur (2) at the (*a*) population, (*b*) community or (*c*) ecosystem level and may be (3) (*a*) point-type or (*b*) zone-type.

Table 3: A rough typology for ecological thresholds

	(1) Major type	*(2) Level of description*	*(3) Width in state space*
Ecological threshold	***(a) Critical load*** ***(b) Extrinsic factor*** ***(c) Regime shift***	***(a) Population*** ***(b) community*** ***(c) ecosystem***	***(a) point-type*** ***(b) zone-type***

This threshold typology is too rough and imprecise to apply to real world situations. In order to operationalize the concept of ecological thresholds, I now apply both the ecological checklist (confer section 3.3.1.1) and the methodology of self-identity (confer section 3.4.2.1.2) to the concept of ecological thresholds, as only then it is possible to specify and analyze ecological thresholds precisely.

A *threshold analysis* is comprised of four steps and correspondingly of four (interrelated) questions (cf. *Figure 8*). Those are: (1) *what is the major type of the threshold?*, i.e. does it correspond to a critical load, extrinsic factor threshold or regime shift; (2) *what ecological unit does the threshold refer to? (threshold of what?)*, i.e. specification of level of description, variables of interest, reference regime, temporal and spatial scale and the self-identity of the ecological unit of concern; (3) *what disturbance does the threshold refer to? (threshold to what?)*, i.e.

specification of the disturbance regime; and (4) *what is the width in state space of the threshold?*, i.e. is the threshold point- or zone-type. Carried out in this way, a threshold analysis helps to operationalize the concept of ecological thresholds for particular ecological situations.

Ecological threshold

⇩

(1) What major type of threshold?

Critical load, extrinsic factor threshold, regime shift

⇩

(2) Threshold of what?

Level of description, variables of interest, reference regime, spatial and temporal scale, self-identity

⇩

(3) Threshold to what?

Disturbance regime

⇩

(4) Width in state space?

point- or zone-type

Figure 8: Threshold analysis

The main interest of this section is the third major type of ecological thresholds (i.e. regime shifts) because it is most important within resilience research. In the following this section outlines the first two major types of thresholds before examining the third type in detail.[73]

3.2.3.3.2 Critical Loads

The first major type of ecological thresholds refers to critical loads. It is generally understood as *a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge* (Cresser 2000).[74] The concept of critical loads has been criticized for its vagueness (e.g. what are significant harmful effects?) and its simplifying assumptions. The latter is reflected by the fact that critical loads are extremely context-dependent. For example, the absolute critical load value for soil at any particular site would depend upon a number of factors, such as soil parent material, annual precipitation, vegetation type or soil depth. It is thus difficult to make generalisations over different ecosystem types (Cresser 2000).

Despite these problems, critical loads have been used widely in environmental pollution management. For instance, abatement strategies have been set up in Europe to control emissions of air pollutants (see the convention of long-range trans-boundary air pollution (LTRAP) (http://www.unece.org/env/ltrap/) or the critical loads of organophosphates and synthetic pyrethroids in letnic waters (van Wijngaarden et al. 2005). Establishment of critical loads values and exceedance maps throughout Europe has been useful in focussing attention upon acid rain as a political issue. For example, the German government publishes every two years the so-called "Waldschadensbericht", a report about the impact of air and soil pollutants on forest ecosystems.[75]

Some generalisations can be drawn from critical loads research even though these lessons may appear as trivial for experts. For example, upland areas with steep

[73] The second type of threshold - the extrinsic factor threshold – will be highly important for the discussion of resilience mechanisms at a large scale (cf. section 3.3) This is the reason why I consider this type of threshold in more detail in this section.

[74] The concept of critical loads represents an inherently normative and prescriptive, rather than a descriptive concept, as terms such as "pollutants" or "harmful effects" stem from human evaluation.

[75] The *Waldschadensbericht* is an example for the ambivalent character of critical loads. On the one side, this report attracts societal and political attention to the relation of air pollutants and forest ecosystems. On the other side, the report has been criticized heavily for its indiscriminate methodology (Ellenberg 1996).

slopes and thin soils evolved from rocks with low weathering rates were most at risk of acidification, soils evolved from rocks with high weathering rates or being used for agriculture are not at risk at all and lakes with very low natural alkalinity are much more at risk than lakes with high alkalinity (Cresser 2000).

3.2.3.3.3 Extrinsic Factor Thresholds

The second major type of ecological thresholds refers to extrinsic factor thresholds, where changes in a variable at a larger scale alter variables at a small scale. The impacts of habitat loss and fragmentation on biodiversity provide a good example for this type of threshold and there is extensive research in this field of study (Debinski & Holt 2000; Fahrig 2001; Bissonette & Storch 2002; Fahrig 2002, 2003; Ryall & Fahrig 2006).

First of all and importantly, habitat loss and habitat fragmentation have to be distinguished. *Habitat loss* refers to decreasing habitat cover, whereas *habitat fragmentation* points to the breaking apart of habitat, independent of habitat loss (Fahrig 2003).[76] This distinction is important as, in general, habitat loss on the one side and habitat fragmentation on the other have fundamentally distinct effects on species richness and composition.

Habitat loss has generally large negative effects on species richness and composition (Fahrig 2002, 2003). For many species there may be a threshold amount of habitat loss at which the probability of population extinction increases from near-zero to near one following a small additional loss of habitat – the *extinction threshold*. The main factors thought to determine this extinction threshold are reproductive rate of the organism, rate of emigration of the organism from habitat, habitat pattern in the landscape (fragmentation) and matrix quality (survival rate of the organism in non-habitat areas) with the first two factors, i.e. the reproductive rate and the emigration rate, having the largest potential effect (Fahrig 2001).

In terms of models, there is no common threshold value across species. Rather, thresholds range from less than 1% of habitat to over 99% habitat, depending on the

[76] Note however, that this distinction has not been made by the lion's share of the relevant authors. Habitat fragmentation is usually defined as landscape-scale process involving both habitat loss and the breaking apart of habitat. Moreover, different authors measure fragmentation in different ways and, as a consequence, draw different conclusions regarding both the magnitude and direction of effects (Fahrig 2003). This is one reason for the usefulness of the distinction between habitat loss and habitat fragmentation.

abilities of a certain species for reproduction or dispersal, for instance. Species with low reproductive potential and a risky dispersal strategy (high emigration rate and low survival in matrix) require very large amounts of habitat for persistence (Fahrig 2001). Knowledge of extrinsic factor thresholds for certain species may help to define *sensitivity thresholds* for species conservation (Huggett 2005). In addition, habitat loss may have large effects on ecological interactions, such as predator-prey relationships and biological control through natural enemies (With *et al.* 2002; Ryall & Fahrig 2006). Yet these effects depend strongly on the feeding habit of the predator in question (i.e. whether the predator is a specialist, omnivore, or generalist) and whether the matrix supports or does not support predator populations (Ryall & Fahrig 2006).

There is also some *empirical* evidence for extrinsic factor thresholds (Huggett 2005). In this case, thresholds may not refer to extinction thresholds but reveal *demographic isolation thresholds*, i.e. the critical distance beyond which species are unable to "move" between patches. Some workers have found empirical evidence for ecological thresholds in relation to the area of native vegetation cover for gliding marsupials (van der Ree *et al.* 2004) and the white-browed treecreeper (*Climacteris affinis*) (Radford & Bennett 2004) in South-East Australia as well as for ant species in the Amazonian forest in Brazil (Vasconcelos *et al.* 2006). In addition, Kremen *et al.* (2004) found an ecological threshold for crop pollination with respect to habitat loss. Crop pollination services provided by native bee communities in California strongly depended on the proportion of natural upland habitat within 1 – 2.5 km of the farm site, a spatial scale that accords well with maximal foraging distances for bee species. In this case, ecological thresholds are related to the *provision of ecosystem services*. In my view, this is a highly valuable approach.

On the contrary, other workers have not found empirical evidence to support the existence of demographic isolation thresholds in natural and modified systems. For example, no evidence was found for the relation between the amount of native eucalypt vegetation cover and the composition of mammal, reptile or bird communities in South-Eastern Australia. Instead, smooth gradients of responses to vegetation cover patterns existed in the landscape (Lindenmayer & Luck 2005).

In contrast to habitat loss, habitat *fragmentation* – in a narrow sense defined as breaking apart of habitat, independent of habitat loss - generally has much weaker effects on biodiversity (Fahrig 2003). Yet results of modelling studies are inconsistent and show weak to relatively large effects of fragmentation *per se* on biodiversity. There is also some affirmative empirical evidence for thresholds in connectivity of habitat with respect to ecosystem services, i.e. crop pollination and seed dispersal, that were generated by changes in the spatial configuration of patches rather than reduction of area *per se* (Bodin *et al.* 2006). Most of the empirical evidence, however, suggests that the effects of fragmentation *per se* are generally much weaker than the effects of habitat loss and that these effects are at least as likely to be positive as negative (Fahrig 2003).

3.2.3.3.4 Dynamic Regime Thresholds

The third major type of ecological thresholds refers to the analysis of dramatic and surprising shifts of an ecological unit between alternative stable states or regimes, where a small change in a driver causes a marked change in the condition of the unit. This type of threshold is termed in this thesis *dynamic regime threshold*. It is most important within resilience research and a good example is the *abrupt shift between alternative stable regimes at the ecosystem level*. As I have considered in detail in section 3.2.3, evidence for dynamic regime shifts stems from a wide array of ecosystem types, e.g. forests, coral reefs, lakes, savannas, deserts, peatland or arctic tundra. Yet *many* ecosystem types can exist in alternative stable regimes but *not all* (Schröder *et al.* 2005). Indeed, the *relative frequency* of the occurrence of alternative stable regimes across systems is higher for systems controlled by environmental adversity, such as deserts, arctic tundra, intertidal mud flats or savanna.

Resilience research has established a data base for dynamic regime thresholds, which can be browsed or searched at URL: www.resalliance.org. To put the empirical evidence in order, Walker and Meyers (2004) suggest a *typology* of ecological as well as social-ecological dynamic regime thresholds. This typology is comprised of *five classes*, according to whether the regime shift has occurred internally or externally in the ecological, social or linked social-ecological system, and *eleven categories* based on the direction and impact of the interactions between the systems. For example, there are thresholds for a shift in the ecological system that is driven by an environmental event from outside with no impact from society and other thresholds with respect to a shift in the ecological system that is driven by the social system.

Several *components* of a dynamic regime threshold can be distinguished (Briske *et al.* 2006). The first component is the threshold *trigger*. Triggers represent changes in specific biotic or abiotic variables and correspond to disturbances, which may be natural (e.g. fire, storms, droughts) or anthropogenic (e.g. heavy land use, pollution, fire prevention). Further threshold components refer to structural and functional thresholds. *Structural thresholds* are based on changes in relative species and growth form composition, spatial distribution of vegetation and bare soil or the presence of invasive species, whereas *functional thresholds* describe modifications of various ecological processes, such as productivity, nutrient pools or water movement across the landscape. The fourth threshold component are *feedback mechanisms*. Feedback mechanisms represent ecological processes that either reinforce (negative feedbacks) or degrade (positive feedbacks) the resilience of stable regimes. This leads to the discussion of "resilience mechanisms" that are responsible for the maintenance of a certain regime, which will be the topic of section 3.3. A regime shift may thus be interpreted as a shift from a dominance of negative feedbacks to a dominance of positive feedbacks. The rate at which the feedback switch occurs will establish the degree of non-linearity or discontinuity of a threshold.

Furthermore, in order to describe an ecological threshold it is critical to specify the variables along which the threshold occurs, the variables that change as a consequence of the shift, and the factors that have driven the change (Walker & Meyers 2004; Bennett *et al.* 2005a).

There are some further characteristics of dynamic regime thresholds. First, thresholds are not constant, rather the position of a threshold along a determining variable can change. Second, some regime shifts are reversible, some irreversible. It should be a high priority to analyse the system attributes that lead to reversibility. Third, threshold changes on a large scale (e.g. reversal of ocean currents) are more rare and difficult to measure. Most regime shifts occur at the landscape scale (Walker & Meyers 2004).

3.2.3.3.4.1 Stability Landscapes and Threshold Progression

The occurrence of alternative basins of attraction and ecological thresholds have been summarized in the *ball-and-cup metaphor* or *stability landscape*. This is a common graphical aid to communicate basic stability concepts, such as ecological resilience or resistance. No one of course is suggesting that ecological systems behave like a literal ball under the action of gravity in a landscape of hills and valleys and there are in fact several other landscape interpretations to consider ecological dynamics in more detail, e.g. the motion of a particle on an “energy landscape”, the “potential landscape” or the “Lyapunov Function Landscape”.[77] However, the strength of the ball-and-cup analogy lies in its versatility and in the intuition it affords (Pawlowski 2006).

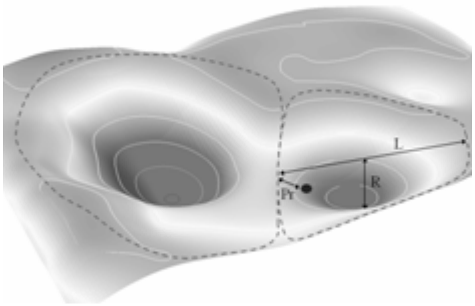

Figure 9: Stability landscape with two alternative basins of attraction and an ecological threshold; L: latitude, R: resistance and P: precariousness; the black dot illustrates the current regime of the ecosystem
from Walker *et al.* 2004

Figure 9 shows a stability landscape for a given ecosystem that exhibits two alternative basins of attraction separated by an ecological threshold. The black dot represents the current regime the ecosystem exhibits at a given moment in time. By using this stability landscape, some authors have specified the meaning of ecological resilience more precisely. Walker *et al.* (2004) suggest that this notion of ecological resilience corresponds to “staying in the same basin of attraction”. Similarly, some authors identify the “size of the basin of attraction” as a measure for resilience (Holling 1973; Scheffer & Carpenter 2003).

This notion of resilience as “staying in the same basin of attraction” can be subdivided into four *aspects* (cf. L, R and Pr in *Figure 9* and the aspect panarchy considered above). Those are: (1) *latitude* (L) defined as the maximum amount the system can be changed before losing its ability to recover which corresponds to the

[77] For more information on these terms confer Pawlowski (2006).

width of the domain of attraction; (2) *resistance* (R) or the ease or difficulty of changing the system which is related to the topology of the domain; and (3) *precariousness* (Pr) defined as the current trajectory of the system, and how close it currently is to a limit or ecological threshold, which, if breached, makes recovery difficult (reversible shift to another regime) or impossible (irreversible shift). The fourth aspect of ecological resilience is dubbed (4) *panarchy* and is related to how the three aspects above are influenced by the regimes of the (sub)systems at scales above and below the scale of interest. In evolved systems that have been subjected to strong selection pressures, the three aspects of ecological resilience have co-developed and are often strongly inter-related. Walker *et al.* (2004) do not believe in or advocate separate measurement of the distinct aspects because of their inter-dependencies. But they do believe that substantive qualitative assessments can be made of each of these aspects of ecological resilience.

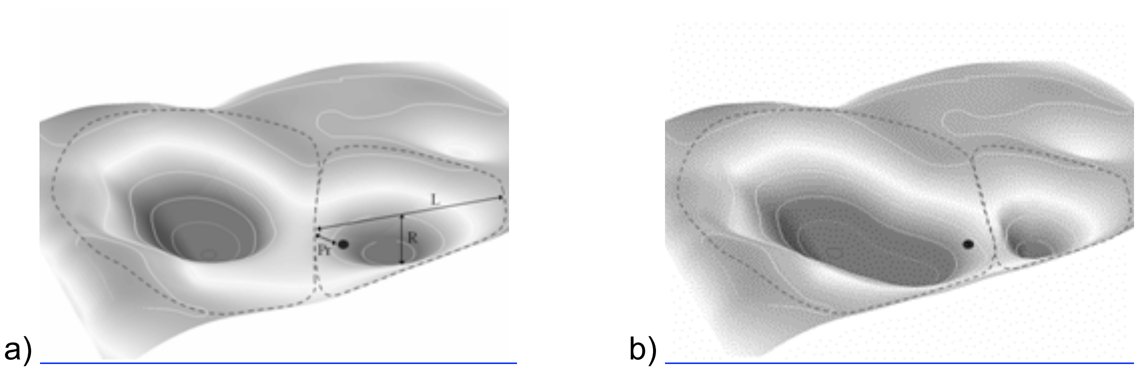

Figure 10: Alteration of stability landscapes. a) Original landscape with two alternative basins of attraction separated by an ecological threshold; b) Altered landscape with an expanded (left side of the figure) and a contracted (right side) basin of attraction. The position of the ecological threshold has changed. Therefore, the ecosystem exists currently in a regime (illustrated by the black dot) in the alternative basin of attraction. from Walker *et al.* (2004)

From the perspective of “resilience theory”, a stability landscape is conceived as dynamic rather than static. Owing to the interactions between slow and fast variables and the alteration of slow variables, it undergoes alterations in the face of exogenous drivers (disturbances) and/or endogenous processes (e.g. plant succession, predator-prey cycles) (Scheffer & Carpenter 2003; Walker *et al.* 2004). Alterations include changes in the number of basins of attraction, changes in the positions of the threshold between basins, or changes in the depths of basins. These changes are often related to a change in slow variables (Walker *et al.* 2004). *Figure 10* illustrates a change in the position of an ecological threshold or the size of a basin of attraction, respectively.

Another issue I want to stress here is that, in the first place ecological thresholds are interpreted more or less as a boundary in state space that exist between two alternative basins of attraction, as shown in *Figure 10*. Yet there are some attempts that give a more exhaustive description of ecological thresholds.

For savannah ecosystems, Briske *et al.* (2006) distinguish between four *threshold categories* within a specific *progression of threshold development*, as illustrated in *Figure 11*. The categories may serve as ecological benchmarks to describe the specific *extent of threshold progression* and include structural, species loss, functional, and property extinction threshold categories. They apply to the progression of a single threshold and specifically reference the fate of the pre-threshold regime after a threshold has been crossed. In other words, the pre-threshold regime does not immediately become extinct when a threshold is crossed and a post-threshold regime begins to dominate the site. For example, residual properties of a grassland community may exist for decades following woodland encroachment and it is these residual ecosystem properties that provide the potential for grassland recovery following woody plant removal (e.g. threshold reversal). Properties of pre-threshold regimes that persist after thresholds have been crossed have been termed *residual pre-threshold properties* (Briske *et al.* 2006). The important point is that threshold categories are anticipated to occur successively, thus they may not necessarily be distinct and substantial overlap may occur among them. The post-threshold regime becomes increasingly dominant and threshold reversal becomes less likely on sites as the residual pre-threshold properties progresses through these various threshold categories.

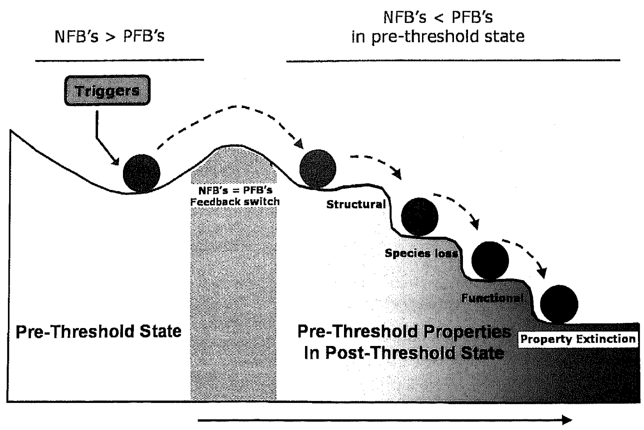

Figure 11: Threshold categories
from Briske et al. (2006: 230)

The first threshold category is termed (*a*) *structural category*. This category is based exclusively on modifications to relative species composition and patterns of species distribution. It is assumed that insufficient time has passed within threshold progression for substantial modification of ecosystem processes to have occurred, even though the switch from negative to positive feedbacks has previously taken place. In most cases, removal of the dominant species from the post-threshold regime is anticipated to reverse the threshold. The second category is dubbed (*b*) *species loss category* and defines the point of threshold progression where insufficient species richness and genetic diversity remain to re-establish the pre-threshold regime even when dominant species of the pre-threshold regime are removed by management prescriptions. It is assumed that ecosystem processes will still support establishment and growth of the pre-threshold dominants, if propogules are introduced to the site. The third threshold category termed (*c*) *functional category* describes the point of threshold progression where positive feedbacks have sufficiently progressed such that ecological processes will no longer support dominants of the pre-threshold regime, even if dominants of the post-threshold regime are removed. Finally, the fourth threshold category is coined (*d*) *property extinction category* and defines the point where residual pre-threshold properties have become extinct and the post-threshold regime completely dominates the site (Briske *et al.* 2006).

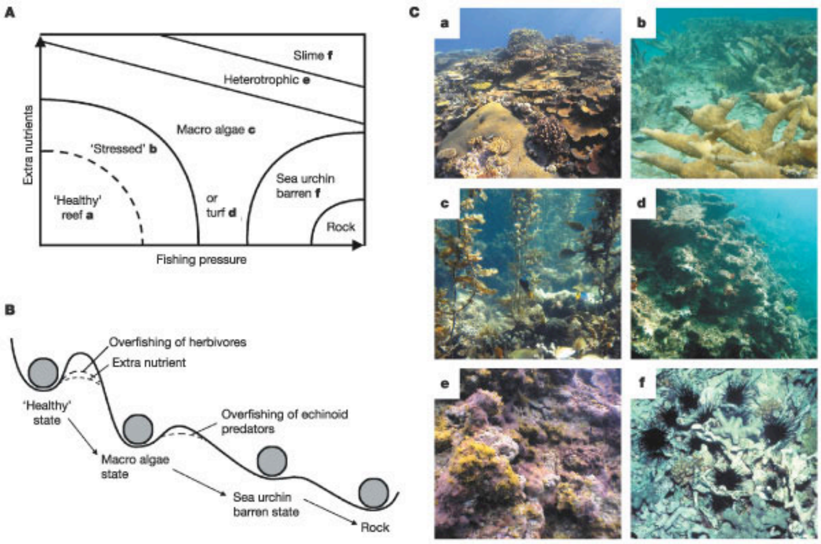

Figure 12: Threshold progression in coral reefs
from Bellwood *et al.* (2004: 828)

Similarly to the threshold categories proposed by Briske *et al.* (2006) for savannah ecosystems, Bellwood *et al.* (2004) suggest a graphic model showing human-induced transitions between alternative regimes for coral reefs, as shown in *Figure 12*. The system shifts progressively from a coral-dominated regime to a macroalgae-dominated regime (due to overfishing of herbivores), via a sea urchin barren regime (owing to overfishing of echinoid predators) to a rocky regime. These different regimes reflect likewise the extent of threshold progression in coral reef ecosystems. Even though the threshold progression metaphor suggested by Briske *et al.* (2006) and Bellwood *et al.* (2004) have only been applied to savannah or coral reef ecosystems, respectively, it may provide an useful tool to understand ecological thresholds in other ecosystem types as well.[78]

[78] Briske *et al.*'s (2006) and Bellwood *et al.*'s (2004) approach to threshold progression may be interpreted as a departure from a strict holistic comprehension of ecosystems, which is typical for the resilience approach (as considered in section 4.2.3.1), to a more individualistic notion of ecosystems. In the former case, ecological thresholds are viewed to separate "holistic regimes" of ecosystems and single species are of no (primary) interest to the researcher. In contrast, in the latter case the threshold is still linked to organic regimes and their "functioning" but also to specific species distributions and abundances. The latter approach to ecological thresholds may thus be regarded as a hybrid form between an organic and an individualistic notion of ecosystems.

3.2.3.3.4.2 Prediction of Threshold Position

Few studies so far have tried to predict the position of an ecological threshold. More often, regime shifts have been observed and reported after the shift has occurred. In some cases, retrospective studies have been conducted where measurements taken before the shift were by chance available and could be compared with data collected after the shift (Walker & Meyers 2004).

However, there are several approaches to *predict the position of ecological thresholds*. Note that the terms *ecological forecast* and *prediction* can be used interchangeably to mean the future probability distribution of an ecological variable, conditional upon initial conditions, parameter distributions, distributions of extrinsic drivers, and the choice of model used to make the calculations (Carpenter 2002). Reliable ecological forecasts would be of high relevance for conservation and environmental management (Clark *et al.* 2001). This is particularly true for regime shifts because these often may be highly undesirable from a societal perspective.

The first approach to predict the position of ecological thresholds proposed by Wissel (1984) refers to the time that an ecosystem needs to return to an equilibrium after disturbance, which corresponds to the stability concept *elasticity* or *engineering resilience* (for the definition of these terms cf. section 4.1.3). Mathematical investigations show that the *characteristic return time* of a system will increase when a slow controlling variable approaches an ecological threshold. This might be a general system behaviour in models, i.e. an ecosystem is more sensitive to disturbances in the neighbourhood of ecological thresholds. This “universal law” examined in the models opens the possibility of predicting the position of an ecological threshold in a real ecosystem. For this purpose one has to determine empirically the characteristic return time for values of the slow variable well distant from the threshold. From this the position of the ecological threshold can be predicted by extrapolation (for more details cf. Wissel 1984). Thus, the general law in models may be used to predict the position of an ecological threshold by extrapolation of empirical data, which are recorded in a safe distance from this threshold.[79]

The second approach to predict the position of ecological thresholds examines the *standard deviations* of certain variables of a system. Carpenter and Brock (2006) use Monte-Carlo simulation to model the eutrophication of a shallow lake from a clear-

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[79] Pointing to a similar insight, Ludwig *et al.* (1997) suggest that long return times may be a diagnostic for low resilience in the vicinity of ecological thresholds.

water regime to a turbid-water regime with respect to water phosphorus. Hereby, the variability of water phosphorus is a leading indicator of shifts between these two regimes. More specifically, the standard deviations (or variance) of total phosphorus density in the water column within the period of summer stratification should increase as a lake approaches a transition from oligotrophic to eutrophic regimes. Carpenter and Brock (2006) suggest that such increases in variability should be a general feature of lakes approaching a regime shift to eutrophication. Surprisingly, the standard deviations of water phosphorus appears to signal threshold crossings a decade or more in advance. Moreover, an observer could detect the changes in standard deviations without knowing the specific mechanisms of the regime shift by using simple, empirical models for ecological time series. For example, studying a shallow lake in Sweden, Hargeby *et al.* (2006) found a long-term oscillation in total organic nitrogen prior to a regime shift from a clear-water regime to a turbid-water regime with a periodicity of eight years.

This feature of shallow lakes may indicate a general characteristic of ecosystems, in that they “stutter” before a regime shift occurs. Carpenter and Brock (2006) expect that increased variability should occur prior to threshold transitions in many ecosystem types. Thus, increasing variability may provide a useful and general indicator of threshold transitions.[80] Should adaptive social responses be evoked, the monitoring of standard deviations can be a highly useful early warning tool for environmental management to sustain essential ecosystem services (Brock & Carpenter 2006).

The third approach proposed by Fath *et al.* (2003) and Mayer *et al.* (2006) refers to *Fisher Information Theory*. *Fisher Information* represents a statistical measure of indeterminacy and can be interpreted as *a measure of the variability in the time the system state spends in the various sections of its steady state trajectory* (Mayer *et al.* 2006). Hereby, shifts between alternative regimes constitute periods of high variability and Fisher Theory is able to identify such transitions between regimes in datasets characterized by a great deal of noise, such as that evident in data collected from field studies of real ecosystems. Indeed, the Fisher Information approach was useful to detect several real-world regime shifts occurring e.g. in the Pacific Ocean

[80] The time window of increased variability will depend on details of the system dynamics and parameters. In particular, it depends on having relatively slow changes in the attractors combined wit relatively rapid relaxation times of a fast variable. In some types of systems, increased variability may occur over a wide zone of conditions near a transition, while in other types of systems the zone of increased variability may be so narrow as to be useless for empirical purposes (Carpenter & Brock 2006).

system, in Sahara and Florida ecosystems or even in the global climate (Mayer *et al.* 2006). Therefore, repeated calculation of the Fisher Information permits the identification of periods of shifts between alternative regimes, as the pattern of variability in the measured data during these time periods is significantly distinct from other time periods. However, this approach requires detailed knowledge of system behaviour, because the less that is known about the system and the number of possible regimes to which the system (or parts of the system) can flip, the less useful Fisher Information becomes, as the interpretation of data is more difficult (Mayer *et al.* 2006).

Apparently and interestingly, all three approaches for predicting the position of ecological thresholds seem to be highly connected, in particular approach two and three. It might be highly valuable to examine the interrelations of these approaches for identifying the position of ecological thresholds.

3.3 Resilience Mechanisms

This section offers a comprehensive analysis of the topic of resilience mechanisms. The term "resilience mechanisms" points to distinct properties and mechanisms that have causal or nomic (Cooper 1998) force in creating ecological resilience at the ecosystem level. "Ecological resilience" as used in this section represents a descriptive *stability* concept. As soon as ecological "stability" is concerned, two major debates in ecological science have to be taken into account. Those are: (1) the classical *diversity-stability debate* and (2) the modern *biodiversity-ecosystem functioning debate*. Terms, concepts and mechanisms of these debates constitute more or less the *lingua franca* for a discussion of resilience mechanisms that cannot be ignored. They also provide insights and theoretical tools that are fundamental for the understanding of ecological "stability".

Surprisingly, the discussion on resilience mechanisms within resilience research is widely detached from the diversity-stability- and the biodiversity-ecosystem functioning debates; there are little cross-references in the relevant literature. This section thus aims at discussing the topic of resilience mechanisms in the light of the diversity-stability and the biodiversity-ecosystem functioning debate. Section 3.3.1 at first explores the state-of-the-art of current research on 'biodiversity and ecosystem functioning'. In section 3.3.2 the results will be applied to the discussion of resilience mechanisms within the resilience approach. Subsequently, section 3.3.3 concludes with some remarks on alternative approaches to resilience mechanisms, the results of the analysis and the relation of the discussion of resilience mechanisms to some of the latest insights within the philosophy of science in ecology.

3.3.1 State of the Art on ‘Biodiversity and Ecosystem Stability’

This section revisits the diversity-stability debate and the discussion on biodiversity and ecosystem functioning. The aim is to picture the state-of-the art of these two fundamental debates in ecology.

3.3.1.1 Diversity-Stability-Debate

The question whether biodiversity begets ecological “stability” has been answered in fundamental differing ways. Three views can be identified corresponding to the chronological development of the debate.[81] The early view until the 1960s held that diversity fosters “stability” (*View 1*). Elton (1958: 145ff) observed that simple communities more easily exhibit destructive population oscillations and invasions, while MacArthur (1955) proposed that the more pathways there are for energy to reach a consumer the less severe is the failure of any one pathway. The mathematical formulation of MacArthur was later falsely proposed as causal relationship between high diversity and “stability” by Hutchinson (1959). Subsequently, Margalef (1968) put this relationship in the larger framework of succession and information theory. Margalef (1968) proposed that highly diverse climax ecosystems have maximal information in controlling their environment being thus “prepared” for further disturbances (Goodman 1975; Trepl 1999). These conclusions were based on either intuitive arguments or loose observations, but lacked a strong theoretical and experimental foundation. The early view, however, became almost universally accepted.

This conventional wisdom was seriously challenged in the late 1960s and early 1970s. The critique was comprised of (*a*) the rejection of early empirical evidence, (*b*) results of mathematical models and (*c*) questioning the relevance of information theory for ecological problems (Goodman 1975; Trepl 1999). Being one of the leading ecologists at this time, May (1972) showed theoretically that the more complex the system the less likely it is to be “stable” (*View 2*). The explanation of this pattern was that the more diversified and the more connected a system the more numerous and the longer pathways along which a perturbation can propagate within

[81] For some good reviews of the diversity-stability debate confer Goodman (1975), Trepl (1999) and Loreau *et al.* (2002a).

the system leading to either its collapse or its explosion (May 1972). Thus, in the 1970s and 1980s the new paradigm emerged that diversity and complexity beget instability, not stability.

Both *view 1* and *view 2* are limited for several reasons. As Pimm (1984) notes, scientist used several different definitions of stability or complexity and various variables of interest. The important point is that “stability” has to be put in quotation marks. It rather represents a meta-category that includes several stability *properties*, such as resilience, resistance or persistence (Grimm & Wissel 1997).[82] Each stability property can be affected by “diversity” in differing ways.

“Diversity”, in turn, has been used in the 1970s and 1980s for (*a*) the number of species and (*b*) evenness within the community (Trepl 1999). Again, each of the components may have fundamentally different effects on specific stability properties. For example, King and Pimm (1983) showed theoretically that biomass “stability” increases if the evenness of the community increases, whereas it is the opposite if there is an increase in the number of species (cf. Trepl 1999).

Thus, the opposing views within the diversity-stability debate may have been mostly due to conceptual vagueness (Pimm 1984). A critical point that nourished the dispute was the ambiguous use of the term “stability”, as considered above. Another critical point is that the stability concepts were applied to different levels of description, such as population, community or ecosystem. For example, Tilman (1996) provided long-term experimental evidence that species richness does stabilize community and ecosystem processes, but not population abundances. This important insight showed that *view 1* and *view 2* are in fact not contradictory but reveal only part of a longer story. Indeed, Elton (1958: 145ff) and MacArthur (1955) focused on the variability of community biomass (i.e. the level of description is the community), whereas May (1972) studied the variability of the biomass of individual species (i.e. the level of description is the species level). Results differed accordingly. Thus, at this point of the debate it became clear that stability statements are only valid for a specific *ecological situation*. This represents a fundamental insight within the debate about ecological “stability” and has been addressed by Grimm and Wissel (1997). These authors have introduced the *ecological checklist for stability statements*. An ecological situation consists of (1) *level of description*, (2) *variable of interest*, (3) *reference dynamic*, (4) *disturbance*, (5) *spatial scale*, and (6) *temporal scale*. If only

[82] For the terminology regarding stability properties confer section 4.1.3.

one of the features of the ecological checklist is changed this results in a new ecological situation and the old stability statement will, in general, no longer be valid (Grimm & Wissel 1997). This corresponds to an *anti-essential perspective on stability statements* (Trepl 1999). Within diversity-stability- and biodiversity-ecosystem functioning-debate almost each of the characteristics mentioned in the ecological checklist lead to confusion. Scientists used different stability concepts or types of disturbances and applied them to different levels of description, variables of interest or scales (Pimm 1984; Tilman 1996; Trepl 1999; Loreau 2000).

Moreover, most of the early studies referring to *view 1* and *view 2* have focused on simplified, artificial systems with deterministic equilibria and ignored much of the potential for compensation of and between species (Loreau *et al.* 2002). *Functional compensation*, however, is in my view one of the mechanism most relevant for the ecological stability of ecosystem processes and is understood in this thesis as the tendency of coexisting, competing species to increase in abundance or replace species should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition (Ernest & Brown 2001b; Vinebrooke *et al.* 2004).

The recognition of the limited applicability of *view 1* and *view 2* lead to "mixed emotions" regarding the diversity-stability hypothesis and some authors claimed a new research program that ought to address crucial topics (e.g. Trepl 1999). This unsatisfying "state of science" paved the way for a new wave of theoretical, experimental, and observational work that developed in the early 1990s (*View 3*). The focus moved from populations, communities and food webs to ecosystems and the interplay between community-level dynamical processes (e.g. functional compensation, facilitation) and ecosystem-level processes (e.g. productivity, nutrient retention) (Loreau *et al.* 2002). *View 3* corresponds likewise to the modern biodiversity-ecosystem function research, which is described in detail in the next section.

3.3.1.2 Biodiversity-Ecosystem Functioning Research

The biodiversity-ecosystem functioning (hereafter: BDEF) hypothesis posits that a reduction in biodiversity will cause a reduction in the amount and "stability" of ecosystem processes. It became widely discussed by ecologists in the early 1990s.

Biodiversity-ecosystem function research strives for two fundamentally distinct goals, namely to identify (*a*) the effects of biodiversity on the amount of specific ecosystem processes and (*b*) the effects of biodiversity on the "stability" of ecosystem processes. The following sections are mainly concerned with the latter topic.

3.3.1.2.1 The Two "Schools"

In the 1990s, some scholars used both resistance to disturbance and return time to equilibrium (i.e. elasticity) as a measure of "stability". Experiments examined artificially assembled communities at a small scale and showed that species-rich plots returned to their pre-drought biomass more quickly than did species-poor plots (Tilman & Downing 1994; Tilman 1996). Most attention was concentrated on the number of species, as this feature is easy to measure (Beierkuhnlein & Jentsch 2005).

The stabilizing effect was viewed to be mostly due to the *portfolio effect*. The portfolio effect holds that independent fluctuations of many individual species may show lower variability in aggregate than fluctuations of any one species, much as a diversified stock portfolio represents a more conservative investment strategy than would any single stock (Srivastava & Vellend 2005). There has been some discussion whether the nature of this effect is biological or rather statistical.[83]

Yet the notion that higher species richness *per se* begets "stability" of ecosystem processes is no consensus at this time. According to Wardle *et al.* (2000), *two schools of thought* divided the scientific community at the beginning of the 21st century differing about how to interpret empirical results.[84] The first school discovered a clear, causal relationship between species richness and (stability of) ecosystem processes (e.g. Naeem *et al.* 1994, 1995; Tilman *et al.* 1996; Naeem & Li 1997; Tilman 1999). In contrast, the second school of thought holds that ecosystem processes are not necessarily driven by species richness *per se*, but rather that the main drivers of ecosystem processes are the key functional traits of the dominant species present and the composition of functional types (e.g. Grime 1997a; Hooper &

[83] Confer the debate between Doak and Tilman (Doak *et al.* 1998; Tilman *et al.* 1998).

[84] The term "school of thought" has been introduced by Wardle *et al.* (2000). Hereby, these authors distinguish between two poles or extreme positions within the biodiversity ecosystem-functioning debate.

Vitousek 1997).[85] The lesson drawn from this debate is that effects of biodiversity on ecosystem processes have to be clearly related to the particular aspects of biodiversity considered in a study, such as species richness, functional diversity, species composition and alike.

3.3.1.2.2 Functional Aspects of Biodiversity

In contrast to a species richness-focused approach, *functional aspects of biodiversity* became a major focus of biodiversity research from the 1990s onwards (the second "school" in the previous section) (Beierkuhnlein & Jentsch 2005). Hereby, the concepts of "functional diversity" and "functional groups" are prevalent. These concepts are slippery, as there are various definitions and measures for both of them. A broad definition views *functional diversity* as the range of those species and organismal traits that influence ecosystem processes and overall functioning. Yet there are several measures of functional diversity, which include (1) the number of functional groups or types (i.e. *functional group richness*); (2) the sum of distances between species in trait space; and (3) the size of the dendrogram required to describe the difference (Petchey & Gaston 2006). Consequently, it is important to examine the relative explanatory power and applicability of the different measures of functional diversity (Petchey *et al.* 2004).

In a like vein, classifications schemes of *functional groups* are plenty.[86] Functional groups order species into clusters according to specific criteria. Yet there are no standardized sets of functional groups that are recognized for the numerous ecosystem types that exist in nature (Bengtsson 1998). Functional groups can be formed at any level of organization and for any sort of "function"[87] (in the case of plants, e.g. phytosociological association, life form, overall morphology, position in the canopy, phenorhythmics or structure of organs such as leaves or roots). Hence, their number is theoretically infinite (Körner 1994). Multiple alternate classifications

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[85] It would be interesting to elucidate the cultural assumptions of these differing schools, e.g. their systems notion (cf. Kirchhoff 2007).

[86] Grouping species by ecological equivalency is a common device for reducing complexity in ecological systems to manageable levels (Thompson *et al.* 2001). Closely related to the concept of functional groups are the concept of guilds, ecological equivalents and niches (Odum 1999: 271ff; Wilson 1999).

[87] The term "function" has many meanings in ecology (Jax 2005). For more details confer section 2.3.

exist in terms of the number of functional groups, the number of species per functional group and the grouping of species (Wright *et al.* 2006).[88]

In fact, it is not clear whether any classification scheme that groups species together can effectively describe the functional diversity of an assemblage. One critical argument refers to considerations of niche theory and what is called *Gause's hypothesis*. According to the *competitive exclusion principle*[89], which has already been proposed by MacArthur and is considered to be a fundamental principle in competition ecology, two complete competitors cannot exist (Krebs 2001: 190; Begon *et al.* 2006: 237ff, 256ff). That means, the number of coexisting species cannot exceed the number of limiting factors or ecological niches. In this model, stable coexistence requires differences between species, which automatically lead to functional complementarity, and hence, "functional groups" (defined as functionally similar species) would not be possible in competitive communities (Loreau 2004).[90]

There are two basic counterarguments. First of all, this view assumes certain theoretical assumptions that are in itself controversial within ecological science. Classical niche theory and more sophisticated Lotka-Volterra models may be replaced or at least supplemented by neutral theory (Hubbell 2001), for instance.

Second, the argument can be rejected for conceptual reasons, as two distinct concepts of the term *niche* are used.[91] On the one side, the concept of ecological redundancy refers to effects on ecosystem processes, thus to the "role" of a species within the community or ecosystem (cf. Jax 2005). This is the classical niche concept proposed by Elton (1927). Opponents of a concept of ecological redundancy use another niche concept, namely the niche concept suggested by Hutchinson (1957). In this interpretation, niche is defined as that space in a n-dimensional hyperspace a species requires for existence. However, the first niche concept is not necessarily related to the second niche concept. Two species might exclude each other by interspecific competition in a given habitat, but might still be redundant with respect to

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[88] Loreau (2004) suggests that neutral coexistence of equivalent competitors (according to neutral theory), non-linear per capita growth rates, and lack of correlation between functional impact and biomass are potential candidates for the presence of functional redundancy in ecosystems. Once more, the question whether niche-theory, assembly-theory or neutral theory are closer to truth cannot be decided here and must remain open, even though the effects of theory would be large at this point.

[89] The competitive exclusion principle was derived from Lotka-Volterra models and Gause's experiments (Krebs 2001).

[90] Confer for a detailed description of the competitive exclusion principle and its interpretation as regards to its systems notion Kirchhoff (2007).

[91] In the relevant literature, several concepts of niche are being used (Trepl 2005: 117ff). For the analysis of Elton's and Hutchinson's niche concepts with respect to their systems notion confer Kirchhoff (2007).

a given ecosystem process. In the following I assume that the concepts of functional groups and ecological redundancy are useful and fruitful.

A further practical difficulty for the classification of functional groups is related to the argument that each species has its own identity in terms of effects on ecosystem processes, for instance. This idea of *species singularity* means that each species is evolutionary and ecologically unique. The classification of functional groups *must* in a sense ignore some variance of species within the functional groups. How much information will be included in the classification system and how much we are willing to neglect will depend on the aim and use in mind (Gindele 1999: 133; Fonseca & Ganade 2001; Chalcraft & Resetarits 2003). All sorts of groupings could be built out of the same set of species by applying different grouping criteria. In addition, by increasing the component resolution it is principally possible to infer that a species did not belong to a specific functional group (Gindele 1999: 134). As Körner notes: "In scaling up from the species to the functional group, we lose the safety of "control" of experimental conditions and enter a field of "less control"" (Körner 1994: 120).

Hence, any grouping of functional groups includes some degree of arbitrariness (Gindele 1999: 134). Grouping species assumes that the traits of importance are discrete rather than continuously distributed among species, the variance in traits is smaller within than between species, and that the same traits are responsible for regulating different ecosystem processes (Wright *et al.* 2006). However, in my view, the arbitrariness in the classification of functional groups loses its force if sharpened by relating the classification to a clearly specified research question. Clearly, the classification scheme is then valid for a specific research question or ideally for a group of research questions (i.e. a validity domain) only.[92]

With the numerous problems in mind *functional groups* can be defined broadly as *a collection of species that perform a similar ecosystem process, irrespective of their taxonomic affinities* (Bellwood *et al.* 2004).

The classification of functional groups involves five main steps: (i) decide which type of functional group is needed; (ii) select the criteria for including species in the study;

[92] The view that the classification of functional groups is useless because there are infinite possibilities to group the species of a community, in my opinion, is partly due to an ontological perspective on ecological units. An epistemological view may be more appropriate here (cf. Jax et al. 1998 and section 2.4). Certainly, functional groups are abstractions that are made by an observer. Functional groups are not real in an ontological sense. Yet the grouping of species can be still useful if related to a specified research question. However, the classification of functional groups is still dependent on the existence of some similarities between species. Otherwise the concept makes no sense. The extent of similarities between species that is necessary to build functional groups is a crucial and controversial point here.

(iii) select which ecosystem processes should be considered; (iv) choosing morphological, physiological or ecological traits that reflect such processes; (v) choosing and applying objective multivariate methods to the species-trait matrix thus produced (Fonseca & Ganade 2001).

Within the relevant literature, several approaches to classify species into functional groups have emerged.[93] A widely-used *a priori* scheme is based on the well-established plant life forms of grasses, non-leguminous plants and forbs. This scheme is criticized strongly, as plant functional traits vary considerably within plant life forms (Walker & Langridge 2002).

An alternative stream of research uses clusters in trait space through multivariate statistics. Functional groups are viewed as sets of species showing either similar effects on ecosystem processes or similar responses to the environment. They thus reflect *functional effect traits* or *functional response traits*, respectively. Accordingly, functional groups are classified with respect to either ecosystem processes, i.e. *functional effect groups*, or responses against disturbances, i.e. *functional response groups* (Diaz & Cabido 2001; Naeem & Wright 2003).

To sum up: within BDEF research, there is a growing consensus that functional diversity is likely to be the component of biodiversity most relevant for ecosystem processes and stability properties (Diaz & Cabido 2001; Naeem & Wright 2003; Petchey & Gaston 2006; Wright *et al.* 2006). In particular, *functional important types of species* are viewed to be a primary concern of further research (Srivastava & Vellend 2005; Cardinale *et al.* 2006; Thompson & Starzomski 2007).

3.3.1.2.3 Glas or Rubber?

Another approach prevalent in the relevant literature on ecological "stability" suggests that due to their divergent characteristics different ecosystem types tend to show *either* resistance *or* elasticity and resilience (in the sense of Grimm and Wissel 1997) (cf. section 4.1.3). For example, redwood forest in California, USA, is highly resistant to fire events but if the forest is burned down subsequent recovery is very slow or even impossible. In contrast, the Californian chaparral – a species-rich bush vegetation - burns down relatively easily but recovers very fast, i.e. its elasticity or resilience is very high (Odum 1999: 39). The hypothesis holds that the two types of

[93] Concer for further literature on the classification of functional groups Hooper *et al.* (2002).

stability exclude each other (i.e. they do not develop at the same time) and that, in general, ecosystems develop resistance under stable and stress-free conditions, as in tropical forests or coral reefs, and resilience or elasticity under changing conditions, as in some tundra or coast ecosystems (Odum 1999: 38f). Metaphorically spoken, ecosystems thus behave either as glass (resistant) or rubber (resilient, elastic) (cf. also Trepl 1999).

3.3.1.2.4 Complementary Approaches to Ecological Stability

From the late 1990s to today, several review papers have been written on the topic of BDEF research providing an overview of current knowledge (McCann 2000; Schwartz *et al.* 2000; Loreau *et al.* 2001; Loreau *et al.* 2002; Hooper *et al.* 2005). Also in the 1990s and at the beginning of the 21st century, another line of research focussed on predator-prey interactions at several trophical levels (e.g. Ives & Hughes 2002). Hereby, scholars examine the mechanisms underlying the "stability" of predator-prey interactions. There has been some effort to unify both streams of research (i.e. BDEF and predator-prey) to gain a more comprehensive framework for the relationship between biodiversity and ecological "stability" and to include several trophic levels (Ives *et al.* 2005; Thebault & Loreau 2006). In addition, there is a productive line of research working on food-web stability (cf. Rooney *et al.* 2006). The authors propose that the stability of complex ecosystems critically depends on the maintenance of the heterogeneity of distinct energy channels, their differential dynamic properties (i.e. the differential productivity and turnover), and the mobile consumers that couple these distinct channels (Rooney *et al.* 2006). Although important and interesting, all these alternative approaches to ecological "stability" are beyond the scope of this thesis. The integration of other lines of research with BDEF research is still in its infancy.

3.3.1.2.5 Ecological "Stability" at the Landscape Scale

A still other line of research that is related to research on biodiversity and ecosystem functioning focuses on the persistence of communities and ecosystems at larger scales. Drawing on the classical *meta-population concept*[94] introduced by Levins, the *meta-community concept* is put forward by authors such as Loreau, Holyoak or Mouquet. A *meta-community* is defined as a set of local communities that are linked by dispersal[95] of multiple potentially interacting species (Wilson 1992; Leibold *et al.* 2004). This concept strives to integrate local and regional processes in the explanation of species coexistence and local species diversity (Loreau and Mouquet 1999). I will treat this concept in some detail here, as it will be important with respect to the issue of resilience mechanisms at the regional scale considered in section 3.3.2.2.

When working on meta-communities scholars use four theoretical frameworks with different historical roots (Chase *et al.* 2005).[96] The first approach, called the (1) *patch-dynamic paradigm*[97], assumes the existence of multiple patches (islands) that undergo both stochastic and deterministic species extinctions (e.g. Yu *et al.* 2001). Dispersal counteracts these extinctions by providing a source of colonization[98] into empty patches. For coexistence to occur, dispersal rates must be limited so that dominant species cannot drive their competitors or prey to regional extinction. Because all patches are identical and there are no permanent refuges for species, it is likely that local within-patch species composition and diversity will change through time.

Alternatively, the (2) *species-sorting paradigm* emphasizes that resource gradients or patch types cause sufficiently strong differences in the local demography of species and the outcomes of local species' interactions that patch quality and dispersal jointly affect local community composition (e.g. Chase & Leibold 2003). This perspective emphasizes spatial niche separation above and beyond spatial dynamics. Dispersal is important because it allows compositional changes to track changes in local environmental conditions. The species-sorting paradigm is closely related to traditional theory on niche separation and coexistence.

The third approach, the (3) *mass-effects* or *source-sink-paradigm* focuses on the effect of immigration and emigration on local population dynamics (e.g. Mouquet &

[94] Confer Hanski (1999a) (1999b) for an overview of the metapopulation concept.

[95] *Dispersal* is defined as "movement of individuals from a site (emigration) to another (immigration) (Holyoak *et al.* 2005: 8).

[96] For an overview of the different frameworks Leibold *et al.* (2004) and Chase *et al.* (2005).

[97] The term "paradigm" is used undiscriminate in this case. The term "theoretical framework", i.e. a specific perspective on and conceptualization of a subject, may be more adequate. Chase *et al.* (2005) rightly speak of "simplified views".

[98] *Colonization* is defined as a "mechanism for spatial dynamics in which populations become established at sites from which they were previously absent" (Holyoak *et al.* 2005: 8).

Loreau 2002; Mouquet *et al.* 2006). Difference in population density (or mass) at different locations, or asymmetric dispersal, can drive both immigration and emigration between local communities. Local extinctions of species may be prevented by immigration from other locations – a process termed *rescue effect*.

Finally, (4) the *neutral paradigm* holds that all species are similar in their competitive ability, movement and fitness (Hubbell 2001). Population interactions among species consist of random walks that alter relative frequencies of species. The dynamics of species diversity are then derived both from probabilities of species loss (extinction, emigration) and gain (immigration, speciation). The neutral paradigm can be regarded as a null hypothesis for the other three views described above.

The four frameworks are built on different assumptions and make divergent predictions (Chase *et al.* 2005). The assumptions of the four models are of at least two types (Leibold *et al.* 2004). First, the models make different assumptions about the nature of differences among local sites. In the case of the patch dynamic and neutral models, the assumptions are that local sites do not differ in any respect except for the species composition that exists at any given moment in time. Alternatively, the mass-effect and species-sorting perspectives assume that there are intrinsic differences among local sites in their attributes so that different species might be favoured at different sites.

Second, these models differ in the assumptions they make about the ecological traits of species involved in the meta-community. In the neutral model, the assumption is that there is no variation in these traits, and consequently no co-variation either. In the patch-dynamics models for competitive meta-communities the assumption is that there is sufficient variation in competitive ability, and that co-variance with dispersal is sufficiently negative to permit regional coexistence. In the mass-effect and species-sorting models, the assumptions are that there are trade-offs in the abilities of species to perform well under different habitat conditions.

However, it is difficult to make a sharp distinction between the four theoretical frameworks. They overlap in their predictions for empirical patterns, and therefore few patterns can definitely differentiate among the models. Furthermore, different systems have shown differential support for each of the frameworks, and thus no single model seems most informative.[99]

[99] For a review of the empirical evidence on meta-communities confer Chase (2005).

The important point is that each of the paradigms illuminates different *aspects* of spatial community dynamics and, thus, may reveal only part of the story.[100] It is unlikely that all of the species interacting in a given meta-community will uniformly conform to any one of these perspectives. Instead, it is likely that each of the sets of processes may play interactive roles in structuring real, i.e. empirically observed meta-communities. Indeed, meta-communities in nature are probable subject to both habitat variability and to local stochastic or non-equilibrium dynamics, limiting the explanatory power of each of the paradigms suggested above. The challenge is to build a more general, synthetic and unified model that incorporates appropriate aspects of each of the paradigms and recognizes that there will be variation among systems (Leibold *et al.* 2004; Chase *et al.* 2005).

The meta-population and the meta-community approaches are extended logically by the meta-ecosystem concept. A *meta-ecosystem* is defined as "a set of ecosystems connected by spatial flows of energy, materials and organisms across ecosystem boundaries" (Loreau *et al.* 2003b: 674). In contrast to the meta-community concept, which only considers connections among systems via the dispersal of organisms, the meta-ecosystem more broadly embraces all kinds of spatial flows among systems, including movements of anorganic nutrients, detritus and living organisms (Loreau *et al.* 2003b, 2005).[101]

3.3.1.3 Six Hypotheses on Ecological "Stability"

Summarizing both the diversity-stability-discussion and the biodiversity ecosystem functioning-debate six[102] hypotheses regarding the relationship of biodiversity and ecological "stability" must be distinguished (Lawton & Brown 1994; Johnson *et al.* 1996; Beierkuhnlein & Jentsch 2005). At first however, it has to be stressed that the hypotheses focus on species richness only, even if graphical representations are often simplified to "biodiversity" (Naeem *et al.* 2002). Indeed, functional diversity,

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[100] The different paradigms can be regarded as (*a*) different but equally valid *perspectives* on the same ecological phenomena. However, there are other possibilities. The different views may be (*b*) culturally determined and not equally valid. Alternatively, (*c*) only one of the paradigms is in fact true. These three options are principally possible and it is hard to decide which notion is true (L. Trepl 2006, *personal communication*).

[101] It represents an important step to integrate the perspective of community, ecosystem and landscape ecology (Loreau *et al.* 2005).

[102] Some authors include a seventh hypothesis, namely the *null hypothesis* on the relationship of species richness and ecosystem processes. The null hypothesis states that there is no relationship between the two (Gindele 1999).

which is not necessarily correlated to species richness, is rarely explicitly addressed (Johnson *et al.* 1996; Beierkuhnlein & Jentsch 2005).

The six hypotheses are formalized by means of graphical illustration, using a measure of "biodiversity" on the x-axis and the "rate of ecosystem processes" on the y-axis. Again, the term "rate of ecosystem processes" used on the y-axis is employed inconsistently within the relevant literature, which is related to the ambiguous use of "function" or "functioning" within ecological science (cf. section 2.3). In what follows I will distinguish between ecosystem processes, e.g. productivity or nutrient retention, and ecosystem "stability" for the sake of clarity.

3.3.1.3.1 Diversity-Stability Hypothesis

The first hypothesis, the *diversity-stability hypothesis* put forward by MacArthur (1955) and Elton (1958) holds that increasing the number of trophical interacting species in an ecological community should increase the collective ability of member populations to maintain their abundances after disturbance.[103] It is usually illustrated as a linear relationship, even though in its original form (MacArthur 1955) the diversity-stability hypothesis did not include assertions of linearity in the effect of species richness and ecosystem processes (Johnson *et al.* 1996). Some authors, however, use this linear hypothesis to address the relationship of species richness and ecosystem processes, which is then termed *linear hypothesis* (Beierkuhnlein & Jentsch 2005). This body of work assumes that each species adds a relatively equal part to the stability of ecosystem processes. The reason for such a pattern is found in the complementarity of species in performing an ecosystem process (Hooper 1998).

3.3.1.3.2 Redundancy Hypothesis

The *redundancy hypothesis* originally proposed by Walker (1992; 1995) states that there are some species that do not contribute remarkably to an increase in neither the rate of ecosystem processes nor ecological "stability". Note that in the original proposition of the hypothesis by Walker (1992), either ecosystem processes or ecosystem "stability" is of concern; this depends on the time scale of the processes considered (Walker 1992). Based on the classification of functional groups, species

[103] This hypthesis corresponds to view 1 in the early diversity-stability debate, as considered in section 3.3.1.1.

have to be identified that perform the same ecosystem process and are thus in a sense "redundant" with respect to this process. The cumulative contribution of species to ecosystem processes or "stability" will then show an asymptotic distribution because the resulting effect should some species go extinct is negligible, as long as all functional groups within the community are maintained. In the following I will explore the work on the concept of ecological redundancy in more detail, because this concept is of high importance for the discussion of resilience mechanisms.

The term "redundancy" is used inconsistently in the relevant literature. In some cases, scholars use "redundancy" for expressing something that is affluent or unnecessary, whereas in other cases scientists use "redundancy" for expressing something that is necessary and valuable. Gindele (1999) exhaustively examines the term "redundancy" and shows its origins in information theory and systems theory as well as some major conceptual problems the term raises.

In information theory, *informational redundancy* ('*informationelle Redundanz*') is defined as that part of an information that does not change its informational content ('*Informationsgehalt*') and could therefore be omitted. Informational redundancy can further be subdivided into conducive redundancy ('*förderliche Redundanz*') and empty redundancy ('*leere Redundanz*'). *Conducive redundancy* signifies constituents of the information that could be left out without diminishing the informational content, yet that can be used to maintain or reconstitute the amount of information, should other constituents of the information be left out after disturbance, for instance. In contrast, *empty redundancy* describes constituents of the information that are inadequate for reconstituting the original informational content. A further distinction in information theory highlights the difference between redundancy and *irrelevance*. Irrelevant information does change the informational content but is not a contribution to the converting of information (Gindele 1999: 128ff).

Within systems theory, the term *structural redundancy* signifies the number and diversity of internal subsystems that have the potential to equitably carry out the same "function". In the engineering sciences, structural redundancy is a common tool to make technical systems more reliable. Thus, this kind of redundancy often transports positive connotations, such as being responsible for the protection against disturbances (Gindele 1999: 131f).

Similarly, in ecology the term "(ecological) redundancy" has two contradictory meanings. The term was introduced by Walker for the purposes of assessing conservation priorities (Walker 1992; Walker 1995), arguing that conservation should focus at first on species that are singular in their contribution to specific ecosystem processes.[104] Objection to the term redundancy arises from the concern that redundancy implies that conservation of redundant species is unnecessary, the unintended corollary of Walker's hypothesis. That is, although Walker (1995) recognized the value of species redundancy, the term itself did not convey this effectively (Naeem 1998). In order to argue for the positive effects of redundancy, Naeem (1998) refers to basic principles from reliability engineering. Similar to structural redundancy within the context of systems theory, as considered above, reliability of a machine is defined as the probability that the machine will perform upon demand. Analogously, in ecological systems *ecological redundancy is valuable because it results in higher probability to maintain certain ecosystem services in the face of disturbances*.

One of the main assumptions of the concept of ecological redundancy is that species can readily be classified into functional groups. I have elaborated on the merits and pitfalls of this concept in some detail in section 3.3.1.2.2. The important point here is, that functional groups are classified according to either (*a*) the effects on a particular ecosystem process (functional effect groups) or (*b*) the effects on responses to disturbance (functional response groups). Using this distinction, by definition, *there is ecological redundancy if more than one species exists within a precisely separated functional effect group*.

Yet the case is not as simple as it appears. There are some major points we have to take into account if we speak of ecological redundant species. First, it is essential to be clear about *what* is meant to remain constant before we can declare redundancy. Suppose a redundant species is lost. Does that mean that there is constancy in the abundance of remaining species, in some measure of ecosystem processes, in plant cover, or solely that remaining species should all remain present? Thus, the concept of redundancy depends upon the purpose of investigation (Gitay *et al.* 1996). Loreau (2004) states that if species loss does not alter function (he probably means: the rate of ecosystem processes), this would correspond to *perfect redundancy*.

[104] Critical voices point to the high uncertainties and impracticability of the concept of ecological redundancy and recommend the preservation of all species for precautionary reasons (Gitay *et al.* 1996).

Second, ecological redundancy is dependent on the interactions between the remaining species. Complete functional redundancy only occurs if, following the removal of a species, there is functional compensation[105] among the remaining species (Walker 1992).[106]

What is the empirical evidence for redundancy within ecological communities? According to Peterson *et al.* (1998), some ecosystems may possess considerable redundancy. For example, Jackson *et al.* (2001) show that the amount of ecological redundancy in some coastal ecosystems decelerated the collapse of these ecosystems due to human exploitation, when compared to other coastal ecosystems. Vinebrooke *et al.* (2003a) provide some evidence that ecological redundancy is high in lower trophic functional groups in boreal lakes in Canada, when faced by experimental acidification, while Kiflawi *et al.* (2006) prove high ecological redundancy in functional groups of reef-associated fish species living on corals in the Red Sea and the Gulf of Aqaba. In addition, Wohl *et al.* (2004) provide some experimental evidence in microcosm suggesting that functionally redundant species may play an integral role to maintain ecosystem processes.

However, ecological redundancy cannot be taken for granted as a general property of functional groups. The degree of ecological redundancy differs across functional groups, trophic levels and ecosystem types. In general, the probability that ecological redundancy is relatively high increases (*a*) *at lower trophic levels*, as there are more species at the same trophic level that could potentially compensate (Klug *et al.* 2000; Chalcraft & Resetarits 2003; Srivastava & Vellend 2005), (*b*) *when resource availability is high* (Beierkuhnlein & Jentsch 2005), and (*c*) *at a small spatial scale*, as stable coexistence at the regional scale requires spatial niche differentiation among habitats but does not at smaller scales (Mouquet & Loreau 2002; Loreau 2004). In addition, it is important to note that species-rich ecosystem types do not necessarily possess a high degree of redundancy within specific functional effect groups. For example, some coral reef ecosystems in the Indo-West Pacific, which have very high species numbers, show very little ecological redundancy in functional groups of bioeroding fish species (Bellwood *et al.* 2003).

[105] For the concept of functional compensation confer section 3.3.1.1.

[106] The concept of redundancy is dependent in part on the degree of expected internal relationships between the populations. If populations are tightly linked, the loss of even a minor species might have strong effects on community structure or ecosystem processes. Again, the degree of expected internal relationships rests on notions of either the organismic or the individualistic paradigm (Krebs 2001: 386ff) (cf. also section 2.4).

Altogether it has become clear that the concept of ecological redundancy is elusive. In conclusion I follow WBGU (2000) and suggest that owing to the many limitations of the concept scientist should formulate more careful statements such as that species x is redundant at a given habitat, for a given process y, for a given temporal scale z.

3.3.1.3.3 Rivet Popper Hypothesis

An alternative hypothesis on the relationship between species richness and ecosystem "stability", the *rivet (popper) hypothesis* (Ehrlich & Ehrlich 1981: preface), suggests that ecosystem "stability" (in this case: resistance) declines when species are deleted, even if the performance of ecosystem processes, e.g. plant productivity, appears outwardly unaffected. This potentially results in sudden shifts to ecosystem regimes that are characterized by an essentially different structure and by different processes. The hypothetic trajectory along a diversity gradient is then more stepwise than linear, as the rivet hypothesis assumes that the amount of ecological redundancy is inherently unpredictable under current and future conditions (Ehrlich & Walker 1998; Gindele 1999: 115).

Originally, the redundancy hypothesis and the rivet popper hypothesis have been regarded as two opposing extreme positions. On the one side, the rivet hypothesis has been interpreted as the notion that each species is "important" for the overall functioning and stability of an ecosystem. On the other side, the redundancy hypothesis has been regarded as the notion that many species could be lost as long as all functional groups are represented by at least one species (Lawton & Brown 1994; Ehrlich & Walker 1998; Gindele 1999: 119).

However, my reading of Ehrlich and Ehrlich (1981) and Walker (1992; 1995) does not suggest this consequence. Similarly to Gindele (1999), I think that the original idea of both authors was quite similar, namely, that the loss of some species may be uncritical to the amount and "stability" of ecosystem processes. To avoid misunderstanding: surely the rivet hypothesis assumes that ecological function is evenly partitioned among species, whereas Walker's analogy of drivers and passengers assumes that there are large differences between drivers that have strong ecological function and passengers that have weak ecological function. But the overall idea that some loss of species richness may be negligible is common to both hypotheses. In this sense, Peterson *et al.* (1998) suggest that both hypotheses can be collapsed into a simple model that can produce specific versions of these models by varying the degree of functional overlap and the degree of variation in ecological role among species.[107]

3.3.1.3.4 Species-Rank Hyothesis

A still other hypothesis, the *species-rank hypothesis*, holds that a small number of abundant species account for a large fraction of ecosystem processes, whereas a large number of rare species account for a large fraction of species richness, but only a small fraction of ecosystem processes. The importance of a species is given by the location of this species in a species rank. The abrupt change in ecosystem structure and processes results from a loss of *dominant species*, which possess the highest biomass and is well-adapted to the given environmental conditions. Should an equal amount of biomass be lost from the ecosystem that is spread over several minor species, the effect on ecosystem structure and processes would be lower (Gindele 1999: 117f).

The species-rank hypothesis is closely related to the *sampling hypothesis*, which points at the lower probability of the occurrence of strongly influential species according to the ecosystem process of interest with decreasing species richness. Only some key species substantially contribute to ecosystem processes and "stability". If the key species that represent important functional groups are preserved there will be no negative effects of species loss on ecosystem processes and "stability" (Beierkuhnlein & Jentsch 2005).

3.3.1.3.5 Insurance Hypothesis

The *insurance hypothesis* hints at the higher probability of flexible functional responses and adaptations to novel environments in species-rich communities. Both the negative temporal co-variance between species abundances and differing responses of species to disturbance result in a reduction in the temporal variance of ecosystem processes (i.e. *buffering effect*) and an increase in the temporal mean of these processes (i.e. *performance-enhancing effect*) (Yachi & Loreau 1999). Loreau

[107] For a detailed analysis of the convergence and divergence of the rivet popper hypothesis and the redundancy hypothesis confer Gindele (1999: 119ff).

explains the mechanisms behind the insurance effect by the following phenomena: the asynchrony of species responses at first increases the range of trait variation available at any time, while a selective process, such as interspecific competition, promotes dominance by species that perform best under the current environmental conditions (Loreau 2000). According to Yachi and Loreau (1999), the importance of insurance effects is determined by three factors. Those are: (i) the way ecosystem productivity is determined by individual species responses to environmental fluctuations, (ii) the degree of asynchronicity of these responses, and (iii) their detailed characteristics including their range of variation.
The insurance hypothesis is closely related to both the redundancy hypothesis and the rivet popper hypothesis. Ecological redundancy and functional compensation are among the fundamental properties ecosystems must possess in order to show the insurance effect against specific disturbances (cf. section 3.3.2.1).

3.3.1.3.6 Idiosyncratic Hypothesis

Finally, the *idiosyncratic hypothesis* implies that it is almost impossible to predict the effects of species loss. There is no linear or non-linear trend but more or less chaotic behaviour of the system (Beierkuhnlein & Jentsch 2005).

3.3.1.4 The Idea of Inequity of Species

The overall idea of the redundancy hypothesis, the rivet hypothesis as well as the species rank hypothesis is the *inequity of the species* present in a community or ecosystem: some species contribute to both the amount and the “stability” of ecosystem processes in a more important way than others. Consequently, ecosystem management and conservation efforts may focus on the most important species rather than on every species present in the community. The idea of inequity of species has been spelled out in differing ways within relevant literature. These different yet potentially overlapping concepts can be used complementary to understand species with a large effect on specific phenomena and processes.
The first concept of inequity among species is the concept of *keystone species*. In its original form (Paine 1969), the concept refers to a species of high trophic status whose activities exert a disproportionate influence on the pattern of species diversity

in a community. Paine's experimental confirmation, in which the removal of the carnivorous starfish (*Pisaster ochraceus)* from intertidal habitat reduced prey species diversity due to intense competition from mussel prey, is now a textbook classic (Krebs 2001: 471; Begon *et al.* 2006: 584). Since Paine's suggestion some authors have argued for the expansion of the concept to include those species whose populations either support or essentially alter the main vegetation pattern of the ecosystem (Bond 1994; Khanina 1998; Bond 2001; Davic 2002; Higdon 2002), while other ecologists have favoured either Paine's (1969) narrow concept or intermediate conceptions, which state that keystone species are species that have a large, disproportionate effect on their community regarding their biomass or abundance (Piraino & Fanelli 1999; Vanclay 1999).

A second concept considers species in their role of engineering their community and environment (Lawton & Jones 1995). *Ecosystem engineers* are species that directly or indirectly modulate the availability of resources to other species, by causing physical state changes in biotic or abiotic materials. *Autogenic engineers* (e.g. corals or trees) change the environment via their own physical structure (i.e. their living and dead tissues), whereas *allogenic engineers* (e.g. woodpeckers and beavers) change the environment by transforming living and nonliving materials from one physical state to another, via mechanical or other means (Lawton & Brown 1994).

The third concept focuses on *dominant species*. Many communities are structured by a few dominant species and apparently these species may be highly important for both ecosystem processes and "stability" (Schwartz *et al.* 2000; Krebs 2001: 474ff). Dominant species are recognized by their numerical abundance or biomass and are usually defined separately for each trophic level. Empirically, dominant species can play a key role in imparting short-term "stability" to ecosystems experiencing environmental disturbance events (Brown *et al.* 2001) or non-random patterns of species loss (Smith & Knapp 2003). This view is unfolded in some detail referring to the *species rank hypothesis* considered in section 3.3.1.3.4.

There may certainly be many other ways that species contribute unequally to ecosystem processes and "stability", as regards to parasite, pathogen or mutualistic species (Ernest & Brown 2001b).[108] Note too, that it is not always easy to separate

[108] A further concept that gives higher status to specific species is the *flagship species concept*. Flagship species are popular, "charismatic" species that serve as conservation symbols and rallying points for the protection of areas, such as the Panda bear (Krebs 2001: 400). I do not regard this concept as equal to the other concepts treated in this section, as flagship species do not necessarily have higher effects on ecosystem processes or "stability" than other species.

each the concept of keystone species, ecosystem engineers and dominant species in a clear manner. The distinction of the concepts depends on the specific conceptualization of the terms. For example, if keystone species are defined broadly (as for example in Khanina 1998 or Bond 2001) then there is large overlap with ecologically dominant species. In this thesis a *keystone species* is defined in an intermediate way as a strongly interacting species whose top-down effect on species diversity and competition is large relative to its biomass dominance within a functional group (Davic 2003), an *ecological engineer* as species that directly or indirectly modulate the availability of resources to other species, by causing physical state changes in biotic or abiotic materials (Lawton & Jones 1995) and an *ecologically dominant species* (or *key species*) as species that provides the majority of the community biomass and is important to ecosystem structure and the functioning of the whole ecosystem (Davic 2002).

3.3.1.5 The Critics of Biodiversity Ecosystem Functioning-Research

Criticism of BDEF-research may be formulated on two different paths. The first path corresponds to a genuinely fundamental, *external* critic. It is argued that BDEF-research is useless and has no meaning at all. According to this view, the intention to investigate the relationship between biodiversity in any form and specific ecosystem processes is useless because there are various measures of biodiversity and potentially infinite ecosystem processes, and hence, an infinite amount of possible combinations. For this reason, BDEF-research is conceived to have no benefit at all, as for any positive relationship with respect to a specific ecosystem process there are various negative relationships with respect to other ecosystem processes *for the same ecological situation* (L. Trepl 2006, *personal communication*).

I do not agree with this critic for the following reason. Even though this critic rightly argues against misplaced generalisations in ecology, I think it still underestimates the limited applicability of the results in BDEF-research. It is not the case that statements in BDEF research or stability statements are meant to be valid with respect to several ecosystem processes. Rather, if a different ecosystem process is concerned this corresponds to a *different ecological situation*. But this fact does not mean that BDEF-research is useless, but rather that the statements are of limited theoretical relevance and have a narrow domain of applicability. This is, in my view, a general feature (and problem) of ecological research and is no good argument against BDEF-research in particular.

As I do not accept this critic, I suggest to follow an alternative path that represents an *immanent* critic of BDEF-research. In my view, BDEF-research is not useless, rather the way it is performed is afflicted with shortcomings. Referring to critical voices in the debate (Goodman 1975; Schwartz *et al.* 2000; Bengtsson 2002; Srivastava 2002; Srivastava & Vellend 2005; Thompson & Starzomski 2007), this view argues that the current practice of BDEF-research is of limited practical relevance for nature conservation and environmental management. Thus, BDEF research has reflected academic concerns more than conservation priorities (Goodman 1975; Srivastava 2002). The immanent critique consists of the following arguments:

First, *results of BDEF research are inconsistent*. Results differ with respect to (*a*) the specific ecosystem process examined. For example within grassland experiments, 74% of studies showed a positive effect of productivity, but only 44% reported a similar effect on decomposition (Srivastava 2002). Results differ with respect to (*b*) the ecosystem type. Only 63% of studies in other systems than grasslands find at least one positive effect. In addition, observations and experimental evidence support the contention that “stability” (here: system variability) responds linearly with species richness in only 50% of cases examined (Schwartz *et al.* 2000). Within a single trophic level, most mathematical models predict saturation of ecosystem processes at a low proportion of local species richness (Schwartz *et al.* 2000). Moreover, the relationship of biodiversity and ecosystem processes varies with (*c*) scale. For instance, biodiversity at local scales typically has strong effects on ecosystem processes, although the opposite is often found at regional scales (Thompson & Starzomski 2007).

Second, *current BDEF-research does not reflect real world species loss*. The assumption of much current BDEF research is that diversity is declining at all spatial scales. As we have considered above, the premise for most of this work has been to examine the consequences of declines in species richness on ecosystem processes. Because of this, many experiments have reduced “natural” or “normal” levels of diversity within species assemblages and then measured the consequences of these reductions on ecosystem processes. This approach woud only be sufficient, however, if most species assemblages declined in diversity. But this is not the case. Surely, global diversity is almost certainly decreasing as the number of extinctions - caused by habitat destruction and pollution, for instance - has greatly exceeded the number of speciation events. By contrast and surprisingly, in many cases – particularly for plants - species diversity at local to regional scales is currently increasing through additions of exotic species that have not been fully offset by extinctions of natives (Sax & Gaines 2003). For example, net species diversity has increased, both on islands (since the arrival of humans within the past few thousand years) and within regions on continents (since the arrival of Europeans within the past centuries) (Sax & Gaines 2003), creating what is termed "novel ecosystems".

In sum, current evidence clearly shows the shortcomings of continuing to focus ecological research exclusively on the effects of decreasing local or regional diversity (as done in BDEF research). Instead future research should also consider the flipside of these same questions, namely, it should address how increases in diversity might affect ecosystem processes (Sax & Gaines 2003). As we have considered above, experiments in BDEF research have examined the impact of species *loss* at local scales. But this is the scale where it is more likely that species *additions* occur due to species invasions.[109] The practical relevance of BDEF research is therefore strongly limited to the rare cases where species loss occurs at local scales. The message is to take into account *real world scenarios* with respect to species extinction and addition.

Third, *species richness is not the main driver of reduced ecosystem processes*. The five top reasons for changes in biodiversity are habitat loss and conversion, climate change, invasive species, pollution and enrichment, and over-harvesting (Sala *et al.* 2000). Each of these may affect ecosystem processes and "stability" directly or indirectly via changes in the biota. Indirect biotic effects can be further divided into effects of composition (including shifts in abundance) and effects of species richness *per se*. If the effects of species richness *per se* on ecosystem processes and "stability" are minimal relative to the other effects, it will be difficult to justify the conservation of species richness on the grounds that reductions will lead to a decrease of ecosystem processes and "stability". For instance, over-harvesting seems likely to show a relatively weak direct effect on ecosystem processes and "stability", while climate change shows a relatively strong effect. Both may show strong indirect effects via composition, but neither show strong indirect effects via

[109] Species invasions is a vibrant field of research in ecology and beyond (Mack *et al.* 2000; Kolar & Lodge 2001; Heger & Trepl 2003).

species richness (Srivastava & Vellend 2005). This is also an important issue that is often neglected within BDEF research.

3.3.1.6 Lessons Learned

The history of the two debates (diversity-stability- and biodiversity-ecosystem functioning-debates), the consensus of BDEF research, its critique as well as the eight hypotheses constitutes what I use as *scientific baseline for the discussion of resilience mechanisms*. Before I will list the lessons learned, in the following I will give some general recommendations for future studies of BDEF-research.

3.3.1.6.1 Recommendations for Future Studies of BDEF-research

First, most of classical equilibrium approaches based on deterministic systems may be inadequate to understand stability properties, such as ecological resilience at the ecosystem level (Loreau *et al.* 2001). They are replaced by a *non-equilibrium ecology* that takes into account the relevance of alternative basins of attraction, disturbance, historical contingency, adaptation at the ecosystem-level, openness as well as heterogeneity (Wallington *et al.* 2005).
Second, there is *no universal relationship of "biodiversity" and "stability"* (Trepl 1999; Schwartz *et al.* 2000; Hooper *et al.* 2005; Thompson & Starzomski 2007). Both terms ("biodiversity" as well as "stability") are slippery, as there is a variety of concepts for both biodiversity (e.g. species richness, functional group richness, regional diversity) and stability (e.g. persistence, resistance, resilience). In addition, the relationship is only valid for a particular ecological situation (for example, the level of description, the variables of interest, temporal and spatial scale). Moreover, results from one ecosystem do not necessarily inform us about other types of systems (Bengtsson *et al.* 2002; Symstad *et al.* 2003). This creates a large matrix of potential combinations of various facets of "biodiversity", distinct stability concepts and potentially infinite ecological situations. Most studies in BDEF research have liberally extrapolated evidence from individual studies to the functional role of species diversity in general (Johnson *et al.* 1996). Yet different types of ecosystems may require different kinds of concepts, measures and experimental design (Loreau *et al.* 2002). As Trepl (1999)

points out there is not a lack of data but a lack of knowledge about the power of explanation or applicability of this data. The potential *domain of resiliency* (Cooper 1998) of the relationship between "biodiversity" and stability concepts such as ecological resilience, that means the cases for which the relationship is valid, its degree of generalization, is in my view, one of the most interesting and pressing questions within BDEF research.

Third, *species richness is only one of the facets of biodiversity that affects ecosystem processes and "stability"*. In fact, other components can be more important, e.g. functional important types of species (e.g. keystone species, ecological engineers, ecologically dominant species), species composition, functional diversity and (the number of) functional groups. A modern approach therefore takes into account the relationship between *the adequate composition of functional attributes* and ecosystem processes or "stability" (Grime 1997b; Bengtsson 1998; Loreau 1998; McCann 2000; Hooper *et al.* 2005; Srivastava & Vellend 2005).

Fourth, most studies have been performed under closed and unrealistic conditions at a small spatial and temporal scale (Schwartz *et al.* 2000; Loreau *et al.* 2001; Diaz *et al.* 2003). Many experiments used artificial species assemblages closed to immigration and propagule supply and containing a single trophic level. It is not at all clear if such studies can be used to inform public and policymakers about the large-scale consequences of biodiversity loss and global change (Bengtsson *et al.* 2002). In contrast, a modern approach evaluates the effects of biodiversity in *a more natural, open and realistic setting at a large temporal and spatial scale* (Bengtsson 2002; Bengtsson *et al.* 2002; Diaz *et al.* 2003; Naeem & Wright 2003; Symstad *et al.* 2003; Cardinale *et al.* 2004; Srivastava & Vellend 2005; France & Duffy 2006; Matthiessen & Hillebrand 2006).

A modern approach uses (*a*) *a more natural and open setting* and includes multiple trophic levels and multitrophic interactions (Schwartz *et al.* 2000; Ives *et al.* 2005; Thebault & Loreau 2006) as well as dispersal, invasion and colonization events (Diaz *et al.* 2003; Naeem & Wright 2003; Symstad *et al.* 2003; Cardinale *et al.* 2004; Srivastava & Vellend 2005; France & Duffy 2006; Matthiessen & Hillebrand 2006).

A modern approach incorporates (*b*) *realistic scenarios of (human-caused) disturbance and species loss or addition*. This means to take into account real world scenarios of human-caused environmental changes through habitat destruction, overexploitation of biological resources, pollution and climate change (Sala et al.

2000), and realistic scenarios of species loss[110] (Solan *et al.* 2004; Zavaleta & Hulvey 2004; Naeem 2006) or *species additions* (Sax & Gaines 2003; Srivastava & Vellend 2005). This implies research on assembly or disassembly rules (Weiher & Keddy 1999; Ostfeld & LoGiudice 2003; Solan *et al.* 2004). For example, the effects of species loss or addition depend strongly on the trophic position of species gained or lost because for example consumers have particular effects on population control and "stability". Changes in species richness at the consumer trophic level alone have very different effects than do simultaneous changes at both plant and herbivore trophic levels (Thebault & Loreau 2006). It is also crucial to acknowledge that at local and regional scales species addition is in fact more likely to occur than species loss (Sax & Gaines 2003). Such "enlightened scenarios" contribute to *ecological realism* within BDEF research (Srivastava & Vellend 2005).

Finally, a modern approach strives for (*c*) *evidence on large temporal and spatial scales*. Most research on stability properties has been performed at small spatial and short temporal scales. When the scale of investigation changes properties of communities and ecosystems do not necessarily change in any coherent fashion (Bengtsson 2002; Bengtsson *et al.* 2002; Cardinale *et al.* 2004). Therefore, evidence on a landscape and long-term scale is critical for the application of ecological resilience to conservation and environmental management issues (Naeem & Wright 2003; Symstad *et al.* 2003; France & Duffy 2006).

3.3.1.6.2 Conceptual Volume

In the previous sections I tried to extract some insights and tools from the diversity-stability-discussion and the biodiversity-ecosystem functioning-debate for applying them to the discussion of resilience mechanisms. These insights and tools lay out a conceptual volume, which is illustrated in *Table 4*. I will use this conceptual volume in the subsequent section to locate the discussion of resilience mechanisms within the classical diversity-stability and the biodiversity-ecosystem functioning-debates.

The conceptual volume in *Table 4* is comprised of the lessons drawn from both the diversity-stability debate and ecosystem functioning debate, which comprise (1) the ambiguous use of terms and the self-identity of ecological units; (2) the ecological

[110] Realistic species loss may be mimicked appropriately by *removal experiments*. For a good overview confer Diaz *et al.* (2003).

checklist; (3) the stability mechanisms proposed in the relevant literature; (4) the six stability hypotheses; (5) the idea of inequity of species; and finally (6) the recommendations for future studies. I will use the numbered insights in the volume in the subsequent sections to align the discussion of resilience mechanisms.

Table 4: Conceptual volume with the lessons drawn from the diversity-stability-debate and biodiversity-ecosystem functioning-discussion

No.	***Insights***	***Description***
1	*Ambiguous use of terms and conceptual clarity*	
1a	Stability is a meta-concept	Stability includes several stability properties: Resistance, resilience, constancy, elasticity, domain of attraction, persistence
1b	Biodiversity is a meta-concept	Biodiversity includes several measures: number of species, species composition, functional diversity
1c	Specify ecological units	Ecological units (population, community, ecosystem) have to be delimited and specified
2	*Limited validity of stability statements*	
2	Apply the ecological checklist to stability statements	Any statement on stability is only valid for a particular ecological situation, characterized by (1) level of description, (2) variable of interest, (3) reference dynamic, (4) disturbance, (5) spatial scale, and (6) temporal scale
3	*"Stability" mechanisms*	
3a	Portfolio effect	independent fluctuations of many individual species may show lower variability in aggregate than fluctuations of any one species
3b	Sampling effect	when there is positive covariance between the competitive ability of a species and its per capita effect on ecosystem processes, the probability of including a dominant, functionally important species will increase with diversity
3c	Functional compensation	the tendency of coexisting, competing species to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition
3d	Facilitative interactions	a positive effect of one species on the functional capability of another
3e	Insurance effect	negative temporal co-variance between species abundances as well as differing responses of species to disturbance result in a reduction in the temporal variance of ecosystem processes (*buffering effect*) and an increase in the temporal mean (*performance-enhancing effect*).
4	*Six "stability" hypotheses*	
4a	Diversity-stability	stability increases linearly with species richness
4b	Rivet	loss of some species may be uncritical unless a critical threshold is reached

4c	Redundancy	loss of some species may be uncritical unless a critical threshold is reached
4d	Species rank hypothesis	a small number of abundant species are critical
4e	Insurance	species richness provides a buffer against human disturbances and environmental fluctuations
4f	Idiosyncratic	no relationship identifiable
5	*Inequity of species*	
5a	Keystone species	a strongly interacting species whose top-down effect on species diversity and competition is large relative to its biomass dominance within a functional group
5b	Ecological engineers	species that directly or indirectly modulate the availability of resources to other species, by causing physical state changes in biotic or abiotic materials
5c	Umbrella species	species with such demanding habitat requirements and large area requirements that saving it will automatically save many other species
	Dominant species	species that provides the majority of the community biomass and is important to ecosystem structure and function
6	*Recommendations*	
6a	Non-equilibrium ecology	stability properties have to be understood within a non-equilibrium paradigm
6b	No universal relationship of “biodiversity” and “stability”	there is a large matrix of possible statements about the relationship of distinct facets of “biodiversity” (e.g. species richness, community composition, functional diversity), various stability properties (e.g. resistance, resilience, elasticity) and different ecological situations
6c	Functional approach complementary to species-richness approach	there is a growing consensus that functional diversity is likely to be the component of biodiversity most relevant for ecosystem processes and stability properties. In particular, functional important types of species are viewed as a primary concern of further research
6d	Future studies	
6di	open and “natural” setting	include multiple trophic levels and multitrophic interactions as well as dispersal and invasion/ colonization events
6dii	realistic scenarios of species addition or loss	take into account real world scenarios of human-caused environmental changes and realistic scenarios of species loss or *species additions* comprising assembly or disassembly rules
6diii	large temporal and spatial scales	generate evidence on a landscape and long-term scale, which is critical for the application of ecosystem resilience to conservation and environmental management issues

3.3.2 Application to the Discussion on Resilience Mechanisms

This section applies the results and tools of the previous sections to the discussion of resilience mechanisms. These mechanisms constitute the "capacity" or "ability" of ecological systems to show ecological resilience to a given disturbance regime.[111] Resilience mechanisms emerge from several facets of an ecosystem, including (*a*) *biotic elements*, such as species richness, functional group richness or ecological redundancy, (*b*) *abiotic factors*, e.g. soil properties or nutrient storages, and finally, (*c*) *disturbance regimes*, which comprise both biotic and abiotic elements. In what follows section 3.3.2.1 and section 3.3.2.2 provide a closer look on the biotic components of resilience mechanisms, while section 3.3.2.3 focusses on the importance of two other factors: abiotic conditions and disturbances.

Before turning to the resilience mechanisms in detail some words on the specific terminology are in order. In the subsequent sections I distinguish between the *local scale* and the *regional scale*. These categories only refer to spatial characteristics, but neither to organisational levels, such as "population", "community" or "ecosystem" nor to any hierarchical theory. That means for instance, it is principally possible to examine an ecosystem at the local scale *and* at the regional scale, depending on the specific spatial or functional delimitation and specification of an ecosystem.

3.3.2.1 Ecological Resilience and Biodiversity at the Local Scale

This section delineates resilience mechanisms at the local scale. Here, the *local scale* corresponds to relatively large areas from 1 – 10 km^2 and may be defined – following Leibold *et al.* (2004) - as locality capable of holding a local and clearly specified community.[112] Numerous properties and mechanisms at several levels of organization (e.g. population, community, ecosystem) are taken into account.

First, ecological resilience is dependent on *population diversity*. Population diversity is comprised of (*a*) *population richness*, i.e. the number of populations of a species in a given area, (*b*) *population size*, i.e. the number of individuals per population, (*c*) *spatial distribution* of the population under study, i.e. the extent of the populations in

[111] This conception of ecological resilience as a capacity or capability of ecosystems assumes a holistic view on ecological units (cf. section 2.4. and 4.2.3.1).

[112] The demarcation of such a community can be highly problematic. On the demarcation of ecological units confer Jax (2002).

a defined area as well as population dispersion and (*d*) *genetic diversity* within and among populations. All these factors may contribute to increased ecological resilience in the face of environmental change (Luck *et al.* 2003). Particularly for ecosystems that are species-poor and thus have little ecological redundancy at the species level, such as salt marshes, reed stands, kelp beds or seagras meadows, genetic diversity can be important to buffer against natural disturbances (Reusch *et al.* 2005). However, some authors propose that while it may be important in some cases, the decline in genetic diversity within species is of relatively low priority compared to the problem of declining populations and the widespread loss of species and habitats (e.g. Holling *et al.* 1995).

Second, ecological resilience is dependent on *the maintenance of functional important types of species and their specific (life-history) traits*. It has been long known that some species reveal a stronger influence on ecosystem "stability" than others (cf. section 3.3.1.4). For example, the loss of top predators can increase the vulnerability of aquatic ecosystems to eutrophication or may "flip" the whole ecosystem to another stable regime (Folke *et al.* 2004), while the invasion of a pocket mouse species in the Chihuahuan desert in the USA had profound effects, due to the impact of the pocket mouse as dominant seed consumer (Ernest & Brown 2001a). Thus, the management of single but salient, important species, such as keystone species (Davic 2003), ecological engineers (Lawton & Jones 1995) and dominant species (Davic 2002) may have important effects on the whole ecosystem. This is in line with some of the findings of BDEF-research (cf. 5 in the conceptual volume in section 3.3.1.6.2).

In addition, the life-history traits of these functionally important types of species are of high concern for the ecological resilience of the whole community. Important traits include high reproductive rate, the emigration rate, high mobility, dispersal ability, phenotypic plasticity, flexible feeding behaviour or physiological tolerance (Grimm *et al.* 1999; Fahrig 2001; Hooper *et al.* 2002; Bulleri & Benedetti-Cecchi 2006). For example, both the reproductive rate and the emigration rate are the most important factors for a species when faced with habitat loss (Fahrig 2001).

This notion corresponds to the *idea of inequity among species* with respect to ecosystem "stability" proposed in the biodiversity ecosystem-functioning discourse, as considered in section 3.3.1.4. In my view, the emphasis on the idea of inequity among species within resilience discourse is the logical consequence of one of the central assumptions of resilience theory, namely the “key variables”- or “rule of hand”-model for ecosystem dynamics. According to this meta-model, ecosystem dynamics can be understood by analyzing a small amount of variables (Walker *et al.* 2006). These variables establish a persistent template upon which a host of other variables exercise their influence (cf. for a detailed treatment section 3.2.2).

Following this notion, a limited number of species controls the critical processes necessary for overall ecosystem functioning. These *keystone process species* are defined as species that provide the essential structure and processes within ecosystem dynamics.[113] Such species modify, maintain and create habitats (Holling *et al.* 1995; Folke *et al.* 1996). From a practical perspective, it is necessary to focus on these keystone process species in order to maintain essential ecosystem services (Holling *et al.* 1995).[114] Clearly, this notion refers to the idea of inequity among species.

Within resilience discourse, both the notions of keystone process species and of inequity among species is reflected by the *driver-and-passenger analogy*[115] proposed by Walker (1992). The *drivers* occurring in a community contribute to the amount and “stability” of ecosystem processes in a more profound way than the *passengers*. Removing the former causes a cascade effect, but loss of the passengers leads to little change in the rest of the ecosystem. In fact, there are really three categories of species: (1) the drivers, (2) the true passengers, and (3) the *potential drivers*, i.e. passengers that currently exert little influence but that could become drivers if the system was radically altered by disturbance or new boundary conditions, such as climate change (Walker 1995).[116] Ecological engineers or keystone species represent

[113] Keystone process species do not correspond to keystone species (cf. section 3.3.1.4). The concept of keystone process species is derived from the key variables- or rule of hand-model proposed within resilience research (cf. section 3.2.2) and signifies species that perform these key variables, whereas the concept of keystone species has different conceptual roots and a different meaning within relevant literature.

[114] In many cases, however, the focus on single species may be oversimplified. For example, the interactions between the two keystone species in the boreal forest in Alaska, namely Pacific salmon (*Oncorhynchus spp.*) and brown bear (*Ursus arctos*) vary in time and space and have fundamental effects on ecosystem structure and processes. Thus, these *keystone interactions* should also be taken into account (Helfield & Naiman 2006).

[115] The value of analogies referring to machines, e.g. cars, as in Walker’s driver-and-passenger analogy are in general controversial within ecological science. On the one hand, it seems plausible that ecological systems just do not behave as simple as these analogies suggest. Furthermore, to understand the purpose of a machine is always straightforward, as these machines are constructed especially for that purpose (Mutschler 2002). In contrast, it is hard if not impossible to imagine such a purpose for ecological systems (Gindele 1999: 137). What is the proper functioning of an ecological system? (Yet the purpose of an ecological system may be set by an observer in terms of the provisioning of ecosystem services, for instance.) On the other hand, analogies may have (*a*) great heurisitic value, (*b*) appeal intellectual thought, and (*c*) may be an useful tool to reduce complexity.

[116] The concept of “potential drivers” is related to the insurance effect, and thus to the concepts of ecological redundancy and functional compensation, as considered below.

typical drivers within a community (Peterson *et al.* 1998). Note that this view on the inequity among species is closely related to the *species rank hypothesis* and *redundancy hypothesis* considered above (cf. 4 in the conceptual volume and section 3.3.1.4).

Third, ecological resilience is dependent on *the amount of ecological redundancy and response diversity within functional effect groups*. The concept of redundancy is prone to uncertainties and dependent on ecological context and the purpose of investigation (Gitay *et al.* 1996; Fonseca & Ganade 2001) (for a detailed consideration of the concept of redundancy confer section 3.3.1.3.2). In addition, it assumes that species can be classified into functional groups (cf. for a detailed description of the concept of functional groups section 3.3.1.2.2). The important insight with respect to functional groups for the discussion here is the distinction between *functional effect groups* and *functional response groups*. By definition, there is ecological redundancy if more than one species exists within a precisely separated functional *effect* group.

Within discourse on resilience mechanisms, ecological redundancy represents something that is valuable. It is conducive for the maintenance of ecological resilience because lost species are replaced by redundant species that continue to perform a given ecosystem process (Walker 1995; Naeem 1998). Thus, the concept is used in analogy to *conducive redundancy* applied in information theory or *structural redundancy* from systems theory. In this sense, redundancy increases the reliability of ecological systems to provide desired ecosystem services in the long run (Naeem 1998).[117]

Theoretically, redundant species may reside either at the same scale (e.g. nitrogen fixation by several plant species) or across different scales (e.g. seed dispersal by rodents and tapirs) (Peterson *et al.* 1998; Elmqvist *et al.* 2003). Empirically, ecological redundancy increasingly occurs (*a*) at lower trophic levels, (*b*) when resource availability is high and (*c*) at a small spatial scale (Klug *et al.* 2000; Loreau 2004; Beierkuhnlein & Jentsch 2005; Srivastava & Vellend 2005).

By definition, redundancy occurring at the same scale fosters *within-scale resilience* and complements redundancy occurring across different scales, which is termed *across-scale resilience* (Peterson *et al.* 1998). According to the resilience approach, ecosystem processes that are replicated across a range of scales can withstand a

[117] This is termed *process redundancy* by Brookes *et al.* (2005).

variety of disturbances. The sum of within-scale resilience and across-scale resilience is dubbed *imbricated resilience* (Holling *et al.* 2002b). The important point is that imbricated resilience derives from overlapping "function" (i.e. the role of species for a given ecosystem process) within scales and reinforcement of "function" across scales as a product of functional diversity and ecological redundancy (Peterson *et al.* 1998; Holling *et al.* 2002b).

Yet the amount of ecological redundancy alone is a poor measure for ecological resilience. Specific disturbance events may delete whole functional groups of species containing a high amount of ecological redundancy if these species show an identical response to a given disturbance regime. Thus, if ecological resilience ought to be a measure of the amount of disturbance an ecosystem can absorb before shifting to an essentially different state, another ecosystem property must be taken into account. Indeed, ecological resilience increases if *the ecologically redundant species within a functional effect group show different responses to disturbance events*, i.e. if they belong to different functional response groups. This property is termed *response diversity* (Hooper *et al.* 2002; Elmqvist *et al.* 2003; Nyström 2006).[118] The term response diversity[119] dates back to studies made by Ellenberg and runs through the diversity-stability- and the biodiversity-ecosystem functioning-debates (Woodward 1994; Tilman 1996; Trepl 1999; McCann 2000; Hooper *et al.* 2002; Hughes *et al.* 2002; Ives & Hughes 2002; Loreau *et al.* 2003a; Symstad *et al.* 2003) up to recent contributions (Hooper *et al.* 2005; Srivastava & Vellend 2005).

According to this line of reasoning, the asynchrony of species responses within the same functional effect group increases its possibility to perform in the long term. This property is termed *temporal niche complementarity* (Loreau 2000). After disturbance, species of a community with low abundance (*minor species*) may replace species showing large abundance (*major species*). There is some empirical evidence for response diversity in functional effect groups found in groups of plants in rangelands (Walker *et al.* 1999), seed dispersers on islands in Western Polynesia (e.g. flying foxes) (Elmqvist *et al.* 2003), freshwater detritivores in lakes (Schindler 1990)

[118] In this sense, Folke *et al.* (2004) distinguish between *real redundancy*, i.e. redundancy that includes response diversity, and *apparent redundancy*, i.e. redundancy that does not incorporate any response diversity. I do not follow their terminology in the following.

[119] Elmqvist et al. define *response diversity* as "the diversity of responses to environmental change among species that contribute to the same ecosystem function" (Elmqvist *et al.* 2003: 488). I do not follow this definition because of the undiscriminating use of the term "ecosystem function". Response diversity is not an entirely new concept. It has been used in several other biological disciplines, e.g. neurology or oncology (Elmqvist *et al.* 2003).

(Carpenter & Cottingham 1997), zooxanthellae, corals and herbivores in coral reefs (Nyström 2006) and in microcosms (Petchey *et al.* 1999) (cf. for a review Elmqvist *et al.* 2003).[120] Note that response diversity may be enhanced by *regeneration traits*. For plants, regeneration traits include seed size, number of seeds per plant, dispersal mode or pollination mode (Hooper *et al.* 2002).

The important point is that a high degree of redundancy and response diversity within functional effect groups together act as internal insurance in carrying out ecosystem processes and foster ecological resilience against specific disturbances (Yachi & Loreau 1999; Hooper *et al.* 2005). This *insurance effect* of biodiversity is closely related to the fundamental stability mechanisms termed *functional compensation*, considered below. Functional compensation is highly dependent on the amount of ecological redundancy within a community. As a consequence, it is more probable at lower trophic levels because these groups frequently have relatively faster growth rates, wider dispersal potentials, and greater species diversity, which results in a higher probability of the presence of tolerant species (Vinebrooke *et al.* 2003b).

The reader might notice a tension between two poles within the discourse on ecological "stability" in general, and resilience mechanisms in particular. On the one side, authors put an emphasis on the idea of inequity among species, as especially the redundancy and the species rank hypothesis are put forward. Thus, functional important types of species are regarded as essential for the maintenance of ecological "stability". On the other side there is an emphasis on the insurance hypothesis, which is primarily attributed to species-rich ecosystems. But, in my view, there is no real tension between these two poles, as the insurance effect is not so much dependent on species richness *per se* but rather on the amount of response diversity within functional effect groups, and thus, on the identity of certain species. Clearly, this is a "functional" approach to the relationship between biodiversity and ecological "stability". Species richness *per se* is *not* sufficient to maintain the ecological resilience of ecosystems to withstand specific disturbance events (cf. 6c in the conceptual volume and section 3.3.1.6.1).

There is another property essential for ecological resilience. Fourth, ecological resilience is dependent on *the maintenance of critical functional groups*. Classifying species into functional groups and weighing them in terms of their importance is a difficult yet promising task (cf. for a detailed treatment of functional groups section

[120] It may be possible to analyze the degree of response diversity through non-random removal experiments, as suggested by Diaz *et al.* (2003).

3.3.1.2.2). A weighing system for functional groups according to their importance for the overall performance, i.e. the functioning of the system requires (*a*) knowledge of the interaction strengths between all pairs of species in a community (and this knowledge is unavailable for most, if not all, natural assemblages) (Micheli & Halpern 2005) and (*b*) a clear definition and specification of the study system. However, some functional groups may be more important than others, because their loss results in serious degradation of the whole functioning of an ecosystem or even in a shift to another stable regime, while the loss of other groups does not have such consequences (Bellwood *et al.* 2004; Micheli & Halpern 2005).

In this thesis I define a *critical functional group* as a functional effect group that is represented by one or a few species only, i.e. if the amount of ecological redundancy within a functional effect group is very low (Walker 1992; Ehrlich & Walker 1998; Bellwood *et al.* 2004). There is some empirical evidence for critical functional groups. For example in coral reef ecosystems, reef-building species, e.g. the staghorn (*Acropora cervicornis*) and the tabular coral (*Acropora palmata*) as well as bioeroding species - by foremost fish species that directly remove algae or seaweed from corals - represent critical functional groups with respect to the ecological resilience of these reefs (Bellwood *et al.* 2004). Similarly in the whole-lake experiments conducted by Schindler and colleagues (1990), the earliest serious changes in ecosystems and food webs occurred when acidification eliminated acid-sensitive organisms that were also the sole occupants of *key ecological niches*[121]. Drawing on a larger research project on the functional effects of biodiversity launched by the Scientific Commitee on Problems of the Environment (SCOPE) and the Global Biodiversity Assessment (GBA), Mooney *et al.* (1995) conclude that sensitive ecological systems *all* have low representation of key functional types, i.e. little functional redundancy within functional effect groups (cf. also WBGU 2000).

Evidence from models predict that the extinction of a functional group is more likely if communities are (*i*) species poor, (*ii*) functionally rich, and (*iii*) have species distributed unevenly across functional groups (Fonseca & Ganade 2001). In addition, groups may show differential sensitivities to a disturbance event depending on their species turnover rates, dispersal potentials and the ecological history of the ecosystem (Vinebrooke *et al.* 2003b).

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[121] The term “key ecological niches” is used as a synonym for “critical functional groups”.

To sum up: ecological resilience at the local scale is fostered by the maintenance of (*a*) population diversity, (*b*) functional important types of species, a relatively high degree of (*c*) ecological redundancy and of (*d*) response diversity within functional effect groups, and (*e*) the maintenance of critical functional groups. Each feature refers to a particular property of an ecosystem that is conducive for ecological resilience. But what is the genuine mechanism responsible for the maintenance of ecosystem processes in the face of change and specific disturbances?

As considered shortly in section 3.3.1.1, this fundamental stability mechanism is termed *functional compensation*. Functional compensation is the tendency of coexisting, competing species to increase in abundance or replace species should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition. From the perspective of competition theory, it is the result of interspecific competition and subsequent increase in abundance of those species that are better adapted to a given disturbance regime (Tilman 1999). A comprehensive concept of functional compensation is comprised of several components. Those are: asynchronous population fluctuations, species replacement, phenotypic variance within functional groups and evolutionary adaptation (McCann 2000; Norberg *et al.* 2001; Symstad & Tilman 2001; Luck *et al.* 2003; Reusch *et al.* 2005). The capacity for functional compensation differs in terms of trophic level. It may be small for extinctions occurring at higher trophic levels, simply because there are few species at the same trophic level that could potentially compensate (Srivastava & Vellend 2005). It is important to note, however, that functional compensation cannot be taken for granted as the mechanism underlying "stability" of ecosystem processes since other mechanisms may be critical as well. In fact, explicit demonstration of compensation among species requires careful experimental control (Hooper *et al.* 2005).

Thus, each resilient-conducive property of ecosystems (e.g. population diversity, ecological redundancy, response diversity) contributes in a particular way to the functional compensation as regards to specific ecosystem processes. Hence, fifth, *ecological resilience is dependent on the potential for local functional compensation*.

3.3.2.2 Ecological Resilience and Biodiversity at the Regional Scale

This section describes resilience mechanisms at the regional scale. Hereby, the *regional scale* corresponds to areas from 10 – 100 km^2. A focus on larger scales is in line with a modern view on the relationship of biodiversity and ecosystem "stability" (Bengtsson 2002; Bengtsson *et al.* 2002; Naeem & Wright 2003; Symstad *et al.* 2003; Cardinale *et al.* 2004; France & Duffy 2006), as considered in section 3.3.1.6.1 and in 6diii in the conceptual volume in section 3.3.1.6.2.

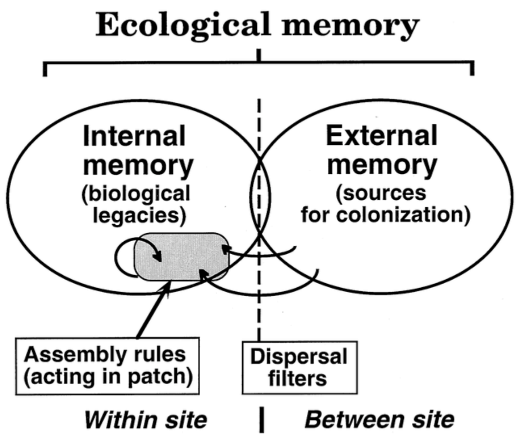

Figure 13: The concept of ecological memory
from Bengtsson *et al.* 2003: 391

At the regional scale, resilience mechanisms refer to the concept of ecological memory. Ecological memory is defined within resilience research as a "network of species, their dynamic interactions between each other and the environment, and the combination of structures that make reorganization after disturbance possible" (Bengtsson *et al.* 2003: 389) or slightly different as "the composition and distribution of organisms and their interactions in space and time (…) [, which] includes the life-history experience with environmental fluctuations" (Nyström & Folke 2001: 407). Yet these definitions provided within resilience research are rather vague, as they are prescriptive rather than descriptive. I understand ecological memory in this thesis as a meta-concept for resilience mechanisms at a landscape scale. Note that the property ecological resilience at a larger scale is also dubbed *spatial resilience*.

As illustrated in *Figure 13*, ecological memory is comprised of at least three interacting components: (*a*) *internal memory* (or *within-patch memory*), (*b*) *external memory* and (*c*) the processes that occur between the patches across scales (Bengtsson *et al.* 2003).[122] At first, this section will provide some insights on internal memory. Subsequently, I will take a closer look at external memory including the processes that occur between the patches across scales.

Franklin and MacMahon (2000) showed that the re-establishment of new ecosystem structure and processes at Mount St. Helen, which experienced a sudden and destructive eruptive episode on the 18th May in 1980, was highly enhanced by organisms and organic structures that survive a disturbance event. Re-establishment was most rapid on the numerous sites that had high concentrations of surviving organisms. These *biological legacies* served as foci for re-establishment (e.g. organisms that survive the disturbance event) and allowed species to colonize (e.g. structural legacies that provide critical protective cover, food or nutrient sources, such as snags and logs in dead tree stumps) (Franklin & MacMahon 2000).[123] Similarly, reorganization in tropical forest after cyclone and fire disturbances is highly facilitated by living remnant tree species (Elmqvist *et al.* 2002).[124] Within resilience research, the remaining biological legacies in a disturbed area are termed *internal memory* (Nyström & Folke 2001; Bengtsson *et al.* 2003). Thus, sixth, ecological resilience is dependent on *the concentrations of biological legacies within the disturbed area*.[125]

By contrast, external memory signifies source areas of colonizing species in the vicinity of the disturbed patch (Nyström & Folke 2001; Bengtsson *et al.* 2003). This concept refers to meta-community and meta-ecosystem dynamics considered in section 3.3.1.2.2. A *meta-community* is defined as a set of local communities that are linked by dispersal of multiple potentially interacting species (Wilson 1992; Leibold *et al.* 2004) and a *meta-ecosystem* as a set of ecosystems connected by spatial flows of energy, materials and organisms across ecosystem boundaries (Loreau *et al.* 2003b). When working on meta-communities scholars use four paradigms (Leibold *et*

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[122] The distinction between internal and external memory has also been made with respect to engineering resilience or elasticity by Grubb and Hopkins (1986). The authors distinguish *in-situ resilience* from *resilience by migration* (Grubb & Hopkins 1986).

[123] Bengtsson (2002) dubs the conducive effect of biological legacies *in situ regeneration*.

[124] The resilience approach distinguishes between "recovery" and "reorganization". According to resilience research, the concept of recovery rather refers to larger scales and is related to the stability concept of elasticity (or engineering resilience), whereas the concept or reorganization rather refers to smaller scales and is considered to be an essential component of the concept of ecological resilience (e.g. Folke 2006).

[125] In this section, it becomes again clear that ecological resilience is an "amphibian" stability concept. It includes fundamentally distinct stability properties such as resistance and resilience. It is not clear to me if this is a weakness or strength of the concept.

al. 2004). Those are: (1) the *patch-dynamic paradigm*, (2) the *species-sorting paradigm*, (3) the *mass-effects* or *source-sink paradigm* and (4) the *neutral paradigm*. As I have considered in section 3.3.1.2.2, each of these paradigms has distinct implications, as they assign a different role to colonization events, species traits or environmental heterogeneity, for instance.

In the relevant literature on resilience mechanisms that refers to a regional scale, in general, a network of source habitats in the surrounding landscape *per se* is viewed to be conducive for ecological resilience. The reestablishment of a disturbed individual habitat will depend on species colonization (immigration and establishment) from a mosaic of habitats within the landscape (Moberg & Folke 1999; Nyström *et al.* 2000; Nyström & Folke 2001; McClanahan *et al.* 2002; Bengtsson *et al.* 2003; Elmqvist *et al.* 2004; Hughes *et al.* 2005). However, to my knowledge these authors within resilience research do not explicitly refer to any of the four paradigms outlined above. Thus, within resilience research there is a indiscriminate treatment of resilience mechanisms at a larger scale. This is a clear shortcoming within resilience research.

It is clear that any synthesis that links the four paradigms of meta-community dynamics to each other would greatly facilitate empirical work and would provide a much more realistic framework for understanding ecological mechanisms at these larger scales (Leibold *et al.* 2004; Chase 2005). Owing to the fact that such a more general framework does not exist up to now, research must explicitly stick to one of these paradigms when investigating resilience mechanisms on a larger scale. Hence, enlightened theory of resilience mechanisms at a larger scale should explicitly refer to one of the four paradigms. This is important because each of the paradigms has limited power of explanation and applicability and this must be made explicit.

In what follows, insights on resilience mechanisms are derived from a *source-sink* or *mass-effects perspective*. This perspective assumes that there is *regional heterogeneity* (patches are different with respect to environmental conditions) and that species trade-off such that they are favoured in some habitats but not others, as they vary in their relative ability to compete in and colonize habitat patches. Source-sink meta-communities are characterized by (*a*) *mass effects* and (*b*) *source-sink effects*. A *mass effect* is defined as a mechanism for spatial dynamics in which there is net flow of individuals created by differences in population size (or density) in different patches, while a *source-sink effect* is defined as a mechanism for spatial dynamics in which there is the enhancement of local populations by immigration in sink localities due to immigration of individuals from other localities where emigration results in lowered populations. Due to these processes a species can persist as sink populations in patches where they are not favoured, if they are maintained by immigration from other (source) habitats. This is termed the *rescue effect* (Chase *et al.* 2005).[126]

It is important to stress that meta-community models do not refer explicitly to the maintenance of ecological resilience, rather they focus on the maintenance of species diversity within the landscape. Yet in relevant literature referring to larger scales, there are to my knowledge no good alternatives to the meta-community framework. Therefore, I stick to the meta-community framework in the following and I assume that local and regional species diversity contributes to the *insurance effect* of biodiversity, even though the insurance effect is clearly dependent not only on species richness but on the provision of response diversity within functional effect groups.

The main proposition of source-sink theory goes as follows (Loreau *et al.* 2003a): If different systems experience different environmental conditions (*environmental heterogeneity*) and fluctuate asynchronously, different species are expected to thrive in each system at each point in time (*temporal niche complementarity*), and dispersal ensures that the species adapted to the new environmental conditions locally are available to replace less adapted ones as the environment changes (*regional functional compensation*). As a result, species richness may enhance and buffer ecosystem processes by virtue of spatial exchanges among local systems in a heterogeneous landscape. That is, in this view species richness at the landscape scale fosters ecological resilience.

Thus, seventh, ecological resilience is dependent on the *existence of a matrix of source habitats in the vicinity of the disturbed area*. In many cases, local functional diversity and species richness is maintained by immigration from surrounding habitats (Loreau & Mouquet 1999; Mouquet & Loreau 2002; Cadotte 2006). This is in line with the empirical evidence found within research on resilience mechanisms at a larger scale (Moberg & Folke 1999; Nyström *et al.* 2000; Nyström & Folke 2001; McClanahan *et al.* 2002; Bengtsson *et al.* 2003; Elmqvist *et al.* 2004; Hughes *et al.* 2005). For example, a network of coral reefs increases the chances both for larger

[126] Note that a source-sink perspective is entirely consistent with niche theory for it is not possible to have more species coexisting at equilibrium than the number of limiting factors (Mouquet & Loreau 2002).

disturbance events to be buffered within the seascape and for destroyed reefs to be naturally restored by colonizing species from surrounding reefs (Nyström & Folke 2001; McClanahan *et al.* 2002). Similarly, isolated reefs may be the most susceptible to climatic change-driven reef degradation (Graham *et al.* 2006). From this follows that larger scales have to be taken into account for including recruitment processes and meta-population analyses (Folke *et al.* 2004; Hughes *et al.* 2005).

But there is another crucial process that has to be acknowledged. The existence of source habitats alone is a poor measure of ecological resilience. In addition, the connectivity between habitats has to be taken into account. In this thesis the concept of *connectivity* includes two interrelated components, namely (*a*) the distances between the habitats of a meta-community, and (*b*) the dispersal rate of the species within the meta-community (cf. for a similar approach Leibold *et al.* 2005).

The first issue highlights that highly connected meta-communities will have patches that have small distances between them, whereas poorly connected communities will have patches that are more distant from each other. The second point refers to the dispersal abilities of specific species and takes into account that these will vary with respect to species identity and landscape pattern. For the latter, Leibold *et al.* (2005) distinguish between four types of landscapes, which are related to the four paradigms of meta-community theory outlined above. Those are: (1) *well-mixed landscapes*, i.e. distances between patches are short enough that individuals in different patches encounter each other with similar likelihood as they do individuals in the same patch over the course of their life cycle, (2) *mass effect landscapes*, i.e. distances are longer than in well-mixed scenario but there is sufficient migration that source-sink relations occur between heterogeneous patches, (3) *species-sorting landscapes*, i.e. distances are too far to support meaningful source-sink relations, but there are sufficiently small that there is no effective dispersal limitation for compositional change, and (4) *patch dynamics landscapes*, i.e. the habitable sites are far enough apart that patches may not be colonized immediately. Clearly, in each landscape different groups of species will have the highest dispersal rates.

Within resilience research, connectivity is related to the concept of *mobile links*. Mobile links are defined as “organisms, which support essential functions by connecting areas to one another and contribute to ecosystem resilience” (Lundberg & Moberg 2003: 87).[127] First of all, there is a distinction between *active mobile links*

[127] The term “functions” hereby refers to ecosystem processes (cf. Jax 2005).

(e.g. insects or birds) and *passive mobile links* (e.g. seeds, larvae, spores) (Nyström & Folke 2001). In addition, there are different functional groups of mobile links. Those are: (1) *resource linkers*, i.e. animals that transport and translocate essential resources, such as organic material, nutrients and minerals, (2) *genetic linkers*, i.e. organisms that carry genetic information between habitats, such as seed dispersers and pollinators and (3) *process linkers*, i.e. organisms that connect habitats by providing or supporting an essential process, e.g. grazers that structure the development of plant communities (Lundberg & Moberg 2003).

In some publications, the resilience approach suggests that increased connectivity among habitats (a larger amount of mobile links) *per se* is conducive for ecological resilience (Nyström & Folke 2001; Hughes *et al.* 2005). This proposition may be inaccurate. No clear distinction is made between the two interrelated components of connectivity, namely (*a*) the distances between habitats, which is the result of habitat loss and fragmentation on the one side and (*b*) dispersal rates on the other side.

The first component, i.e. *habitat loss and fragmentation*, has been discussed in some detail in section 3.2.3.3.3 and I will treat it only shortly here. In general, habitat loss has large negative effects on species richness and composition, whereas habitat *fragmentation* – in a narrow sense defined as breaking apart of habitat, independent of habitat loss - generally has much weaker effects on biodiversity and these effects are at least as likely to be positive as negative (Fahrig 2002, 2003). For many species there are *extinction thresholds* above a specific amount of habitat loss. These extinction threshols are dependent on each the reproductive rate of the organism, the rate of emigration of the organism from habitat, the habitat pattern in the landscape (habitat loss and fragmentation) and the matrix quality (survival rate of the organism in non-habitat areas), while the first two factors, i.e. the reproductive rate and the emigration rate, have the largest potential effect (Fahrig 2001). In addition, for many species there are *demographic isolation thresholds*, which signifies the critical distance beyond which species are unable to “move” between patches (Huggett 2005). Thus, it indeed matters whether habitat patches within a meta-community are more or less distant to each other.

The second component of connectivity, i.e. the dispersal rate, is discussed extensively in the literature in meta-community theory. Hereby, the *dispersal rate* is defined as the movement of individuals from a habitat to another per time (Holyoak *et* *al.* 2005). Dispersal rates of specific species will vary and different groups of species will have the highest dispersal rates in different landscapes (Leibold *et al.* 2005).

There is an important point here. Some studies of meta-communities distinguish *local diversity* from *beta diversity* (among-habitat variation of species diversity) and *regional diversity* (diversity at regional or landscape scales) and relate these measures to dispersal rates. Hereby, we can view local and beta diversity as additive quantities, equalling regional diversity. The important point is that both theoretical and empirical evidence suggests that, in general, local species diversity is highest at *intermediate dispersal rates* (Loreau *et al.* 2003a; Cadotte 2006).[128] On the one side, too low a dispersal rate means that both stochastic extinctions and negative interactions (e.g. competitive exclusion) cause local populations to become extinct without rescue. On the other side, at too high rates dominant competitors are introduced into all local communities and, thus, spatial variation in fitness is homogenized by immigration resulting in reduced local and regional diversity (termed *biotic impoverishment*) (Leibold *et al.* 2005; Cadotte 2006).[129] Furthermore, in highly connected meta-communities diseases, pests, fires and some invasive species may be spread more easily (Noss 1987; Simberloff & Cox 1987; McClanahan *et al.* 2002; van Nes & Scheffer 2005). Indeed, highly connected systems have the largest probability of exhibiting nonlinear shifts to undesirable regimes (Peters *et al.* 2004). Thus, there are in fact some reasons why too high a connectedness is destructive for both local and regional diversity.

According to Cadotte (2006), these results are scale-dependent and have important conservation implications. For example, if managers are interested in maintaining maximal diversity over a fragmented landscape, then perhaps restricting, or at least not enhancing, dispersal would be best ensure regional diversity. However, if the concern is a single local community, then enhancing immigration may be the best option. Thus, both increasing and decreasing landscape connectivity can either increase or decrease species diversity and the average magnitude and temporal variability of ecosystem processes, depending on the initial level of landscape

[128] This is true for both Lotka-Volterra models and lottery models with respect to source-sink dynamics (Mouquet *et al.* 2006).

[129] Loreau *et al.* (2003a) nicely illustrate this point: "Local species diversity and the insurance effects that are related to it collapse at both low and high dispersal rates. When dispersal is very low, each local community behaves as a closed system in which competitive exclusion proceeds; and, when dispersal is high, the whole metacommunity behaves as a single large closed system in which competitive exclusion also proceeds" (Loreau *et al.* 2003a: 12768).

connectivity and the dispersal abilities of the organism considered (Loreau *et al.* 2003a).

The discussion of connectivity between patches of the meta-community is also closely related to the concept of *corridors* that connect habitat patches within real landscapes, e.g. forest patches within agricultural land. The idea that corridors should be maintained between habitats for conservation objectives is an automatic consequence of the equilibrium theory proposed by MacArthur and Wilson (1967). According to the original theory, corridors would act by increasing the immigration rate. Once a species is locally extinguished at a specific habitat, the expected time to the next re-immigration is lowered by the availability of corridors (cf. for literature on the history of the concept of corridors Simberloff & Cox 1987).

On the one side, there is some empirical evidence for the merits of corridors for conservation purposes (cf. for a review Simberloff & Cox 1987). Advantages include each the increase of immigration rates to a specific habitat, provision of increased foraging area for wide-ranging species, provision of predator-escape cover for movements between patches, provision of a mix of habitats and successional stages accessible to species that require a variety of habitats for different activities or stages of their life-cycles and provision of alternative refugia from large disturbances (Noss 1987).

On the other side however, proponents of corridors assume that (*a*) stochastic population extinction must be continual in sink habitats at measurable rates in the short term, and (b) particular kinds of corridors will actually be used by the organisms for which they are intended and they will be used at rates high enough to forestall extinction, and these assumptions may not be met in many empirical cases (Simberloff & Cox 1987). Furthermore, disadvantages of corridors include each the spread of epidemic diseases, insect pests, invasive species, weeds and other undesirable species into habitat patches, the potential decrease of genetic variation among populations or sub-populations, the spread of fire and other abiotic disturbances and finally high economic costs (Noss 1987).

Consequently, the merit of corridors has to be assessed on a case-to-case basis. In some situations corridors may be of great use, and in others one can reasonably argue that they will be irrelevant or even detrimental (Noss 1987; Simberloff & Cox 1987). The necessary width of the corridors will vary depending on habitat structure and quality within the corridor, the nature of the surrounding habitat (the *matrix*), human use patterns, and the particular species that we expect to use the corridor (Noss 1987).

However, from a source-sink perspective, in general, intermediate dispersal rates maximize both local and regional species diversity.[130] These two properties build the basis for *regional functional compensation* between species or phenotypes in time at a regional scale if disturbances occur. This process is also termed *temporal niche complementation* (Loreau 2000). That means, for example, that minor species increase when major species decline and lost species are replaced by better adapted species colonizing from the larger species pool (cf. for the comprehensive concept of functional compensation section 3.3.1.1). Therefore, intermediate dispersal rates maximize the *insurance effects* of biodiversity at regional scales (Loreau *et al.* 2003a; Leibold *et al.* 2005), and, hence, spatial resilience to specific disturbance events.[131]

Thus, eighth, ecological resilience is dependent on *appropriate levels of connectivity of the meta-community, i.e. (a) small to moderate distances and (b) intermediate (but not high) dispersal rates between habitat patches*.

There is a further property that is important for ecological resilience. The source-sink model shows that immigration from an external source is able to maintain a high local diversity in a system that would otherwise tend toward competitive exclusion of all but one species (Loreau & Mouquet 1999). But the operation of this process requires that diversity be maintained in the source itself, which looks like transferring the problem of the maintenance of diversity to the source. This theoretical difficulty can be resolved by taking into account *regional heterogeneity* of the environment at the landscape scale, which can be defined as the differences among local communities in resource supply (Mouquet *et al.* 2006).

Within resilience research, in general, increased heterogeneity at the landscape scale is viewed to foster the range of local environmental adaptations which, in turn, begets resilience to change (Chapin *et al.* 2004; van de Koppel & Rietkerk 2004). Indeed, regional heterogeneity may weaken the tendency for large-scale catastrophic regime shifts (van Nes & Scheffer 2005).

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[130] As in the case of the intermediate disturbance hypothesis, the term “intermediate” is problematic, as it is at first not clear what the term refers to (K. Jax, *personal communication*).

[131] This insurance effect of regional species richness is indeed valuable for ecological (spatial) resilience of landscapes to specific disturbance regimes. In contrast, the conservation of single species may require other conservation measures, as discussed with respect to the concept of corridors above. The important point is that measures to increase the ecological resilience of a given landscape may in some cases be detrimental for other conservation objectives.

However, theoretical evidence suggests that *intermediate regional heterogeneity* is optimal to ensure ecological resilience (Mouquet *et al.* 2006). If local communities are too similar in their resource supply points, then dispersal does little to increase species diversity because no unique sources are available to any species. If local communities are too different, then dispersal again has only a small effect on diversity because differences in local competitive ability are too great and inferior competitors cannot be rescued from competitive exclusion by immigration. Maximal local and regional species richness is found when the heterogeneity between habitats is intermediate (Mouquet *et al.* 2006). Thus, ninth, ecological resilience is dependent on *an intermediate degree of regional heterogeneity*.

In sum, spatial resilience, i.e. ecological resilience at a regional scale, is the dynamic capacity of a network of patches to avoid shifts to an undesirable regime at the regional scale. It results from (*a*) the amount of biological legacies within the disturbed patch, (*b*) the existence of source areas of potentially colonizing species in the vicinity of the disturbed patch, (*c*) moderate distances and intermediate dispersal rates between patches and (*d*) regional heterogeneity at an intermediate level. Again, each of these features are properties that are conducive for ecological resilience. But the decisive mechanism for ecological resilience is the potential for regional functional compensation, understood as the tendency of coexisting, competing species to increase in abundance or replace species should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition. Thus, each resilient-conducive property of ecosystems at larger scales (e.g. biological legacies, source habitats, intermediate dispersal rates) contributes in a particular way to the functional compensation as regards to specific ecosystem processes. Hence, tenth, *ecological resilience is dependent on the potential for regional functional compensation*.

3.3.2.3 The Context of Oher Factors

Effects of biodiversity on ecological resilience have to be understood in the context of other factors, i.e. in relation to disturbance regimes and abiotic conditions, such as soil properties or climate. Each of these factors can have a greater effect on ecological resilience than does biodiversity (Tilman 1999; Beierkuhnlein & Jentsch 2005; Hooper *et al.* 2005). In what follows section 3.3.2.3.1 describes the effects of disturbance on ecological resilience, while section 3.3.2.3.2 delineates the effects of abiotic factors.

3.3.2.3.1 Ecological Resilience and Disturbances

In ecology, the concept of *disturbance* can be understood in two fundamentally different ways.[132] The *relative definition* of disturbance seeks to define disturbances as causing deviation from the normal dynamics of an ecosystem. Thus, destructive events like moderate fires in grasslands or tree falls in old-growth forests, which "characterize" these ecosystems, are not considered to be "disturbances". In this view, disturbances are events that change the characteristic ecosystem structure and processes (e.g. elimination of fire from a grassland or introduction of fire to old growth-forests that had no history of this disturbance type) and cause alteration from what is usual or expected (White & Jentsch 2001). Note that the relative definition of disturbance is also related to the concept of *natural variability*, which is widely used by natural resource managers (Landres *et al.* 1999).

In contrast, the *absolute definition* of disturbance is based on any physical and measurable changes in variables or in the disposal of resources, whether or not these changes are recurrent, expected or normal (White & Jentsch 2001). In this sense, disturbance is defined as "any relatively discrete event in time that disrupts the ecosystem, community or population structure and changes the resources, substrate availability or physical environment" (White & Pickett 1985: 7).[133]

The former, the relative definition of disturbance assumes that (*a*) we can define the normal dynamics of ecosystems, and (*b*) disturbances should be compared on the grounds of statistical precedence. Both assumptions might be problematic (White & Jentsch 2001).[134] However, within resilience research for several reasons[135] the relative definition of disturbance is prevalent and I will follow this path in the following.

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[132] Confer White and Jentsch (2001) for a detailed review on the concept of disturbance.

[133] The different concepts of "disturbance" have to be understood in the larger context of specific perspectives on nature and ecological systems, e.g. equilibrium theories (Jax 1998/99).

[134] The first assumption, i.e. that we can define the normal dynamics of ecosystems is indeed a deeply fundamental point for resilience theory. There must, in fact, be some kind of continuity or normal dynamics for ecosystems for a stability concept, such as ecological resilience, to refer to. White and Jentsch (2001) argue that disturbance events are dependent on both the temporal and spatial scale. For example, disturbance events may change if climate changes. Therefore, White and Jentsch argue, there are no constant conditions for ecosystems. I reject this argument here and argue for some *intermediate stability* (in the sense of Grimm 1998) for most ecosystem types. In fact, the classification of ecosystem types or biomes suggests this intermediate stability.

It is further important to distinguish between a *disturbance event*, i.e. a relatively discrete disturbance in time, and a *disturbance regime*, i.e. the distribution and characteristics of disturbance events in space and time (Jax 1998/99). The latter, the disturbance regime, is characterized by several variables (White & Pickett 1985: 6ff; Jax 1998/99). Those are: (1) the *organism-related variables*, i.e. the ecological unit under study, the variables of interest (e.g. biomass, age-structure of a population), the variables that govern the impact of the disturbance (magnitude, extension, frequency), disturbance thresholds and (2) the *variables of description*, which, in turn, consist of (2a) *cause-related variables*, i.e. frequency, predictability, intensity, extension, distribution and duration, and (2b) *impact-related variables*, i.e. intensity of impact, spatial extension of impact, size, shape and location of the areas affected, time of disturbance (relative to the life cycle of the organism considered), specific sensibilities of the objects and synergistic effects of different disturbances.

It is important to note that this comprehensive concept of a disturbance regime is an ideal notion. In most of the empirical cases, it will not be possible to estimate each of the variables considered above. Rather, from a practical standpoint, it will be crucial to specify the most relevant and important variables, e.g. frequency, intensity, size of the areas affected and spatial distribution (Jax 1998/99).

The relative definition of disturbance highlights several distinctions. First of all, one may distinguish roughly between *natural disturbances* and *human-caused disturbances*. The term “natural disturbances” refers to events such as wind, floods, volcanic eruptions, and alike, while human-caused disturbances correspond to pollution, habitat loss and fragmentation or climate change, for instance. Yet this distinction is not straightforward, because most disturbances have both human and natural aspects. For example, a flooding event, e.g. the big flood in Germany 2002, is caused by a “normal” natural event (e.g. that rivers change the direction of flow and the amount of water they carry) but the *intensity* of flooding is greatly enhanced by a human-caused factor, i.e. current land-use practices, as the overflow is much higher in agricultural land than in forest areas or wetlands, for example. Similarly, the human-caused climatic change greatly increases the risk of floods, storms and alike.

However, even if this is true, statements about disturbance events are only valid for a specific temporal and spatial scale.

[135] Only the relative definition of disturbance allows to view episodic disturbances on a small spatial and temporal scale as “normal” for ecosystem dynamics and as conducive for a system’s ecological resilience. I will consider this important point below. This is, of course, only an internal or strategic motivation to prefer the relative definition of disturbance, not a good reason for it.

Even though the distinction between human-caused disturbances and natural disturbances is not sharp, I regard it is useful for conceptual purposes in the following.

Second, one may distinguish between *large, infrequent disturbances* and *small, frequent disturbances* (Romme *et al.* 1998). Large disturbances may be delimited from small disturbances by taking a statistical approach (assemble an empirical distribution of sizes, intensities or frequencies of disturbances and determine which represent the extreme tail of the distribution). But it is still an arbitrary decision where to draw the line between "large" and "small" or "infrequent" or "frequent". In this thesis, I follow Romme *et al.* (1998) and understand a *large, infrequent disturbance* as those that lie toward the far end of a statistical frequency distribution, are of magnitudes that humans perceive to be large, and that have qualitatively different effects compared to small, frequent disturbances in causing e.g. a dynamic regime shift of the whole ecosystem (i.e. a *threshold effect*). Examples for this type of disturbances are volcanic eruptions, big fires, extreme floods or storms. Correspondingly, *small, frequent disturbances* are defined as disturbances that lie at the close end of a statistical frequency distribution, are of magnitudes conceived as normal by humans and does not cause a threshold effect in the specific ecosystem. Thus, disturbances may occur at *a small or large spatial and temporal scale* ("small" and "large" according to the statistical distribution considered above).

A further distinction highlights the difference between *pulse disturbances* and *press disturbances* (Bengtsson 2002; Folke *et al.* 2004). *Pulse disturbances* are mostly natural disturbances that occur in an episodic fashion within the normal and typical behaviour of ecosystems. In contrast, *press disturbances* are mostly human-caused disturbances that occur very frequently within land-use systems. Examples include heavy grazing by herbivores in rangelands or the extensive and continuing pollution of lakes by phosphorous emissions. Clearly, this is closely related to the relative definition of disturbance and is laden with its difficulties (Jax 1998/99; White & Jentsch 2001).[136]

Finally, there are two further terms with respect to the concept of disturbance we have to clarify here for a use in the subsequent chapters. First, c*ompounded disturbances* are disturbances that deviate the normal, pulse dynamic of disturbances and are of high magnitude or have a higher frequency that disrupts the

[136] Note that, from an absolute perspective on disturbances, press disturbances are actually no disturbance events (because they are not discrete enough) but a change in abiotic conditions (cf. Jax 1998/99).

normal cycle of disruption and recovery within communities (Paine *et al.* 1998). Second, the term *synergistic disturbances* refers to various different disturbances that occur at the same time and have inhibitory synergies. Examples are the multiple stressors of coral reefs, e.g. fishing, pollution and sedimentation (McClanahan *et al.* 2002). Both kinds of disturbance events may have multiplicative rather than additive effects on the ecological unit in question. Disturbances are *multiplicative* or *synergistic* when their combined effect is larger than predicted from the sizes of the responses to each stressor alone, and *antagonistic* when the cumulative impact is smaller than expected (Vinebrooke *et al.* 2004).

Using these distinctions and following Bengtsson (2002), it is now possible to distinguish roughly between three types of disturbances. The first type, (1) *pulse disturbances*, are natural disturbances that occur episodically followed by a reorganization phase. Pulse disturbances are usually parts of natural[137] ecosystem dynamics and hence most organisms affected by such disturbances are adapted to survive them or to recolonize disturbed areas. Examples are small-scale fires in rangelands or insect outbreaks in forest ecosystems. The second type, (2) *large, infrequent disturbances*, correspond to the category we explored above. These are uncommon surprises that have a frequency so low that most organisms are unlikely to adapt to them. Examples are volcanic eruptions, large-scale fires or extreme floods. Finally, the third type, (3) *press disturbances*, are chronic stress agents that often, but not always, are of anthropogenic origin. Most species have not been able to respond evolutionary to anthropogenic disturbances of this type. Examples include acid rain, heavy metals, intense monoculture agriculture, heavy and continued grazing or ongoing pollution events.[138]

Note that this typology rests on a relative definition of disturbance. Therefore, the typology stresses the evolutionary adaptations of populations to certain disturbance regimes. For example, many plant species in fire-prone environments require smoke-related chemical cues for seed germination (Moritz *et al.* 2005). Consequently, it becomes important to distinguish between the “expected” and the “unexpected” (Shea *et al.* 2004). Species are adapted to usual and normal, “expected” disturbance events, e.g. fire events or insect outbreaks, whereas they are not adapted to unusual,

[137] Again, it is essential to specify the “natural” or normal dynamics of an ecosystem type. In this thesis, however, I regard it as useful to speak of normal dynamics, for example of a boreal forest or savannah.
[138] Of course, there are species that are adapted to these conditions, due to their evolutionary history, but not due to the specific human disturbances they experience today.

"unexpected" disturbance events, e.g. many human-caused disturbances and large, infrequent disturbances. Certainly, this is in stark contrast to the absolute perspective on the concept of disturbance. From this perspective, a real adaptation to a disturbance event is viewed as impossible (Jax 1998/99).

I will use this rough typology in the following to illustrate the ambivalent effects of disturbance on ecological resilience. However, it is well beyond this thesis to develop a comprehensive picture of the relationship of "disturbance" and ecosystem "stability" or resilience. This may be a fruitful field for further research.

On the one side, the occurrence of *disturbance type 1*, i.e. pulse disturbances, is viewed to be conducive for the ecological resilience of an ecosystem (Bengtsson 2002; Bengtsson *et al.* 2003). This notion is the consequence of a crucial concept of the background theory of ecological resilience, namely the meta-models of both the adaptive cycle and the panarchy (cf. for a detailed treatment 3.2.1). The occurrence of the back loop of the adaptive cycle, i.e. the creative destruction (Ω) phase and the reorganization (α) phase, is regarded as a normal and important part of ecosystem dynamics. In addition, the Revolt-interaction across scales as part of the panarchy model is viewed as a common process. In this sense, the entire back loop functions as an engine for the renewal of ecosystem dynamics. As Walker and Abel point out:

> "[t]he reorganization phase allows a new combination of species to become established. The new combination is potentially better adapted to the environmental conditions that followed the disturbance. The process repeats itself. Resilience is maintained through these repetitions" (Walker & Abel 2002, 312).[139]

Correspondingly, Bengtsson *et al.* (2003) term Holling's four phase cycle the *ecosystem renewal cycle*. Small-scale disturbances are conceived as intrinsic parts of ecosystem dynamics. That means, natural pulse disturbance is endogenous to the cyclic process of ecosystem dynamics (Folke *et al.* 1998; Berkes 2003).

This is in line with a notion of *complex adaptive systems* put forward by S. A. Levin (Levin 1998). As Levin points out: "The maintenance of diversity and individuality of components [of complex adaptive systems, F.B.] implies the generation of perpetual novelty" (Levin 1999: 432), by which the structure and processes of the hierarchy can be reorganized.

[139] Note that Walker and Abel (2002) use resilience for the *extended-ecological meaning* of ecological resilience (cf. section 4.1.2).

Thus, the ecological resilience of ecosystems at a landscape or regional scale is fostered by the destruction of systems at a smaller scale (Pickett & White 1985b; Gunderson 2000), i.e. by the occurrence of natural, pulse disturbances at a small spatial and temporal scale. As Walker *et al.* point out:

> "Some loss of resilience, at some scales, is an inevitable feature of the cross-scale dynamics in complex adaptive systems. Losses, however, can be managed so as to be confined to smaller organizational scales, with less consequent social and environmental dislocation. All else being equal, a system that loses resilience at small, and more manageable, scales of organization (e.g. patches) will be more resilient than one where these losses occur at larger scales (e.g. landscapes)" (Walker *et al.* 2004).

But what is the evidence for this view? The work just considered rests rather on intuitive arguments and some theoretical evidence. However, there is also some empirical evidence for the notion that natural, pulse disturbances at a small scale foster the ecological resilience of ecosystems at larger scales. The *intermediate disturbance hypothesis* (IDH) originally proposed by Connell (1978) states that the species richness of a specific community peaks under intermediate disturbance regimes. This pattern of species coexistence is due to a competition-colonization trade-off. Too much disturbance, and longer-lived species cannot persist in the system (these species typically show relatively poor colonization abilities and need a long time to mature and reproduce); too little disturbance, and competitive superiors drive pioneer species to extinction. At intermediate disturbances there is a parameter zone where both good competitors and good dispersers coexist indefinitely, with the competitively inferior species occupying the recently disturbed sites. The IDH has been supported in a huge range of community types (from aquatic to terrestrial) at scales ranging from microcosms to the entire landscape (for a comprehensive review of the empirical evidence cf. Shea *et al.* 2004).

However, it is important to note that there are several limitations to the IDH. First, different studies emphasized different aspects of "intermediate" disturbance. In empirical studies, for instance, intermediacy was defined in terms of intensity (17 cases), frequency (13 cases), time since disturbance (three cases), extent (two cases) and duration (one case) (Shea *et al.* 2004). It is thus important to explicate the aspect of intermediacy of concern. Second, even though the IDH refers to a common pattern in nature, this pattern may be generated by different classes of mechanisms.

According to Roxburgh *et al.* (2004) these are: (1) the *storage effect*, i.e. the species' attributes allow gains made during favourable periods to be stored in the population for use during unfavourable periods; (2) the concept of *relative nonlinearity*, i.e. differences in the responses of competitors to fluctuations in resource availability that are relatively nonlinear; and (3) a third category that incorporates all of those mechanisms that do not rely upon temporal variability. Despite these difficulties, the IDH refers to a fairly general pattern in nature (Shea *et al.* 2004).

In sum, the evidence from the intermediate disturbance hypothesis suggest a conducive effect of pulse disturbances on species richness. In this case, again there is no direct relationship of pulse disturbances and ecological resilience. Rather, pulse disturbances often result in species richness and this, in turn, is proposed to lead to increased functional compensation on local and regional scales, which in turn, promotes ecological resilience. Thus, the main proposition is that, eleventh, ecological resilience is fostered by *the occurrence of intermediate, pulse disturbances at a small spatial and temporal scale*.[140]

In contrast, there is a strong inhibiting role of "disturbance" for ecological resilience. In fact, the term *ecological resilience* is defined in this thesis as *the capacity of an ecosystem to resist disturbance and still maintain a specified state*. That is, (operational) statements of (specified or targeted) ecological resilience ought to be conceptualized *with respect to specific disturbances*.

The relation of ecological resilience to a particular disturbance regime is mainly the topic of section 3.4.2.2. In what follows I will only hint to some crucial ideas. As *disturbance type 1*, i.e. pulse disturbances, are considered as conducive, the concept of ecological resilience refers only to *disturbance type 2*, i.e. large, infrequent disturbances, e.g. storms or big floods, and *disturbance type 3*, i.e. press disturbances, e.g. heavy land use practices or continued pollution. In addition, most situations in environmental management are characterized by *multiple* and potentially *compounded* or *synergistic* disturbances.[141] Thus, I propose that the *concept of*

[140] The conducive effect of pulse disturbances at a small spatial and temporal scale has been termed *pulse stability* by Odum (1999: 309f).

[141] With respect to synergistic disturbances, Vinebrook *et al.* (2004) suggest a further property that is conducive for ecological resilience. The ecological resilience (in this case: resistance) of communities that experience multiple *disturbances of type 2 or 3* will depend on species tolerances being positively correlated. That means, either disturbance eliminates certain species, but leaves more species that are tolerant of the other disturbance than if species are unrelated. This, in turn, increases the potentially possible functional compensation between species. This pattern is termed *stress-induced community tolerance* (Vinebrooke et al. 2004). However, before this proposition gets some status of plausibility, more theoretical and empirical work is needed.

ecological resilience should be applied to multiple, potentially compounded and synergistic press or large-infrequent disturbances only.

3.3.2.3.2 Ecological Resilience and Abiotic Variables

The wider context to study resilience mechanisms includes the abiotic environment of the community. In fact, *abiotic variables*, i.e. the physical and chemical conditions that affect or are affected by the community, e.g. climate, soil properties or nutrient storages, may have greater effects on the "stability" or ecological resilience of a community than biotic factors, such as response diversity or functional compensation abilities (Hooper *et al.* 2005), particularly at a landscape or regional scale (Loreau *et al.* 2001).[142]

The abiotic sphere of ecosystems or landscapes has been the scientific object primarily of geography and, later on, also of landscape ecology and ecosystem research. It is well beyond this thesis to even delineate the state of affairs of these scientific disciplines. In ecology, there is some empirical evidence for the crucial effects of abiotic variables on ecological resilience. For example, Grimm *et al.* (1999) point to the relevance of abiotic conditions in the Wadden Sea, Germany. To enable the resilience mechanisms played out by organisms (i.e. the biotic mechanisms, such as response diversity or functional compensation), care must be taken that the abiotic processes that enable these mechanisms in the first place are a primary concern of conservation efforts.

Similarly, many of the case studies of resilience research show that abiotic variables play an important role for the provision of ecological resilience. Many of the slow variables, i.e. those variables that control the position of an ecosystem within a stability landscape (cf. section 3.2.3.3.4.1), are abiotic factors. For example, Carpenter *et al.* (2002) identify soil and lake mud phosphorus as controlling slow variable for lakes in the Great Lake Region, while Jansson and Jansson (2002) stress the central role of nutrients and their balanced status in the Baltic Sea ecosystem and suggest nutrient storages, such as phosphorus, and the nitrogen/phosphorus ratio as slow variables. In the case of the Everglades freshwater marshes in the USA, Gunderson and Walters (2002) specify also the slowly changing

[142] The abiotic conditions that either do not affect or are not affected by the community do not belong to the *environment* of a community but constitute what is termed the *surrounding*.

soil phosphorus level as controlling factor. Moreover, Walker and Abel (2002) suggest *determinants of ecosystem resilience* for rangelands in Australia and Zimbabwe, which comprise biotic factors such as plant communities with high species richness, high genetic variability or ecological memory in seed-banks, but also slow abiotic factors, for instance, soils with low erodibility that maintain infiltration rates under grazing pressure, climates that have periods of higher rainfall that allow vegetation processes to recover from disturbances, and landscapes that have sufficient relief to allow water and wind to concentrate nutrients and water in fertile patches.

There is a crucial point here. According to the resilience approach, the loss of several resilience mechanisms - such as ecological redundancy or ecological memory - that is related to biotic variables of the ecosystem results in systems that are less resilient to change. But in many cases, it may be finally the changes in the slow variables, which are, in turn, often abiotic variables, that define the loss of ecological resilience, as proposed by Gunderson and Walters (2002).

This view leads to the fundamental question *whether either biotic variables or abiotic variables are most relevant for ecosystem "stability" or resilience*. Viewed historically and following Naeem (2002), the relevance of each factor has been dependent on the choice of a distinct theoretical framework within ecological science. According to Naeem (2002), in ecology two fundamental and interacting theoretical frameworks have to be distinguished with respect to the discussion on ecological "stability"(Naeem 2002), i.e. (1) the *Community Ecology Paradigm* (CEP) and (2) the *Biodiversity-Ecosystem Function Paradigm* (BEFP).[143] These two theoretical frameworks correspond to two commonly separated intellectual streams or sub-disciplines in ecology, namely, community ecology and ecosystem ecology.

The CEP (or "community ecology paradigm") holds that patterns in the distribution and abundance of species are a function of abiotic and biotic factors. Abiotic factors set regional patterns in distribution and abundance while biotic factors secondarily modify regional patterns. The CEP does not admit a strong role for ecosystem processes in understanding nature and portrays biodiversity as a passive consequence of intrinsic structure (e.g. number, type, and arrangement of interspecific conditions) and extrinsic factors (e.g. climate, geology, and chance events). Thus, the emphasis is on the abiotic context of biodiversity.

[143] Again, the term "paradigm" is widely used within ecological science, but in most cases does not correspond to the influential meaning proposed by Kuhn (1962).

In contrast, the BEFP (or “biodiversity ecosystem-functioning (ecosystem ecology-) paradigm”) views the environment primarily as a function of biodiversity. The BEFP recognizes the abiotic conditions of the environment as driven, at least in part, by ecosystem processes (e.g. nutrient cycling and energy flow), which is, in turn, the result of biodiversity. Thus, the emphasis is on the biotic components of ecosystems, that is, on biodiversity.

With the discussion on resilience mechanisms in mind one might ask: which theoretical framework is true? Interestingly, Naeem (2002) suggests that neither the CEP nor the BEFP are true or false in an absolute sense. Rather the two paradigms represent two extreme positions in an *ecological dialectic* that leads to progress in ecological science. Thus, according to Naeem (2002), both paradigms are correct in a relative sense and constitute steps in the evolution of understanding nature, not accurate representations of the true workings of nature. Thus, and this is the message of this section, ecological resilience does not derive of *either* biotic components *or* abiotic conditions, but essentially from the interplay of both abiotic and biotic factors. Abiotic and biotic variables with respect to ecological resilience have to be understood as *interrelated and interacting components*. Therefore, the question on how ecological resilience does emerge is not so much about either/ or but, metaphorically spoken, on *the relative strengths of biotic vs. abiotic controls on ecological resilience* (for the metaphor cf. Huston & McBride 2002). Thus, twelfth, ecological resilience is dependent on *the value of controlling abiotic variables*.

3.3.3 Resilience Mechanisms: Concluding Remarks

This section synthesizes some of the findings of the previous sections and concludes with provisional results. I at first sketch some alternative approaches to resilience mechanisms in the relevant literature (section 3.3.3.1), second, point to the salient issues in the search for resilience mechanisms (section 3.3.3.2), before I third present a list of resilience properties and mechanisms (section 3.3.3.3). Subsequently fourth, resilience mechanisms are considered in their relation to recent insights taken from the philosophy of science in ecology (section 3.3.3.4).

3.3.3.1 Alternative Approaches to Resilience Mechanisms

Some authors examine the concept of resilience mechanisms within a larger framework of *complex adaptive systems* (Levin 1999; Dizon & Yap 2006).[144] In this context, resilience mechanisms can be viewed as a special case of mechanisms used by complex systems in general to prevent collapse. These mechanisms of complex adaptive systems include (*a*) redundancy, (*b*) modularity, (*c*) feedback and (*d*) evolvability (Dizon & Yap 2006). The first mechanism, i.e. *redundancy,* involves the compensation of a lost or damaged component by another one with a similar or overlapping function[145] and corresponds to the discussion on ecological redundancy and response diversity, considered in section 3.3.1.3.2 and section 3.3.2.1. The second mechanism, i.e. *modularity*, ensures that damage in one part of the system does not spread to the entire system. This refers to the intermediate but not high connectivity between habitat patches of the meta-community as well as to intermediate heterogeneity, considered in section 3.3.2.2. The further mechanism, *feedback* and *evolvability*, defined as the capacity to generate heritable phenotypic variation, may refer roughly to the concepts of natural pulse disturbances and to population diversity. The message of this is that insights from complex adaptive systems may help to gain a deeper understanding of resilience mechanism.

A further line of research follows another path. Jeltsch *et al.* (2000) suggest that it is useful to focus on the crucial factors that are responsible for the *existence* of biomes

[144] It is again far beyond this thesis to delineate the approaches to complex adaptive systems in any detail. For futher literature confer Dizon and Yap (2006).

[145] The term "function" is used here in the sense of a specific role in the whole system (cf. Jax 2005).

or ecosystem types itself. These *ecological buffering mechanisms* prevent the ecosystem type from crossing the “boundaries” to other ecosystem types.[146] For example, the buffering mechanisms allow a savannah ecosystem, i.e. a tropical or subtropical mixed tree (or shrub)-grass community, to persist in critical situations where this system is driven to its boundaries, e.g. pure grasslands or tropical forest. For savannahs, ecological buffering mechanisms include fire, browsers, e.g. kangaroos, prairie dogs or rabbits, and micro-sites favouring tree establishment and survival (Jeltsch *et al.* 2000). The concept of “ecological buffering mechanisms” is a fundamentally different approach compared to the approach of “resilience mechanisms”. The former focuses more on the “negative” reasons, i.e. on the mechanisms that prevent the non-existence of specific ecosystem types, the latter emphasizes the positive reasons for their existence.[147]

3.3.3.2 The Salient Issues of My Discussion on Resilience Mechanisms

Using the *conceptual volume* proposed in section 3.3.1.6.2, it is possible to locate the properties and mechanisms found as being conducive for ecological resilience within a larger framework of both the diversity-stability discussion (DS) and the biodiversity-ecosystem function debate (BDEF), described in sections 3.3.1.1 and section 3.3.1.2. This section tries to find the overlapping consensus of each the DS, BDEF and resilience research. There are three salient issues.

First, there is a focus on a *functional approach to biodiversity* (cf. 6c in the conceptual volume in section 3.3.1.6.2). That is, ecological resilience is viewed to be dependent not so much on species richness *per se* but on functionally important types of species, functional diversity, the maintenance of (critical) functional groups, ecological redundancy, response diversity and functional compensation. All these concepts refer to a functional approach to biodiversity, as most of them consider a (functional) “role” of certain species in the whole ecosystem (Jax 2005). As we have considered in section 3.3.1.2.2, within BDEF research, there is a growing consensus that properties, such as functional diversity or functional important types of species, are likely to be the components of biodiversity most relevant for ecosystem processes and stability properties (Diaz & Cabido 2001; Naeem & Wright 2003;

[146] Note that Jeltsch *et al.* (2000) suggest that the concept of ecological buffering mechanisms does not imply any equilibrium or non-equilibrium assumptions.

[147] But confer section 3.3.2.3.1 on the role of natural, pulse disturbances, as a sort of “negative” reason.

Hooper *et al.* 2005; Srivastava & Vellend 2005; Cardinale *et al.* 2006; Petchey & Gaston 2006; Wright *et al.* 2006; Thompson & Starzomski 2007).

Second, authors attempt to *include more ecological realism* (Srivastava & Vellend 2005) into specific studies (cf. 6d in the conceptual volume). That is, studies focus not only on small scales but on larger scales (e.g. landscape, region) and try to include processes, such as dispersal or colonization events (Nyström & Folke 2001; Bengtsson 2002; Bengtsson *et al.* 2002; Bengtsson *et al.* 2003; Diaz *et al.* 2003; Naeem & Wright 2003; Symstad *et al.* 2003; Cardinale *et al.* 2004; Hughes *et al.* 2005; Srivastava & Vellend 2005; France & Duffy 2006; Matthiessen & Hillebrand 2006). However, resilience research fails to acknowledge any perspective gained by the meta-community framework (e.g. species-sorting, mass-effects) and this is a clear shortcoming. Here, BDEF-research offers useful theory in order to address the relation between biodiversity and ecological resilience at larger scales (Leibold *et al.* 2004; Holyoak *et al.* 2005).

Finally, authors increasingly recognize that any relation of biodiversity and ecological resilience is *not universally valid* (cf. 6b in the conceptual volume). This becomes clear if the *ecological checklist* proposed by Grimm and Wissel (1997) and considered in detail in section 3.3.1.1 is applied to the discourse of resilience mechanisms. That is, I propose that statements about ecological resilience are only valid for particular *ecological situations* of the following kind (cf. *Table 5*): (1) *Level of description*: ecosystem; (2) *variables of interest*: ecosystem structure and processes of a specified regime with a certain self-identity in specific bounds; (3) *reference dynamic*: adaptive cycle and panarchy in the sense of Holling (2001); (4) *disturbance*: a specified disturbance regime in the sense of White and Pickett (1985); (5) *spatial scale*: to be specified; and (6) *temporal scale*: to be specified. In addition, statements are only valid for a specific ecosystem type, as considered below. Note that there are principally infinite ecological situations. Therefore statements about resilience mechanisms have to be specified in terms of the ecological checklist. Hence, they are valid for these situations only.

Table 5: Focus of statements about ecological resilience

Feature of the ecological checklist	**Ecological situation ecological resilience**
Level of description	Ecosystem
Variables of interest	Ecosystem structure and processes of a specified regime with a certain self-identity in specific bounds
Reference dynamic	Adaptive cycle and panarchy
Disturbance	To be specified
Spatial scale	To be specified
Temporal scale	To be specified

There are some limitations of the discussion on resilience mechanisms. The first limitations has been mentioned above concerning the lack of any framework for larger scales. The second limitation is the relation of resilience mechanisms to the discussion of invasive species. Invasive species are one of the most important factors in global change (Sala *et al.* 2000) and of high concern in ecological research (Mack *et al.* 2000; Heger & Trepl 2003). Despite this importance, theory about invasive species is widely disregarded within resilience discourse. For example, most theory on resilience mechanisms focuses on the effects of the loss of (specific) species. However, many management situations at a local, landscape or regional scale are characterized by the *addition* of species due to the occurrence of invasive species (Sax and Gaines 2003). This is an important area for further research.

3.3.3.3 The List of Resilience-Conducive Properties and Mechanisms

The overall purpose of the chapter about "resilience mechanisms" has been to find a list of potentially-conducive resilience properties or mechanisms that are backed-up by empirical and theoretical evidence to a certain degree.[148] Before I turn to the list it

[148] Naeem notes that deep understanding of the relation between biodiversity and the amount and "stability" of ecosystem processes necessitates to question fundamental issues in ecology. Those are: "What constitutes a

is necessary to introduce the distinction between "property" and "mechanism". I understand in the following *property* as a particular structural or functional characteristic of an ecological unit. In contrast, a *mechanisms* refers to a "direct interaction that results in a phenomenon" (Pickett *et al.* 1994: 39). Apparently, some of the resilience *mechanisms* explored above, may better be termed "resilience *properties*". Thus, in what follows I list several properties and mechanisms that are conducive for the ecological resilience of ecosystems to a certain disturbance regime, which are described in *Table 6*.

Table 6: An open list of resilience properties and mechanisms;
P: property, M: mechanism

Concept	**Type**
population diversity	P
functional important types of species	P
amount of ecological redundancy & response diversity	P
critical functional groups	P
local functional compensation	M
concentrations of biological legacies	P
source habitats	P
Connectivity	P
regional functional compensation	M
intermediate, pulse disturbances at a small spatial and temporal scale	M
value of controlling abiotic variables	P

These are: (1) *population diversity*, which includes (*a*) *population richness*, i.e. the number of populations of a species in a given area, (*b*) *population size*, i.e. the number of individuals per population, (*c*) *spatial distribution* of the population under study, i.e. the extent of the populations relative to their maximum in a defined area

diversity effect? What constitutes a mechanism? How is evidence marshalled to support invoked mechanisms? What constitutes a parsimonious explanation?" (Naeem 2003: 619).

and population dispersion and (*d*) *genetic diversity* within and among populations (Luck *et al.* 2003; Reusch *et al.* 2005); (2) *the maintenance of functional important types of species and their specific life-history traits*, such as real and potential drivers, keystone process species, keystone species, ecological engineers or dominant species (Grimm *et al.* 1999; Ernest & Brown 2001a; Fahrig 2001; Hooper *et al.* 2002; Folke *et al.* 2004; Hooper *et al.* 2005; Bulleri & Benedetti-Cecchi 2006); (3) *the amount of ecological redundancy*, i.e. species that perform the same ecosystem process, *and response diversity*, i.e. ecologically redundant species showing different responses to disturbances, *within functional effect groups* (Woodward 1994; Tilman 1996; Peterson *et al.* 1998; Trepl 1999; McCann 2000; Hooper *et al.* 2002; Hughes *et al.* 2002; Ives & Hughes 2002; Elmqvist *et al.* 2003; Loreau *et al.* 2003a; Symstad *et al.* 2003; Hooper *et al.* 2005; Srivastava & Vellend 2005; Nyström 2006); (4) *the maintenance of critical functional groups*, i.e. those functional effect groups that have low ecological redundancy and whose lost results in a dynamic regime shift (Mooney *et al.* 1996; Bellwood *et al.* 2004; Micheli & Halpern 2005); (5) *local functional compensation*, i.e. the tendency of coexisting, competing species within a patch to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition (Vinebrooke *et al.* 2004); (6) *the concentrations of biological legacies within the disturbed area*, i.e. organisms and organic structures that survive a disturbance event and serve as foci for regeneration and allow species to colonize (Franklin & MacMahon 2000; Elmqvist *et al.* 2002); (7) *the existence of source habitats in the vicinity of the disturbed patch* (Loreau & Mouquet 1999; Moberg & Folke 1999; Nyström *et al.* 2000; Nyström & Folke 2001; McClanahan *et al.* 2002; Mouquet & Loreau 2002; Bengtsson *et al.* 2003; Cadotte 2006); (8) *appropriate levels of connectivity within the meta-community, i.e. (a) small to moderate distances and (b) intermediate (but not high) dispersal rates between habitat patches* (Fahrig 2002, 2003; Loreau *et al.* 2003a; Leibold *et al.* 2005; Cadotte 2006); (9) *an intermediate degree of regional heterogeneity* (Mouquet *et al.* 2006); (10) *regional functional compensation*, i.e. the tendency of coexisting, competing species at a landscape scale to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition (Vinebrooke *et al.* 2004); (11) *the occurrence of intermediate, pulse disturbances at a small spatial and temporal scale* (Bengtsson *et* *al.* 2003; Shea *et al.* 2004); and (12) *the value of controlling abiotic variables* (Carpenter & Cottingham 2002; Gunderson & Walters 2002; Jansson & Jansson 2002; Walker & Abel 2002).

This list is the result of extensive study of the relevant literature. Yet it is by no means complete or comprehensive. There are two reasons for this. First, the terms “stability” and “ecological resilience” are highly slippery, as both terms are used vague and ambiguously for fundamentally different meanings at different levels of description. Thus, various mechanisms may be suggested from different perspectives. Second, the literature on stability or resilience mechanisms is but extensive. It has been impossible for me to take into account every relevant voice of the discussion within this thesis. Hence, several important mechanisms may be added to the list by others.

3.3.3.4 A Note on ‘Resilience Mechanisms and Philosophy of Science’

Is it possible to explicate the mechanisms that are most important for the emergence of ecological resilience? As considered in section 2.4, within ecology it seems rather pointless to search for *the* single cause or mechanism to explain a given phenomena. In many cases, the presupposition of simple causation has in fact impeded progress (Hilborn & Stearns 1982). Rather, several theories will be necessary to find an appropriate explanation (Mayr 2000, 2007). A growing consensus in the philosophy of science of ecology suggests that there are no single but multiple and overlapping causes that are not entirely separable (Hilborn & Stearns 1982; Levin 1992; Pickett *et al.* 1994; Holling & Allen 2002; Paine 2002).

As we have considered in the previous section, there are various properties and mechanisms that show responsible for the emergence of ecological resilience. Furthermore, these properties and mechanisms occur at different levels of description, i.e. the population (e.g. regeneration traits, genetic diversity), the community (e.g. ecological redundancy, response diversity) or the meta-community (e.g. intermediate connectivity and heterogeneity). Thus, the message of this chapter is that no single mechanism is *the* cause for ecological resilience, rather it is the interplay between the variety of mechanisms occurring at different levels of description that is responsible for the emergence of ecological resilience at the ecosystem level. These mechanisms may be regarded as “(biophysical) components” of ecological resilience (Walker *et al.* 2002). Using the classification by Reuter *et al.*

(2005) (cf. section 2.4), ecological resilience can thus be understood as emergent and connective property of ecological systems, in that it results from the interaction of many different properties and mechanisms at lower levels of organization.

3.4 Operationalization

The question whether or not it is possible to operationalize resilience has been considered as highly relevant for sustainability science by numerous authors (e.g. Kopfmüller *et al.* 2001; Ott 2001). This section comprehensively investigates the possibilities to operationalize the concept of ecological resilience. At first, section 3.4.1 clarifies the term “operationalization”. Subsequently section 3.4.2 examines the components of a resilience analysis. Hereby, section 3.4.2.1 describes the of-what part that identifies what exactly ought to be resilient. Subsequently, section 3.4.2.2 pictures the to-what that specifies the disturbances a certain ecosystem ought to be resilient to, and section 3.4.2.3 elucidates the concrete measures for indicating ecological resilience empirically, which includes the threshold approach (section 3.4.2.3.1), the self-identity approach (section 3.4.2.3.2), the resilience mechanisms approach (section 3.4.2.3.3) and several alternative approaches (section 3.4.2.3.4). Finally, section 3.4.2.4 explores the options to generalize statements about ecological resilience by means of disturbance scenarios.

3.4.1 What Does 'Operationalization' Mean?

For any scientific concept the *possibility for operationalization* is one of the criteria to judge its usefulness (Jax 2002: 14). According to Pickett *et al.* (1994: 85ff), clear modes of operationalization indicate a mature theory. Yet what does "operationalization" mean?

According to the *Cambridge Dictionary of Philosophy*, *operationalism* represents a program in philosophy of science that aims to interpret scientific concepts via experimental procedures and observational outcomes (Audi 1995). Similarly, the *Encyclopedia of Philosophy* defines operationalism as "a program which aims at linking all scientific concepts to experimental procedures and at cleansing science of operationally undefinable terms, which it regards as being devoid of empirical meaning" (Edwards 1967: 543). Operationalism as a concept was first introduced by Bridgman in 1927. Bridgman proposed that every scientifically meaningful concept must be capable of full definition in terms of performable physical operations and that a scientific concept is nothing more than the set of operations entering into its definition (Edwards 1967: 543f).

Within current philosophy of science however, operationalism in its original meaning is regarded as out-dated (Edwards 1967: 543f). Therefore, Jax (2002: 126ff) distinguishes operationalism in its original meaning *sensu* Bridgman from the *possibility to operationalize* ('*Operationalisierbarkeit'*), which is defined rather broadly as the possibility to identify inter-subjectively for a concrete situation whether a term applies to empirical reality. Such a broad concept of operationalization is considered to have high scientific value within current philosophy of science (Pickett *et al.* 1994; Poser 2001).

Regarding the concept of resilience, an operational definition is achieved by relating the concept to a series of defining operations, and then demonstrating that such an entity so defined can play an important role in theory (Peters 1991: 77). In the following sections this thesis follows the broad concept of operationalization and understands *operationalization* as a generic concept to summarize the terms "measurement", "estimation" and "indication".

3.4.2 Resilience Analyses

The concept of ecological resilience is operationalized by means of a *resilience analysis*. A resilience analysis is comprised of four parts. Those are: (1) the *of-what part*, that specifies what exactly ought to be resilient; (2) the *to-what part*, that identifies the disturbances a certain ecosystem ought to be resilient to; (3) measures for the empirical indication of ecological resilience; and (4) disturbance scenarios. *Table 7* provides a preview of the most important issues treated within these parts. I will investigate each part in the following.

Table 7: Conceptual framework for a resilience analysis comprised of four parts

No.	Name of part	Description
1	*Of-what part*	(a) What nature we ought to and want to preserve where, for whom and for what reasons? (b) What ecological units are being considered?
2	*To-what part*	To what kind of disturbance regime the system ought to be resilient to?
3	*Indication of ecological resilience surrogates*	(a) threshold approach (b) self-identity approach (c) resilience mechanisms approach
4	*Scenario building*	Is the system resilient across several likely disturbance scenarios?

3.4.2.1 The Of-What Part

The of-what part of a resilience analysis specifies the objects (and interrelations) in nature that ought to be resilient to certain disturbances. This enterprise seems to be rather straightforward. But in fact it is a complicated undertaking that is comprised of several steps. To put it bluntly: this section is about specifying *what exactly* is desirable and, hence, ought to be resilient *and* (!) *what exactly* is not desirable and, hence, ought to be prevented. That means, the concept of ecological resilience does not relieve us from reflecting on which nature we want to preserve. The following steps are essential for this concern.

3.4.2.1.1 Step 1: Selecting Ecosystem Services and Scale

The first step refers to the question *which* nature (or parts of nature) we want to protect. This seems obvious but the answer is far from being clear. An answer requires first of all a practical-political agreement on the goals we commit ourselves to: which nature we *ought to* preserve? According to Honnefelder (1993), this question has four fundamental aspects. Those are: (1) the *temporal aspect*: which "state" of nature we ought to preserve, e.g. the state around 1400 AD or the current; (2) the *regional aspect*: which nature do we mean - the almost untouched wilderness areas in the German Alps or the *Englischer Garten*, a big park in Munich; (3) the *qualitative aspect*: what do we want to preserve – peaceful social conditions (e.g. nature for recreation), healthy circumstances (e.g. clean air and water) or aesthetic enjoyment (e.g. biodiversity at the species and the ecosystem level); and (4) the *social aspect* - for whom do we want to preserve nature, e.g. the extremely vulnerable[149] communities, the property holders, the whole humanity or the future generations (Honnefelder 1993).

Thus, the decision what is desirable or undesirable, respectively, in nature is a complex task. Hereby, it is important to distinguish between *assessment* and *evaluation*. The former signifies whether a statement is true or not (referring to questions such as 'how many species are there?' or 'is this species really indigenous or endemic?'), whereas the latter refers to specific *values* inherent in a statement (Eser & Potthast 1997). Apparently, the specification what we want to preserve requires to acknowledge not only an ecological assessment but also the weighing of philosophical (ethical), political and broader cultural, social and economic *values* (Honnefelder 1993; Jax & Rozzi 2004).[150] Values enter the determination of

[149] *Vulnerable communities* are groups of people that are exposed to and susceptible to a high amount of disturbances and global change. Confer Adger (2006) for a detailed review of the concept of vulnerability.

[150] Eser and Potthast (1997) distinguish between six levels of evaluations ('*Bewertungen*') concerning nature conservation. Those are: (1) '*Naturschutzfachliche Bewertung'*, i.e. evaluation of specific areas with ecological

conservation goals in many different ways, e.g. in our images of nature, in our economic values, in our political preferences, in our moral and aesthetic attitudes towards human and non-human nature, and even in our decisions about what is important in science (Jax & Rozzi 2004).

Thus, the question 'what nature we want to preserve' is essentially at the interface between societal decisions and scientific knowledge (Jax & Rozzi 2004). This interface is comprised of several components, e.g. reasonable ethical judgements[151] with respect to both eudaimonistic values[152] and moral values[153] (Eser & Müller 2006), the explicit specification of ecological units[154] (Jax *et al.* 1998; Jax & Rozzi 2004; Jax 2006) and the weighing and ordering of criteria and goals (Honnefelder 1993; Jax & Rozzi 2004; Dietrich 2006). It is far beyond this thesis to describe the interface between societal decisions and scientific knowledge in any detail.

Yet one promising approach for a comprehensive evaluation of different values refers to the concept of ecosystem services and I will expand on this concept in the following.[155] Developed and defined by Gretchen Daily as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" (Daily 1997: 3), the world's ecosystem services have recently gained much attention in the comprehensive study *Millenium Ecosystem Assessment* (MEA) called for by former United Nations Secretary-General Kofi Annan in 2000 and carried out by more than 1300 experts worldwide (UNEP 2005a).[156] In this thesis, *ecosystem services* are defined as "the benefits that people obtain from ecosystems" (Bennett *et al.* 2005b), as this definition is characterized by clarity and brevity.

criteria; (2) '*Politisch-administrative und juristische Bewertung'*, i.e. the political and legal consideration of social, economical, and broader societal aspects, (3) '*Naturwissenschaftliche Bewertung'*, i.e. evaluation of the consistency and validity of the basic theory, concepts and terms with respect to criteria, (4) '*Praxisorientierte Bewertung'*, i.e. evaluation of the applicability of goals and criteria, (5) '*Strategische Bewertung'*, i.e. evaluation of the feasibility of measurements concerning nature conservation with respect to tactical and pragmatic concerns; and (6) *Ethische Bewertung*, i.e. evaluation of the goals and criteria concerning nature conservation from an ethical perspective.

[151] Dietrich (2006) unfolds a sound approach for ethical judgements with respect to conservation problems.

[152] Eudaimonistic values refer to the achievement of individual happiness.

[153] Moral values refer to the physical or psychological integrity of particular entities.

[154] For a detailed description of the specification of the self-identity of ecological units confer section 3.4.2.1.2.

[155] It is controversial whether or not the concept of ecosystem services includes moral values and takes into account the *demarcation problem*, i.e. what entities in nature ought to be considered morally (Krebs 1997).

[156] Some scholars view the emergence of the concept of *ecosystem services* as the new "rising star" among the boundary objects in environmental discourse, which might complement or substitute the former *biodiversity* (Kurt Jax, *personal communication*). For a detailed treatment of ecological resilience as a boundary object confer section 4.1.4.

The MEA subdivides ecosystem services into four types. Those are: (1) *supporting services*, e.g. nutrient cycling, soil formation and primary production; (2) *regulating services*, e.g. climate regulation, flood regulation, disease regulation, water purification; (3) *provisioning services*, e.g. food, fresh water, wood and fiber, fuel, and (4) *cultural services*, e.g. aesthetic enjoyment, spiritual, educational and recreational values. Hereby, the supporting services can be viewed as an essential background enabling the continued performance of the other types of ecosystem services.

The important point for my argument is that the concept of ecosystem services includes not only ecological criteria but also economic values (e.g. provisioning services, such as food, timber), social values (e.g. fresh water or food for vulnerable communities) and cultural values (e.g. cultural services, such as aesthetic enjoyment, spiritual values). Within the ecosystem services approach the question 'what nature to preserve' is still a melange of different values and no easy task.

To make things even more complicated, there may be trade-offs between different ecosystem services (Rodriguez *et al.* 2006). Individual ecosystem services may be thought of as different elements of an interrelated whole or "bundle". Attempts to optimize a single service often lead to reductions or losses of other services – in other words, they are traded-off (Holling & Meffe 1996). As human societies continue to transform ecosystems to obtain greater provision of specific services, we will undoubtedly diminish some to increase others (Foley *et al.* 2005). Ecosystem services trade-offs may occur (i) in space, (ii) in time, and (iii) across different ecosystem services (Walker *et al.* 2002; Rodriguez *et al.* 2006).

Ecosystem services trade-offs in space refer to effects of a management decision that are borne by others than those benefiting from the targeted ecosystem service, termed *externality* in environmental economics. For example, the highly productive, intensive agriculture within the USA (and other industrialized countries) relies on the addition of either natural or chemical fertilizers. The effects of the high level of artificial fertilization have resulted in massive changes in downstream areas. The cumulative effect of small scale fertilization by many individual farmers has been the creation of a hypoxic ("dead") zone in the Gulf of Mexico. The dead zone has resulted in declines in the shrimp fishery, as well as in other local fisheries in the Gulf region. That is, attempts to maintain and increase the provision of one service, food, have caused substantial declines in many ecosystem services in another location (Rodriguez *et al.* 2006).

Ecosystem services trade-offs in time refer to the fact that management decisions often focus on the immediate provision of ecosystem services, at the expense of this same ecosystem service or other services in the future. A striking example is the focus on provisioning services and the loss of ecological resilience at the same time.

Finally, different ecosystem services may be in conflict within the same region. For example, *ecosystem services trade-offs across services* exist with respect to different forms of forest management. The management of a forest for tree production may also affect water quality downstream (a regulating service) or decrease the value of the land for recreation (a cultural service). Often, trade-off decisions show a preference for provisioning, regulating or cultural services (in that order) and neglect supporting services (Rodriguez *et al.* 2006).

The important point for my argument here is that it is essential to specify what ought to be preserved: *which ecosystem services* at *what temporal and spatial scale* (Walker *et al.* 2002; Brand 2005: 130ff).[157] This specification is relatively easy to make if ecosystem services are to be sustained for the survival of humans, as in some management situations in developing countries. In these cases, the priorities are relatively clear. The task becomes more complicated if the preservation of ecosystem services is to be achieved for a good and meaningful human life ('*gutes menschliches Leben*'), as in many management situations in industrialized countries. In these circumstances, there is no alternative (in democratized countries) to a societal discourse about different life forms with respect to nature and the environment (Honnefelder 1993): what nature ought to be preserved where, for whom and for what reasons?[158]

Ideally, an answer to this question for concrete environmental management situations is found by means of *participative and collaborative methods* (Walker *et al.* 2002; Berkes 2004; Carlsson & Berkes 2005). For example, *collaborative management* or *co-management* represents a governance system that combines state control with local, decentralized decision making and accountability and combines the strengths and mitigates the weaknesses of each. It is a partnership in which government agencies, local communities and resource users, non-governmental organizations and other stakeholders negotiate, as appropriate to each

[157] If the concept of ecosystem services does not include moral values, the of-what part would be merely utilitarian. Yet in the following I assume that the ecosystem services-framework allows for the consideration of moral values as well.

[158] This question is more or less the topic of environmental ethics (e.g. Krebs 1997).

context, the authority and responsibility for the management of a specific area or set of resources (cf. for a review of the definitions Carlsson & Berkes 2005). That is, the decision about 'what nature we ought to and want to preserve' is made by stakeholder assessments and practical discourse.[159]

In order to develop an appropriate decision some tools from environmental planning may be used, e.g. development of goal targets ('*Zielentwicklung*') or model development ('*Leitbildmethode*'). By means of these tools it can also be made explicit what development and states are undesirable for a particular region (von Haaren & Horlitz 2002). Note that it is again far beyond this thesis to draw a comprehensive picture of the participative process necessary for decisions in environmental management. The important point is that an answer to the question 'what nature we want to preserve', and thus, the question 'what parts of nature ought to be resilient' requires a practical discourse that balances an ecological assessment with ethical, social, economic and cultural values.

3.4.2.1.2 Step 2: Specifying the Self-identity

Suppose a stakeholder group that has achieved a decision about the maintenance of certain ecosystem services at a specific temporal and spatial scale. Correspondingly, these ecosystem services ought to be resilient over the long run. In general, ecosystem services are generated by ecological units, such as populations, communities or ecosystems. Thus, one might infer that the preservation of services does only mean to preserve the ecological units and their performance at the spatial and temporal scale of concern. However, ecological units can principally be viewed in infinite ways and it is crucial to specify exactly what we mean by an ecological unit with respect to a given research question (Jax *et al.* 1998; Jax 2006). That means, the next step in the of-what part of a resilience analysis is to specify exactly what ecological unit is meant to be resilient *and* what ecological units ought to be prevented.[160] Hereby, the concept of *identity* and/ or *self-identity* is relevant.

The concept of (ecological) *identity* has been recently introduced in resilience reseaech by Cumming and Collier (2005) and extended by Cumming *et al.* (2005).

[159] The concept of practical discourse is closely related to the concept of transdisciplinarity (e.g. Hirsch-Hadorn *et al.* 2006; Scholz *et al.* 2006).

[160] This step is related to and interacts with the first step within the of-what part (K. Jax 2007, *personal communication*).

Cumming *et al.* (2005) propose that the identity of whole social-ecological systems includes (1) components that make up the system, (2) the relationships between components and (3) the ability of both components and relationships to maintain themselves continuously through space and time. The identity of complex systems also comprises (4) innovation and self-organization within a system. The concept of identity is then used to operationalize ecological resilience step by step (Bennett *et al.* 2005a). According to Cumming and Collier (2005) however, the treatment of identity within resilience discourse remains partial and should be subject of ongoing discussions.

To further this discussion, this section applies the methodology of self-identity developed by Jax *et al.* (1998) and Jax (2006) to the resilience approach. The concept of self-identity is a way to delimit and specify ecological units, such as population, community or ecosystem. The methodology of self-identity allows to formulate the criteria to specify *any* ecological unit of concern. In contrast to the proposal made by Cumming *et al.* (2005) it emphasizes variables that can be operationalized easily. Operationalization is rather problematic when a definition of a system's identity includes variables as "innovation" and "self-organization", which need clear definitions in themselves before they can be judged as useful or not (K. Jax, *personal communication*).

In order to illustrate the self-identity approach of Jax *et al.* (1998) and Jax (2006) and its usefulness for operationalizing ecological resilience I now turn to an example frequently investigated within resilience research, namely rangelands. According to Walker (2002), both key variables such as annual grasses, perennial grasses and woody plants (trees and shrubs) as well as critical drivers such as rainfall, fire and grazing are seen to control ecosystem dynamics. Sustained grazing pressure by herbivores and lack of fire reduces the competitive effect of grasses on shrubs, leading to increasing woody plants. Periods of drought with high stock numbers bring about death of perennial grasses, leading to reduced grass cover. When followed by a high-rainfall season, this leads to a profusion of new woody plants. If grazing pressure is maintained, there is a threshold in the increasing woody-to-grass biomass ratio when the competitive effect of the woody plants prevents the build-up of sufficient grass fuel to carry a fire. The system then stays in the woody regime for several decades until woody plants begin to die, opening up the system for increased grass growth and the reintroduction of fire (Walker 2002; Walker & Abel 2002).

Hence, two alternative stable regimes occur in rangelands of this type. The first regime matches a grasses-woody plants mix, which is desirable for pastoralists to produce commercial livestock whereas the second regime corresponds to a woody plant-dominated state, which is highly undesirable for pastoralists (Walker 2002). That is, the grassy regime is desirable owing to the provision of ecosystem services for pastoralists, e.g. the production for livestock.

What is the self-identity *sensu* Jax (2006) of the desirable regime identified in this case? There are four dimensions of self-identity that have to be specified. *(1) If the boundary of the regime is defined topographically or functionally*: the boundary is defined topographically with respect to the ecosystem type (semi-arid and arid rangeland). *(2) The expected internal degree of relationships*: there is high competition between each annual grasses, perennial grasses and woody plants as well as herbivory of livestock on both annual grasses and perennial grasses. Moreover, fire strongly affects the abundance of woody plants. That is, the degree of internal relationships is high. *(3) The set of selected elements:* selected elements represent annual grasses, perennial grasses, woody plants (trees and shrubs) and herbivores (predominantly domestic livestock) as well as the interactions mentioned above. Therefore, an intermediate amount of elements is considered. *(4) The degree of component resolution*: the component resolution is fairly low as only floristic life forms and kinds of herbivores are taken into account. At first, neither individual species nor species composition but rather the ratio between plant life forms play a crucial role.

The sum of these four criteria represent the minimum of what has to be communicated if an ecological unit such as an ecosystem is to be investigated. Principally, the last three criteria, that is, internal relationships, selected phenomena and component resolution, represent gradients as there are various possible degrees for each of them. They can be arranged on axes and then assembled into a three-dimensional model which lays out a volume in which any ecological unit can be localized. The resulting *SIC-model* (**S**elected elements, **I**nternal relationships, **C**omponent resolution) represents a very efficient way to communicate explicitly about an ecological unit being focus of a particular study (Jax 2002). *Figure 14* illustrates the SIC-model for the rangeland example explored above.

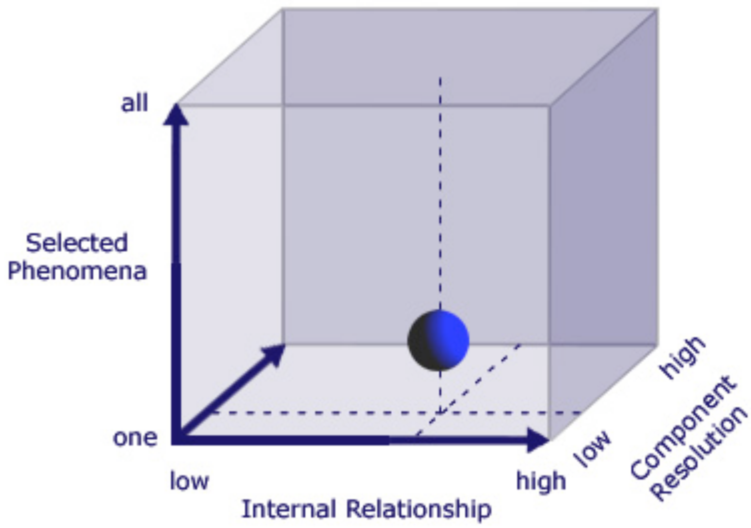

Figure 14: The SIC-model in the case of a rangeland as specified by Walker (2002)

There is an important point here. The *self-identity approach* assumes that concepts of ecological units - as abstractions that select particular features of reality, which are of interest to an observer - can neither be true nor false, but only useful (i.e. appropriate) or useless (i.e. inadequate) (Jax 2006). So we may ask: is the self-identity specified in the case of the rangeland example useful concerning the research question?

First, the research question proposed by Walker (2002) corresponds to *keeping the ecosystem in the desired regime or basin of attraction*, namely the grassy regime. More particular, it corresponds to the *maintenance of the ecosystem services* required for the land-users, in this case the productivity of grass species for pastoralism. Second, as illustrated in *Figure 14*, the self-identity of the grassy regime is specified as follows: the selected elements are at low to medium level, the internal relations at medium to high level and the component resolution is quite low. Is this specification appropriate concerning the research question?

As indicated by Jax and Rozzi (2004) and Jax (2006), such a specification of the self-identity (selected elements: medium level; internal relations: at least medium level; component resolution: medium to low) is highly useful if environmental management aims at sustaining the provision of ecosystem services. As Jax notes:

"If ecosystem management has, in contrast, the aim of maintaining ecosystem services, the appropriate definition consequently will be different. The component resolution will be lower (on the level of functional groups to species), internal relations will be at least at medium level (some regulation or constancy required, specific process rates secured), the selected variables also at medium level, and the boundaries should be process related. This ecosystem can be considered intact even if some species become extinct or are replaced." (Jax 2006: 253).

Thus, the case of Brian Walker's rangeland (2002) is a good example for an useful specification of the self-identity assumed that the research question is to preserve essential ecosystem services. This has larger consequences for research on resilience issues. In my view, within resilience discourse, the aim of environmental management is generally to maintain (desirable) ecosystem services in the face of human impact and natural fluctuations. This is, in fact, one of the definitions proposed for ecological resilience (cf. ecosystem services-related definition in section 4.1.2). Thus, to meet this research question the *self-identity within resilience discourse ought to be specified as follows: selected elements: medium level; internal relations: at least medium level; component resolution: medium to low; boundaries: process related*.[161] This would be an useful specification for the self-identity.

To sum up: the important points of the previous sections are (*a*) using the methodology of self-identity enables scientists to exactly specify which ecological unit(s) ought to be resilient for a given temporal and spatial scale and for a given purpose (e.g. preservation of specific ecosystem services). In addition, (*b*) it is crucial to specify the ecological units of interest in a useful way in order to meet the given research question.

There is another important point here. The of-what part of a resilience analysis necessitates to specify also the states of the ecological units that are *not* desirable. It is crucial to identify the amount of change in the ecological units that would constitute a new system, which is at the same time undesirable (Cumming *et al.* 2005; Jax 2006). That is, it is essential to specify the "bounds of desirability" and the "bounds of undesirability". Correspondingly, the resilience analysis then refers to the bounds of desirability.

Within resilience research, the decision about what is desirable or undesirable refers to the concept of alternative stable regimes, explored in detail in section 3.2.3. In the

[161] According to K. Jax (*personal communication*), this is only a loose approximation, which has to be specified more adequately for specific cases.

case of Walker's rangeland described above the desirable states are given by the grassy regime while the undesirable states are made up by the woody regime. Thus, within resilience discourse, the bounds of desirability is given by the distance of the current position of the regime to the ecological threshold occurring "between" the alternative stable regimes. Correspondingly, the resilience analysis refers to the desirable regime only.

Yet there are some obvious shortcomings to this notion. First, as I have considered in section 3.2.3.1, *many* ecosystems show alternative stable regimes but *not all* (Schröder *et al.* 2005). In particular, ecosystems that are controlled foremost by competitive forces, e.g. forest or coral reefs, are not as likely to exhibit alternative stable regimes as those that are controlled strongly by environmental adversity, e.g. savannahs or lakes (Didham & Norton 2006). That means, the bounds of undesirability are not always given by the undesirable regime as such, simply because in some ecosystems there is no alternative stable and undesirable regime. Second, changes to the ecological units of concern may be undesirable even if there is no shift to an alternative stable and undesirable regime. For example, in a forest ecosystem managed for tree production some ecosystem services, such as the water quality downstream or the value of the land for recreation, may be lost before the forest shifts to an alternative stable regime that is characterized by different tree species. How to specify the bounds of desirability and undesirability, respectively, in these cases?

Alternatively, there is a second option to identify the amount of changes that would constitute a new and undesirable system "state" that is related to the concept of self-identity explored above (Jax 2006). Similarly to ecological thresholds in state space, the scientist must specify the acceptable deviation from the original self-identity of the regime with respect to the research question. Hereby, the self-identity identified corresponds to the desirable *baseline condition* and the acceptable deviation may be evaluated by some criteria that are connected to the question 'what nature we ought to preserve' described in step one of the of-what part in section 3.4.2.1.1 (Jax & Rozzi 2004). Beyond this acceptable deviation a system would lose its self-identity and constitute a new system (Jax *et al.* 1998; Cumming *et al.* 2005; Jax 2006).

Take the example of the forest managed for tree production. If the ecosystem services except tree production, e.g. water quality downstream or recreation, ought to be maintained, a system state that does not provide these services would be undesirable. In this case, the acceptable deviation from the baseline condition is related to the question whether or not the system state provides the desirable ecosystem services.

Certainly, decisions concerning the bounds of desirability and undesirability are dependent on value judgements made in step one of the of-what part of the resilience analysis. They therefore may appear in a sense arbitrary. But as considered in step one of the of-what part, the decision on what is desirable or undesirable depends on a practical discourse and stakeholder assessments, i.e. it is ideally made for good reasons. Correspondingly, the resilience analysis specifies the bounds of desirability with respect to a research question (Jax 2006) or problem definition (Bennett *et al.* 2005a).

3.4.2.2 The To-What Part

The to-what part of a resilience analysis specifies the disturbances to which a certain ecosystem is to be resilient to (Carpenter *et al.* 2001; Walker *et al.* 2002). This is a crucial step in the analysis, assumed that for most management situations there is probably no general resilience, i.e. the general capacity of a system that allow it to absorb unforeseen disturbances, but rather a *specified resilience*, i.e. the resilience of a specified system to a *specified* disturbance regime (Walker & Salt 2006: 120f).

The term “disturbance” refers to a complex scientific concept. As I have considered in section 3.3.2.3.1, a relative definition of disturbance distinguishes between three types of disturbances. I will repeat these types shortly in the following. The first type, (1) *pulse disturbances*, are natural disturbances that occur episodically followed by a reorganization phase. Examples include small-scale fires in savannahs or insect outbreaks in forest ecosystems. In general, this type of disturbance is conducive for ecological resilience. The second type, (2) *large, infrequent disturbances* are uncommon surprises that have a frequency so low that most organisms are unlikely to adapt to them. Examples are volcanic eruptions, large-scale fires or extreme floods. Finally, the third type, (3) *press disturbances*, are chronic stress agents that often are of anthropogenic origin. Most species have not been able to respond evolutionary to anthropogenic disturbances of this type. Examples include intense monoculture agriculture, heavy and continued grazing or ongoing pollution events.

Moreover, it is essential to distinguish between a single *disturbance event*, i.e. a relatively discrete disturbance in time, and the *disturbance regime*, i.e. the distribution and characteristics of disturbance events in space and time (Jax 1998/99). The disturbance regime is characterized by many variables but the ones most important are frequency, intensity, size of the areas affected and spatial distribution (Jax 1998/99).

A disturbance regime may include c*ompounded disturbances* that deviate the normal, pulse dynamic of disturbances and are of high magnitude or have a higher frequency that disrupts the normal cycle of disruption and recovery within communities (Paine 2002) and *synergistic disturbances*, which represent various different disturbances that occur at the same time and have inhibitory synergies. Examples are the multiple stressors of coral reefs, e.g. fishing, pollution and sedimentation (McClanahan *et al.* 2002). Both kinds of disturbances may have multiplicative rather than additive effects on the ecological unit in question (Vinebrooke *et al.* 2004).

What is the focus of an operational concept of ecological resilience regarding the concept of disturbance? As I have considered in section 3.3.2.3.1, the resilience approach and also this thesis uses a relative definition of disturbance. Furthermore, the first type of disturbance, natural pulse disturbances are generally viewed as conducive for ecological resilience. Therefore, ecological resilience should be applied to press or large-infrequent disturbances only. In addition, most situations in land-use are characterized by compounded and/ or synergistic disturbances. Therefore, *the focus of a concept of ecological resilience are compounded and/ or synergistic press and/ or large-infrequent disturbances*.

There is another important point concerning disturbances. It is crucial to specify the *type of disturbance regime*. This is essentially so for two reasons. First, different types of disturbances can have entirely different effects on the same system. Take for example a forest ecosystem. Disturbances such as acid rain or bottom ozone concentrations cause damage foremost to the roots and the leaves of the trees that might affect the whole tree if damages are severe enough, whereas massive insect outbreaks literally destroy a whole forest area, as experienced in the *Nationalpark Bayerischer Wald*, in Bavaria, Germany. Second, in order to resist or respond to these different effects ecosystems require completely different resilience (-conducive) properties or mechanisms. Resistance to acid rain and ozone refers by foremost to the properties and regeneration traits of the key process species or dominant tree species, whereas the buffering of an enormous insect outbreak requires the existence of source habitats in the vicinity of the disturbed patch and intermediate connectivity between the patch, respectively, in order to re-colonize the disturbed patch after the disturbance. That is, it is entirely decisive to specify the kind of disturbance regime an ecosystem ought to be resilient to.

A further important point is to evaluate for concrete situations whether a certain disturbance ought to be included in the analysis or not. For example, if we consider a coral reef in the Caribbean we obviously should include disturbances such as pollution pulses from sewages emitted by some nearby villages or fishing pressure by the local fishing industry, but should we include scenarios of climate change? This problem refers to multiscale scenarios, which will be the topic of section 3.4.2.4.

3.4.2.3 Empirical Indication of Ecological Resilience

The third part of a resilience analysis examines the methods developed to measure ecological resilience empirically. The empirical estimation is the “keystone” of a resilience analysis because it is in this part resilience is actually operationalized, i.e. estimated, indicated or measured. It rests on the previous parts, the of-what part and the to-what part, which together specify *what exactly* ought to be resilient *to exactly what*.

Ecological resilience is hard to measure. This may be related to the specific systems notion of resilience research. As will be considered in section 4.2.3.1, the resilience approach champions a holistic and organic systems notion. An ecosystem is viewed as a real entity (i.e. ontological realism) and to be similar to an organism. That means, ecosystems are viewed as “delimited wholes” that self-organize by means of the interactions of their components (e.g. populations). Yet it is highly contested whether ecosystems can be viewed in analogy to an organism, as there is a long-standing controversy regarding holistic and individualistic systems notions in ecology (confer section 2.4) (Trepl 1987; Jax *et al.* 1998; Jax 2006; Kirchhoff 2007; Voigt 2008).

Is is my impression that the discussion on the operationalization of ecological resilience in a sense unconsciously reflects this controversy. Scholars within resilience research realize a tension between a concept of general resilience and a concept of specified resilience. *General resilience* refers to a rather holistic systems notion and is defined as "the general capacity of a system that allow it to absorb unforeseen disturbances". In contrast, *specified* (or *targeted*) *resilience* refers to a rather individualistic systems notion and is understood as "the resilience of a specified system to a specified disturbance regime" (Walker & Salt 2006: 120f). Yet to my knowledge the resilience approach does not explicitly take into account the enduring controversy between holistic and individualistic systems notions in ecology.

In resilience research, the way out of this tension between holistic and individualistic systems notions is to focus on the empirical indication of ecological resilience by means of resilience surrogates that focus on specific components of the ecosystem. Resilience scholars realize that ecological resilience – understood as an "emergent property" of "ecosystems" – is elusive (cf. Carpenter *et al.* 2005). Consequently, recent work proposes that it is hard to measure ecological resilience directly and suggests using proxies that are derived directly from theory for use in assessing resilience. These proxies have been termed *resilience surrogates* by Carpenter *et al.* (2005) and Bennett *et al.* (2005a). The important point here is that these resilience proxies do not correspond to a real holistic measure of ecological resilience.[162] Rather than "measurement", in the following I therefore speak of the *indication* of ecological resilience.

All the approaches put forward to estimate ecological resilience and described in the following section are in a sense surrogates or proxies for ecological resilience. According to Carpenter *et al.* (2005) these surrogates should be consistent and repeatable, in the sense that independent observers given the same information would assess the surrogate in the same way. In addition, to prove whether the surrogates are consistent with the theory on ecological resilience, models, long-term observations and comparisons across case studies should go hand in hand.

3.4.2.3.1 The Threshold Approach

The first method to indicate the degree of ecological resilience of a specific ecosystem refers to the concepts of alternative stable regimes and ecological

[162] Sucb a holistic measure might be impossible anyway (Trepl 2005: 381, Jax 2006). The point I want to make here is that resilience research has long been searched for a holistic measure, but recently had to take into account that such a measure is impossible to find or at least elusive.

thresholds. This is a characteristic feature of “resilience thinking” (Walker & Salt 2006). As we have considered in detail in section 3.2.3.1, the assumption of the existence of alternative stable regimes is rather weak because, empirically, *many* ecosystems can exhibit alternative basins of attraction but *not all* (Schröder *et al.* 2005). The following approach to estimate ecological resilience is thus only useful for ecosystems that experience alternative stable regimes and ecological thresholds. I will term this method *threshold approach* in the following.
The threshold approach rests on the use of *models*. For a long time, ecology has not been too much concerned about modelling until the 1970ies when a sharp increase in research about models took place. Within resilience discourse, authors have largely used *state variable* or *top-down models* to examine ecological resilience. I will expand here a bit on the ecological research about models in the following to show some of the chances and shortcomings of the different model approaches.

3.4.2.3.1.1 Some Notes on the Use of Models in Ecology

Models are ‘purposeful representations’ and tools for problem-solving. The purpose of a model is to capture the essence of a problem and to explore different solutions of it (for further literature cf. Grimm 1999). Principally, models never picture the whole of ecological relationships in nature but focus on essential elements of a study object with respect to a certain research question. They help to bridge the gap from the *empirical notion* of alternative stable regimes and ecological thresholds for whole systems to a (*theoretical and/ or causal) analytical theory*. Thus, models may be viewed as a mediator between empirics and theory (Mittwollen 2001).[163]
Correspondingly, a critical first step in model development is resolving which mechanistic details are important for describing dynamics analytically and which details can be safely abstracted (Schmitz 2000). Theoretical ecology is often handicapped at this stage because resolving which details sufficiently describe dynamics requires a fairly complete empirical understanding of the way species interaction mechanisms are causally linked to long-term dynamics. In practice, this insight is difficult to obtain from empirical research because logistical constraints usually prevent observations and experiments from being conducted long enough to

[163] Note that the purpose of models is viewed in fundamentally different ways: (1) models enable the understanding, application and test of theories; (2) models are useful tools for theory development; (3) models mediate between empirics and theory; and (4) models are parts of scientific explanations (Mittwollen 2001).

evaluate the linkages between mechanisms and long-term dynamics. When faced with this dilemma, theoretical ecologists are often forced to make *educated guesses* about which variables of interest or mechanisms to include in their analytical models (Schmitz 2000). I will return to the problem of educated guesses with respect to the distinct model approaches below.

There are two fundamentally different approaches to modelling in ecology: (1) the top-down approach and (2) the bottom-up approach.

Introduced foremost by the Odum-brothers (Odum 1971, 1999), the *top-down approach* (e.g. *state variable models*) rests on the notion that ecosystems behave like gigantic, cybernetic systems (Jopp & Breckling 2001). Researches try to aggregate the complexity of ecosystems into a few state variables applying a general conceptual framework, e.g. stability concepts, such as ecological resilience, equilibrium or coexistence, to principally all kinds of ecological units. A characteristic feature of the top-down approach is that it deliberately abstracts from features of individuals, and hence, that it must comprise educated guesses about the variables and parameters that are to be included in the model (Schmitz 2000). This is a clear shortcoming of this approach. I will return to the top-down models below since this approach is highly prevalent within resilience research.

By contrast, the *bottom-up approach* starts with the parts (i.e. the individuals) of a system and then tries to understand how the system's properties (e.g. stability, coexistence) emerge from the interaction among these parts. Bottom-up approaches take into account important features of individuals, as they are driven by the suspicion that much of what we have learned from state variable models about theoretical issues, e.g. stability or regulation, would have to be revised if the discreteness, uniqueness, life cycles and variability of individuals were to be taken into account, as well as the fact that most interactions are local and that space matters (Grimm 1999).

A fruitful bottom-up approach refers to *individual-based models* (IBMs) (Grimm 1999; Schmitz 2000; Rademacher *et al.* 2004; Breckling *et al.* 2005). The IBM approach extends the potential of ecological models to cope with heterogeneity and complex ecological interaction networks with variable structures. It allows realistic descriptions and is highly adaptive to particular contexts and questions under investigation (Breckling *et al.* 2005). IBMs that are pattern-oriented identify multiple patterns or "phenomenological generalizations" (Cooper 1998) in the real system and then provide a model structure that principally allows these patterns to emerge in the model. As a result, the model will finally be ‘structural realistic’, i.e. contain the key structures and processes of the real system (Rademacher *et al.* 2004).

Are IBMs useful for ecological research? On the one hand, there are several advantages to using IBMs compared to state variable models. First, the appeal of using an IBM is that it can handle a rich amount of biological detail at a low level of organizational complexity (e.g. behavioural and physiological ecology of individuals). In other words: IBMs open up modelling to empirical knowledge because it is possible to integrate observations directly in the model (Grimm 1999). Therefore, it does not require extensive *a priori* assumptions about which elements and mechanisms of a system are causally relevant (Schmitz 2000). That is, the educated guesses are reduced to a very low degree. Second, IBMs can principally take into account individual variability and its potential significance for population dynamics. Third, IBMs are better testable than state variable models because they refer more often to certain species and certain biological mechanisms that can be studied in reality (Grimm 1999). On the other hand, individual-based modelling frequently has no reference to a conceptual framework and may ultimately lead to mere “stamp collection”, as numerous but unrelated characteristics of species are investigated (Grimm 1999).

Hence, both approaches to modelling are partial.[164] As illustrated in *Figure 15*, there is a problem with the “down path” of the top-down approach in that the concepts devised at the top-level are usually hard to test because they do not refer at all to the entities which comprise an ecological unit: individuals. But the “up path” of the bottom-up approach is also difficult to follow if one has no clear idea what questions to ask at the population level. And these questions are provided by the conceptual framework of the top-down approach (Grimm 1999). Thus, the two approaches to modelling should not been regarded as exclusive alternatives but rather as complementary approaches that are mutually dependent (Grimm 1999). This is an important point I will return to below.

[164] There is in fact a continuum of modelling approaches between the two poles of top-down and bottom-up in the relevant literature (Grimm 1999).

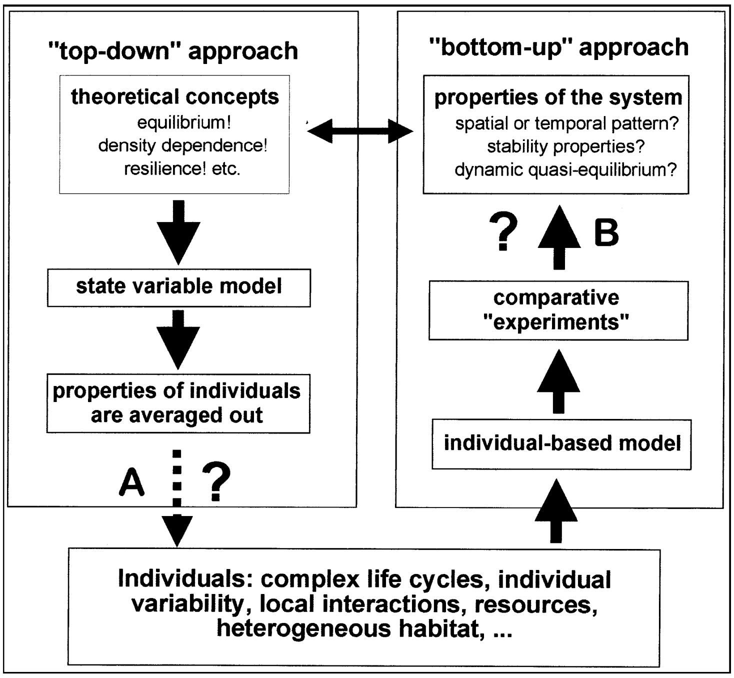

Figure 15: The complementarity of top-down and bottom-up modelling
from (Grimm 1999)

3.4.2.3.1.2 Bifurcation Diagrams

Regarding the threshold approach for estimating the amount of ecological resilience, top-down models are prevalent (Carpenter *et al.* 2001; Scheffer *et al.* 2001; Gunderson & Holling 2002; Gunderson & Pritchard 2002; Peterson *et al.* 2003b; Bennett *et al.* 2005a; Walker *et al.* 2006). As described in detail in section 3.2.2, it is assumed that important changes in ecosystem dynamics can be understood by analyzing a few, typically no more than five, *key variables*. This is referred to as the “rule of hand”, appealing to the five fingers of a human hand (Yorque *et al.* 2002; Walker *et al.* 2006). The key variables are, in turn, subdivided into *fast variables* and *slow variables* according to the turnover rates in space and time. The slow variables are viewed to be crucial for ecological resilience, as they control the system’s position in a stability landscape. For example, rangelands are viewed to be controlled by slow variables, such as rainfall, fire, grazing and woody plants (trees and shrubs), and fast(er) variables, such as annual grasses and perennial grasses. Dependent on the slow variables, rangelands may exhibit two fundamentally different regimes, namely a locally stable grassy regime and a locally stable woody regime (Walker 2002;

Walker & Abel 2002).[165] The important point here is that most resilience scholars employ a top-down approach to ecological modelling.

As we have considered above, the top-down approach, however, faces a severe problem. State variable models are dependent on *educated guesses* about which mechanistic details are important for describing dynamics and which details can be safely abstracted. To put it bluntly: the key variables are not self-evident (Volker Grimm, *personal communication*). It is but an educated guess to decide what variables constitute the key variables, which are to be included in the model, and which variables exist at the whim of these key variables. This is a clear shortcoming within resilience research with respect to model formulation. To meet this shortcoming attempts should be made to integrate top-down models with bottom-up models, as proposed by Grimm (1999).[166]

The threshold approach for the estimation of ecological resilience is thus restricted to ecological situations in which both assumptions hold: ecosystems must exhibit alternative stable states and it must be possible to identify the key controlling variables. When these necessary preconditions are met, the threshold approach tracks ecological resilience by means of a bifurcation diagram. As illustrated in *Figure 16*, this diagram plots the equilibria of an ecosystem on axes of a fast variable and a slow variable. In the case of a shallow lake, for instance, the fast variable is represented by the abundance of macrophytes while the slow controlling variable corresponds to the phosphate concentrations in the sediment of the lake. The plot then shows upper and lower sets of stable basins of attraction, i.e. regions in state space where the system tends to remain, separated by two ecological thresholds and an unstable set of equilibria. The important point here is that the value of the slow variable is regarded to control an ecosystem's position in state space, and thus, to be responsible for the maintenance of ecological resilience of the whole system. Therefore, the amount of the ecosystem's ecological resilience is estimated as the distance from the current value of the slow variable (CV_{sv}) to the value of the ecological threshold (ET_1) (Carpenter *et al.* 2001; Peterson *et al.* 2003b). In other

165 Note that the identification of the key variables is highly related to the research question of concern (Jax 2006). There may be several sets of key variables for different research questions. Clearly, the focus in resilience discourse is on the provision of ecosystem services.

166 This critical inquiries to the top-down approach, which is used by resilience scholars, is only provisional. A sound and extensive critique would certainly be of high interest and importance for progress in resilience research.

words: in this approach, the slow variable performs as a resilience surrogate, as it is this variable that controls the position of the whole ecosystem within state space.

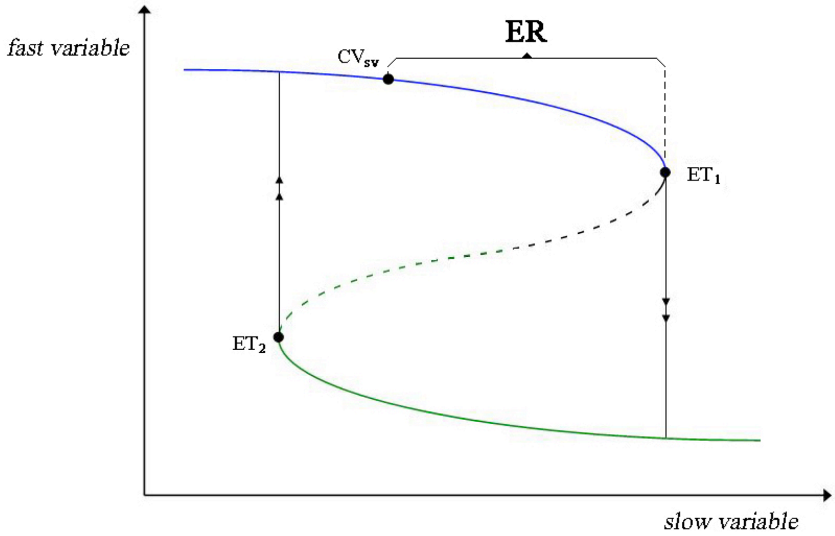

Figure 16: Bifurcation diagram with a fast and a slow variable
from Brand and Jax 2007

Thus, the estimation of ecological resilience means to examine both the current value and the threshold value of the slow variable. Principally, the former is easy to measure, as for instance the phosphate concentrations in lake sediment. What appears to be more difficult is to predict the location of the ecological threshold point (or zone). This further difficulty of the threshold approach is hardly acknowledged but evident: in order to estimate the ecological resilience by means of the 'slow variable surrogate' it is necessary to *predict* the position of the ecological threshold as regards to the slow variable (maybe besides across-site comparisons). This is a difficulty because predictions in ecology are hard to achieve. Yet there are at least three (perhaps interrelated) methods for predicting the position of ecological thresholds. Those are: (1) the extrapolation of empirically estimated return times of controlling variables well distant from the threshold (Wissel 1984); (2) the examination of standard deviations of fast variables in the vicinity of thresholds (Carpenter & Brock 2006); and (3) the repeated calculation of the Fisher Information of a specific ecosystem (Mayer *et al.* 2006).[167] These sound methods have not been of much impact in the relevant literature but it would be interesting to examine their characteristics and interrelations more profoundly.

3.4.2.3.1.3 Archetypal Models

A more sophisticated method to estimate resilience surrogates that is also a threshold approach, i.e. it is related to the concept of key variables and ecological thresholds, has been suggested by Bennett *et al.* (2005a). I consider this publication a milestone paper for the operationalization of ecological resilience and that is why I will consider it here in full detail.

Bennett *et al.* (2005a) suggest a step-by-step approach to operationalize and estimate ecological resilience.[168] The main steps are: (1) assessment and problem definition, (2) identifying feedback processes, (3) designing a systems model and (4) determining resilience surrogates.

Step one means to determine the focal or *key variables* as well as the conditions that are of interest and removing from consideration those that are not. It refers to the two questions: ‘what aspects of the system should be resilient?’ and ‘what kind(s) of change would we like the system to be resilient to?’. According to Bennett *et al.* (2005a) the problem definition means to identifying the system of interest, the desired state of the system, the external disturbances and drivers that are of interest and potential impediments or aids to maintaining the system in that state. Clearly, this corresponds to the *of-what part and to-what part of resilience analysis* considered in detail in the previous sections 3.4.2.1 and 3.4.2.2.

Step two means to identify feedback processes that maintain or change a condition. Bennett *et al.* (2005a) define feedback processes as the processes that determine the nature of the interactions among key variables and distinguish between positive and negative feedbacks. Step two refers to the questions: ‘what variables are changing?’, ‘what processes and drivers are producing these changes?’, ‘what forces control the processes that are generating change?’. The answers to the questions will

[167] For a detailed investigation of the different methods to predict dynamic regime thresholds confer section 3.2.3.3.4.2.

[168] In fact, Bennett *et al.* (2005a) propose this approach for whole social-ecological systems and, hence, for social-ecological resilience. In my view, however, in their publication the boundary between a descriptive concept of ecological resilience and a more vague boundary object of resilience (*sensu* social-ecological resilience) (Brand & Jax 2007) gets more and more diluted.

define the variables of the system that should be examined, the processes internal and external to the system that are producing important changes, and the connections among these processes. According to Bennett *et al.* (2005a) answering these questions should result in a rough understanding of the key processes that define a system and the likely locations of feedback loops, i.e. the output of a process influences the input of the same process. Clearly, the approach suggested by Bennett *et al.* corresponds to a top-down or state variable approach to the modelling of ecosystems. All their examples used make educated guesses about the key variables of interest. As we have considered in section 3.4.2.3.1.1, this is a clear shortcoming of this approach.

Step three means to design a system model that includes all the key elements of the system and the feedback processes or linkages among the elements. It refers to the questions: 'what are the key elements and how are they connected?', 'what positive and negative feedback loops exist in the model and which variables do they connect?', 'what, if any, are the intervening factors that influence or control these feedback loops?' and 'what (if anything) moves the system from being controlled by one feedback loop to another?'.

Note that the methodology of self-identity claims to specify the boundaries of the ecological unit, the selected elements, the expected degree of internal relationships as well as the component resolution (Jax *et al.* 1998; Jax 2006). Therefore, in my view step 1 – 3 in Bennett *et al.*'s approach can also be interpreted as the *finding of the self-identity of the system of interest*.

Finally, step four means to use the systems model to identify resilience surrogates. According to Bennett *et al.* (2005a), resilience surrogates relate either to *the distance of the system to a threshold*, which has been termed *precariouness* by Walker *et al.* (2004) or to *the movement of the threshold itself*.[169] The former refers to (*a*) the distance of the state variable from the threshold, (*b*) the rate at which the state variable is moving toward or away from that threshold, and (*c*) the outside controls or shocks that may change the direction or rate of change of this state variable, while the latter refers to (*a*) the effects of a change in slow variables on the position of the threshold, and (*b*) the factors that bring these changes in slow variables.

In sum, step four corresponds to answering the following questions: 'as indicated by the feedback loops, what is the threshold value of the state variable?', 'how far is the

[169] Note that this view is related to the distinction made by Beisner *et al.* (2003) between a *community perspective* and *ecosystem perspective* concerning alternative stable states or regimes (cf. section 3.2.3.2).

state variable from the threshold value?', 'how fast is the state variable moving toward or away from the threshold?, 'how do outside shocks and controls affect the state variable and how likely are those shocks and controls?', 'how are the slow variables changing in ways that affect the threshold location?' and 'what factors control the changing of these slow variables?'.

To extract resilience surrogates Bennett *et al.* (2005a) subsequently generate *archetypal system models*. They discuss four types of archetypal models, namely 'limits to growth', 'limits to growth with a threshold', 'tipping point' and 'shifting tipping point'. To illustrate their methodology I will describe the shifting tipping point archetypal model in the following in more detail. As illustrated in *Figure 17*, in the 'shifting tipping point' model, there are two potential thresholds and the one in effect at any given time depends on the condition of the system. A third process can control the relative balance between the two competing limits, causing the tipping point between them to shift over time. Although ecological resilience can be quantified by measuring how far the system is from the threshold that will cause it to enter an alternative stable regime, in this system the threshold can move and disappear. Therefore, the additional resilience surrogates that exist for this archetypal model are the rate and direction in which the threshold is moving.

Consider the savannah ecosystem shown in *Figure 17*. Grass (grassland), trees (woodland), fire and elephants have been identified as the key variables. As has been shown empirically, at densities of over approximately 0.5 km^2, the activities of elephants and the linked interactions between woodlands and fire can convert savannah woodlands to shrub lands or grasslands. In recent decades, growth in elephant populations in Southern Africa, together with habitat loss, has lead to increasing conversion of woodlands in many protected areas. As proposed by Bennett *et al.* (2005a), in this example,an important consideration for developing resilience surrogates for elephant management lies in the different speeds at which the variables fire, grass, elephants and trees change (cf. section 3.2.2 for the concept of key variables). The sustained pressure exerted by elephants on the slower variable (tree cover) can allow the faster variables (grasses and fire) to capture the system.

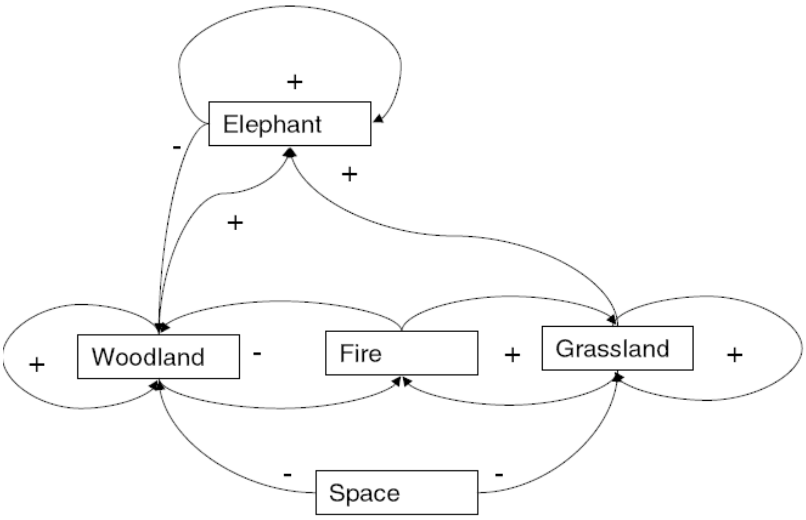

Figure 17: The shifting tipping point model. Growth of two variables is each inhibited by one another. In this case woodland and grassland both exist in positive feedback loops, however each limits the growth of the other. Fire and elephant woodland destruction mediate this relationship. Elephant numbers depend upon availability of woodland allowing the system to exist in different states that can be long-lasting
from (Bennett *et al.* 2005a: 13)

The important point is that ecological resilience in the 'shifting tipping point' example can be quantified by measuring *how far the system is from the threshold that will cause it to enter an alternative stable regime* and *the rate and direction in which the threshold is moving*. Possible resilience surrogates include quantifying how far the elephant density is from the threshold at which elephants will start to degrade their own habitat, the relative proportions of mature trees, shrubs and grasses in the system and the relationship of both elephants and vegetation to the number and extent of fires (Bennett *et al.* 2005a). Note that this approach is obviously much more detailed and comprehensive than the original bifurcation approach considered above. Apparently, the Bennett *et al.* (2005a) approach is only useful for ecosystems that exhibit alternative stable regimes and ecological thresholds. But as we have considered above, many ecosystem can exist in alternative stable regimes but not all (Schröder *et al.* 2005). How is ecological resilience operationalized in cases where ecosystem do not exhibit alternative stable regimes and ecological thresholds? To answer this question, in the following I will turn to another approach that is based on the methodology of self-identity.

3.4.2.3.2 The Self-Identity Approach

The second approach to operationalize ecological resilience refers to the concepts of *identity* and *self-identity*, respectively. I have explored the fundamentals of these concepts in section 3.4.2.1.2. In short, the self-identity of an ecological unit, e.g. an ecosystem, consists of the specification of (*a*) the boundaries, (*b*) the set of selected elements, (*c*) the required degree of internal relationships and (*d*) the component resolution (Jax 2006). According to this approach a system is considered resilient, if it is able to maintain its self-identity over the time horizon of interest in the face of internal change and external disturbances (Cumming *et al.* 2005).[170] I will term this approach the *self-identity approach* in the following.

An obvious advantage of the self-identity approach is that it is applicable to literally *all* kinds of ecosystems. In contrast to the threshold approach concerned above, it is not obliged to relate to the existence of ecological thresholds, i.e. it does not have to assume that ecosystems exhibit alternative stable regimes. Furthermore, important and undesirable changes in ecosystems may occur *before* the system shifts to a potential alternative stable regime. The important point here is that the self-identity approach possesses universal applicability.

How does this method proceed? The self-identity approach is comprised of essentially three steps. Those are: (1) specification of the desirable self-identity (*setting the baseline condition*), (2) specification of the amount of change that would result in the loss of self-identity (*identifying the bounds of desirability and bounds of undesirability*), and (3) estimation of the distance between the current state/ regime of the system to the bounds of undesirability (*estimating the amount of ecological resilience*).

The first step means to specify the self-identity of the system that is desirable on a specific temporal and spatial scale. This step of the self-identity approach corresponds to the second step of the *of-what part*, which is described in detail in section 3.4.2.1.2. The methodology of self-identity and the SIC-model proposed by Jax *et al.* (1998) and Jax (2006) are critical tools here. It is also very much dependent on step one of the of-what part, that is, on societal decisions on ‘what nature we want to and ought to preserve’. Hereby, this step sets a fixed *baseline condition* against which scientists can quantify changes in the system.

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[170] For this operational definition of ecological resilience confer section 4.1.2.

The second step of the self-identity approach means to specify the states or regimes that are undesirable from a societal perspective. It is essential to specify quantitatively the amount of change in the self-identity that would result in the loss of the desirable self-identity and constitute a new and undesirable state/ regime. For example, the self-identity of an old-growth forest may be lost, if desirable ecosystem services, such as the water quality downstream or the value of the land for recreation, are not longer being provided due to extensive timber extraction, for instance. In addition, as regards to changes in system self-identity, one may distinguish between *qualitative changes*, e.g. species extinctions or invasions, and *quantitative changes*, e.g. a certain amount of habitat or abundance of species (Cumming *et al.* 2005). Thus, the second step of the self-identity approach specifies the *bounds of desirability* and *bounds of undesirability* that are of concern in a certain resilience analysis.

At first sight, decisions about the bounds of desirability and bounds of undesirability for ecosystem states appear rather arbitrary, as they mostly are not related to a rather clearly delimited 'natural' threshold. However, the apparent arbitrariness points to a very important tenet: every decision about bounds of desirability or bounds of undesirability is a value judgement. It points to the fact that it is, ideally, a societal choice what we conceive as desirable or undesirable in nature (Honnefelder 1993; Jax & Rozzi 2004). Thus, the decision about bounds of desirability and undesirability refers to the *first step of the of-what part* considered in section 3.4.2.1.1.

The self-identity can be quantitatively defined in relation to boundaries within a state space of the (key) variables of interest (the selected elements of the self-identity). Dependent on the amount of variables considered the self-identity may thus be a *multivariate entity* (Cumming *et al.* 2005). For example, in their resilience analysis Cumming *et al*. (2005) identify the bound of desirability for a rain forest in the southwestern Amazon with respect to road development.[171] The bounds of desirability extends to 25% forest cleaning, a decline of approximately 30% in hardwood seedling recruitment and a 50% decline in rubber production. That is, up to this boundaries the rain forest would keep its desirable self-identity despite changes in key variables. In contrast, beyond the boundaries in state space the rain forest

[171] Cumming *et al*. (2005) do in fact consider the bounds of desirability in state space for whole social-ecological systems and refer hereby to boundaries in economic as well as social variables. As I am only concerned here with a descriptive ecological concept of ecological resilience (Brand & Jax 2007) I only treat the ecological part of their study.

would be conceived as undesirable, owing to the loss of essential ecosystem services. Decisions about the location of the boundaries in this study are of course in a sense arbitrary yet are made explicit and could be made the object of further discussions about desirable states of the rain forest in this area. The important point is to make explicit the bounds of desirability and bounds of undesirability with respect to the variables considered in a certain study.

To sum up: the first and the second step specify what ecosystem states are conceived as desirable and what states are regarded as undesirable in a particular study. The third step of the self-identity approach then tries to estimate the ecological resilience of the system of concern. Ecological resilience is estimated as the distance between the current ecosystem state and the bounds of undesirability with respect to the variables considered. In the example of the rainforest, ecological resilience would be estimated as the distance of the current forest cleaning, hardwood seedling recruitment and rubber production to the bound of undesirability, i.e. 25% forest cleaning, 30% hardwood seedling recruitment and 50% decline in rubber production.

Note that again it is important to relate this estimation of ecological resilience to a specific disturbance regime because different disturbance have of course different impacts on the ecosystem of concern. For instance, there are fundamental differences in the impacts of each the extensive use of timber, the use of fruits and seedlings or invasive species.

How to evaluate the self-identity approach? On the one side, it is appealing for its universal applicability, but on the other side appears to be kind of abstract. Ecological resilience is pictured as a measure of aggregated distances in state space, and therefore does not refer to a genuine capacity or ability of an ecosystem. Yet a concept of ecological resilience claims to refer to the ability of an ecosystem to maintain a certain ecosystem state despite the occurrence of specific disturbances. Hence, the SI-approach may miss the point. Certainly, the methodology of identity or self-identity is a sound approach to specify ecological units but may be inappropriate to reflect a capacity or ability of an ecosystem (K. Jax, *personal communication*). In my view however, the SI-approach gains a lot by relating it to disturbance scenarios (as in Cumming *et al.* 2005). Ecological resilience is then estimated as the capacity of an ecosystem to maintain its identity or self-identity in the face of various disturbance regimes. This fruitful extension is examined in some detail in section 3.4.2.4.

3.4.2.3.3 The Resilience Mechanisms Approach

The third approach to estimate ecological resilience refers to the concept of *resilience mechanisms*. I will therefore dub this approach *resilience mechanisms approach* in the following.

The term "resilience mechanisms" points to distinct properties and mechanisms that have causal or nomic force (Cooper 1998) in creating ecological resilience. It is assumed that ecological resilience represents an emergent property that can be functionally reduced to the interplay of several properties and mechanisms. In section 3.3.3.3 I have identified twelve fundamental resilience properties and mechanisms, respectively, on (*a*) the local scale and on (*b*) the regioanl scale.

Resilience properties at the local patch scale are: (1) *population diversity*, which includes population parameters, such as size, as well as genetic diversity within and among populations; (2) *the maintenance of functional important types of species and their specific life-history traits*, such as real and potential drivers, keystone process species, keystone species, ecological engineers, umbrella species or dominant species; (3) *the amount of ecological redundancy*, i.e. species that perform the same ecosystem process, *and response diversity*, i.e. ecologically redundant species showing different responses to disturbances, *within functional effect groups*; and (4) *the maintenance of critical functional groups*, i.e. those functional effect groups that have low ecological redundancy and whose lost results in a dynamic regime shift. The critical resilience mechanisms is (5) *local functional compensation*, i.e. the tendency of coexisting, competing species within a patch to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition.

The resilience mechanisms/ properties at a regional scale are: (6) *the concentrations of biological legacies within the disturbed area*, i.e. organisms and organic structures that survive a disturbance event, serve as foci for regeneration and allow species to colonize; (7) *the existence of source habitats in the vicinity of the disturbed patch*; (8) *appropriate levels of connectivity within the meta-community, i.e. (a) small to moderate distances and (b) intermediate (but not high) dispersal rates between habitat patches*; and (9) *an intermediate degree of regional heterogeneity*. The critical resilience mechanisms is (10) *regional functional compensation*, i.e. the tendency of coexisting, competing species at a landscape scale to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition.

Further resilience mechanisms are: (11) *the occurrence of intermediate, pulse disturbances at a small spatial and temporal scale*; and (12) *the value of controlling abiotic variables*.

The message of this list is that there is no single cause for ecological resilience. Rather it is the interplay between the variety of mechanisms occurring at different levels of organization that is responsible for the emergence of ecological resilience at the ecosystem level. Resilience mechanisms may thus be regarded as (*biophysical*) *components* of ecological resilience (Walker *et al.* 2002). It must be noted that in particular ecosystems specific resilience properties or mechanisms will be existent or absent and different properties or mechanisms will be important to a particular degree.

How to quantify resilience mechanisms in order to gain a measure of ecological resilience? Considered in terms of quantification, the 12 resilience properties and mechanisms behave differently. Some of them (which I call *class 1*), i.e. (2) the maintenance of functional important types of species and (4) the maintenance of critical functional groups can only be examined qualitatively as existent or absent, even though parts of them, i.e. the specific life-history traits can be quantified. Others (*class 2*), i.e. (1) population diversity, (3) the amount of ecological redundancy, (7) the extent of source habitats in the vicinity of the disturbed patch, (8) appropriate levels of connectivity within the meta-community, i.e. (a) small to moderate distances and (b) intermediate (but not high) dispersal rates between habitat patches, (9) an intermediate degree of regional heterogeneity, (11) the occurrence of intermediate, pulse disturbances at a small spatial and temporal scale and (12) the value of controlling abiotic variables, can principally be examined quantitatively. Still others (*class 3*), i.e. (6) the concentrations of biological legacies within the disturbed area can only be observed after the disturbance has already occurred. The critical mechanisms (5) local functional compensation and (10) regional functional compensation are more or less the result of the interplay of the other resilience properties.

The important point here is that the interplay of the resilience-conducive properties (RCPs) results in a specific potential for functional compensation, which represents - along with some alternative stability mechanisms - the most relevant resilience mechanism. Functional compensation is understood in this thesis as the tendency of coexisting, competing species to increase in abundance or replace species should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition (Ernest & Brown 2001b; Vinebrooke *et al.* 2004). Such a comprehensive concept of functional compensation includes several components. These are: asynchronous population fluctuations, species replacement, phenotypic variance within functional groups and evolutionary adaptation. The important point here is that *the potential for functional compensation is taken as a surrogate for ecological resilience*. It is thus the potential for functional compensation that makes ecosystems capable to show ecological resilience to various disturbances. Following the resilience mechanisms- (RM-) approach, the potential for functional compensation, and hence the amount of ecological resilience is a function of the various RCPs:

$$\text{ER} = \text{potential for FC} = f_{\text{DT, ET}}\,(\text{FIS, ERed, RD, BL, SH, C, RH, SCD, AV})$$

where AV: Abiotic variables, BL: Biological Legacies, C: Connectivity, DT: Disturbance Type, ER: Ecological Resilience, ET: Ecosystem Type, ERed: Ecological redundancy, FC: Functional compensation, FIS: Functional important species, SCD: Small-scale disturbances, RD: Response Diversity and RH: Regional heterogeneity. Clearly, the amount of ecological resilience is dependent on the specific ecosystem type and the disturbance regime of concern.

Note that the list of RCPs is limited for at least the following reasons. First, at the regional scale the properties are not related directly to ecological resilience but to functional diversity and species richness, which are taken as substitutes or surrogates for ecological resilience. Second, the large-scale properties are proposed from a source-sink perspective only and could be enlightened by other paradigms for meta-communities, e.g. the species-sorting paradigm. And third, such a list is in a sense arbitrary and incomplete, as it is possible to find other relevant RCPs. The usefulness of the list ought to be tested with the help of long-term empirical examples.

Based on this certainly limited list, the RM-approach for the estimation of ecological resilience may proceed as follows. Step 1 carries out an assessment of the RCPs that are present in the specific study area. Subsequently, step 2 estimates the degree of the gradual RCPs, e.g. the degree of response diversity or connectivity. Finally, step 3 estimates the amount of ecological resilience by weighing and assessing the RCPs with reference to the specific disturbance regime and ecosystem type. As a result, a proxie of ecological resilience may emerge.

The strength of the RM-approach is that it explicitly points to ecological resilience as a genuine capacity or ability of an ecosystem in referring to the various properties and mechanisms underlying this capacity. Yet it may be highly problematic to assess the exact amount of resilience mechanisms in relation to the disturbance regime and the ecosystem type.[172] The RM-approach is surely at an immature theoretical stage but in my view represents a fruitful avenue for further research.

3.4.2.3.4 Alternative Approaches

Apart from the three fundamental approaches described in the previous sections, some authors have suggested alternative approaches for the empirical estimation of ecological resilience. To my knowledge, three other approaches have emerged in the relevant literature, namely the 'probabilistic resilience'-method proposed by Peterson (2002b), the 'cost of restoration'-method suggested by Martin (2004) and quite recently the 'digital remote sensing'-method championed by Washington-Allen *et al.* (2008). These approaches have not been of much impact in the literature on resilience and have not yet been developed further since their proposition.

Probabilistic resilience

Peterson (2002b) argues that the threshold approach can be applied only to relatively simple models. However, ecological managers often do not have simple models of their systems and, therefore methods of estimating ecological resilience that do not depend on the construction of system models would greatly facilitate the applicability of resilience-based management. Thus, Peterson (2002b) suggests a method for estimating resilience based on landscape simulation models, termed "probabilistic resilience".

Ecological resilience is analyzed in models that are described in terms of transitions among discrete states. This approach describes alternative states that are distributed

[172] Of course, the resilience mechanisms-approach is based on the specification of the of-what-part and the to-what part of the resilience analysis (cf. section 3.4.2.1 and section 3.4.2.2).

in landscapes, e.g. forest and savannah, and identifies zones of ecotones. In terms of ecological resilience, ecotones can be thought of as the edge that separates regions that are dominated by two alternative stable states. From this perspective, ecotones are areas of low resilience, where small changes can cause a site to shift from one state to another. Percolation theory is used to define sites that exist on the edge that separates to alternative states.

The behaviour of a discrete state can be assessed in terms of the probabilities of leaving that state and remaining in that state. The probability that a state will persist is a measure of its resilience. If the probability that a state will persist is less than the probability that it will not, then it is vulnerable to change. By mapping these probabilities across space, the areas of vulnerability and resilience in a landscape can be estimated. Based on Monte Carlo simulation used to estimate those probabilities, Peterson (2002b) applies cross-scale edge for examining the transition probabilities of ecotones and alternative states in order to get a simple measure of landscape resilience.

Costs of restoration

An alternative approach is suggested by Martin (2004). A mathematical formulation of resilience is proposed in the framework of the viability theory. The author focuses on the set of trajectories starting from any state of the system and associated with different control functions. At each trajectory, a cost of restoration is being associated if the property is lost and then restored. This cost is infinite if the property cannot be restored. Martin (2004) defines the resilience value at one state of the system as the inverse of the minimal cost over all trajectories starting at the state to which the system has jumped following the disturbance. The cost of restoration is then suggested as a way of estimating ecological resilience.

Vegetation indeces

Quite recently, Washington *et al.* (2008) have proposed an approach to measure ecological resilience in drylands. They use vegetation indices, i.e. proxies of vegetation characteristics such as phytomass, that are derived from spectral properties of Landsat imagery. These indices are then feeded in a mean-variance analysis. Note that Washington *et al.* (2008) employ a slightly different notion of ecological resilience, as they focus on alternative concepts, such as amplitude, i.e.

the magnitude of response of a vegetation indice to a disturbance, and malleability, i.e. the degree of recovery of a resource after a disturbance.

All these approaches are interesting but have not yet been of large impact in the relevant literature. They could be used as complementary to the other rather established approaches.

3.4.2.3.5 Concluding Remarks

As we have considered in the previous section, the estimation of proxies or surrogates of ecological resilience is possible by following several paths. It is important to acknowledge that each of these proxies may refer to different aspects of ecological resilience. As we will consider in some detail in section 4.1, the concept of ecological resilience is interpreted in several ways and numerous meanings have been ascribed to the term. According to Brand (2005: 111ff), the concept of ecological resilience can be understood as a melange of different stability concepts, such as resistance, elasticity or constancy, which are applied to different scales (e.g. local, regional).
Thus, it come as no surprise to notice a diverging focus of different resilience surrogates. While the threshold approach (cf. section 3.4.2.3.1) refers to *resistance* of the slow variables to disturbance, the self-identity approach (cf. section 3.4.2.3.2) is related rather to *constancy* of a specified system state and the resilience mechanisms approach (cf. section 3.4.2.3.3) also to the learning and innovation-aspect of ecological resilience (e.g. small-scale disturbances), as conceived by Folke (2006). It is questionable whether it is advantageous or disadvantageous that the concept of ecological resilience is comprised of several aspects. The important point here is that the different methods to estimate resilience proxies ought to be used complementary for achieving a “robust” measure of ecological resilience.

3.4.2.4 Options for Generalizations: Disturbance Scenarios

There is an interesting tension within resilience theory. This tension exists between a concept of *specified* (or *targeted*) *ecological resilience*, i.e. the resilience of a specified system to a specified disturbance regime, and a notion of a more *general ecological resilience*, i.e. the general capacity of a system that allow it to absorb unforeseen disturbances (Walker & Salt 2006: 120f).

Some authors employ ecological resilience in a specified sense. It is assumed that a general measure for ecological resilience is an illusion and cannot be achieved. As Walker and Abel point out:

> "[t]he notion of a general resilience – that is ecosystems that are resilient in the face of any and all disturbances for all purposes (production, species diversity, aesthetic value, and so on) – is not achievable and the quest for it clouds understanding" (Walker & Abel 2002: 295).

This view corresponds to the approaches that we have considered in the previous sections. If we stopped the resilience analysis with the previous step of the indication of ecological resilience we would arrive at a measure of ecological resilience to a *single* specified disturbance regime (i.e. the concept of ecological resilience *to* what). The approach proposed by Bennett *et al.* (2005a), for instance, represents a good example for a concept of specified ecological resilience.

In contrast, other authors (sometimes even the same authors) use a notion of general ecological resilience. It is assumed that systems might be resilient to several and unforeseen disturbance regimes. For instance, Holling argued that a management approach based on ecosystem resilience assists to build systems that "can absorb and accommodate future events in whatever unexpected form they may take" (Holling 1973: 21). Likewise Walker *et al.* propose to focus on "maintaining the capacity of the system to cope with whatever the future brings, without the system changing in undesirable ways" (Walker *et al.* 2002: 2).

The tension between a specified concept and a general notion of ecological resilience is typical for stability concepts.[173] Indeed, stability statements are prone to unjustified generalisation. Most ecological studies only allow statements about a particular system and even these statements are of extremely limited validity. Yet scientists are no less subject to their desires, hopes and dreams in their work than anyone else. A strong temptation for a more mature ecology prevails but is at the

[173] This may be due to the long-standing controversy in ecology between holistic and individualistic system notions, as considered in section 2.4.

same time a far off horizon. Therefore, stability statements are frequently generalized without sufficient justification (Grimm & Wissel 1997).

Is there a rather fruitful approach for a more general measure of ecological resilience? As far as I view the relevant literature, the answer is yes.

Generalizations for statements about ecological resilience may be achieved by the use of *scenarios*. As opposed to terms such as prediction or forecast, *scenarios* are defined as “a set of plausible narratives that depict alternative pathways to the future” (Bohensky *et al.* 2006: 1052).[174] Building on this definition, scenario planning is understood as the creation and use of such scenarios in a structured way to stimulate thinking and evaluate assumptions about future events or trends and to make uncertainties about these explicit (Bohensky *et al.* 2006). Scenario planning uses a set of scenarios, each of which is a plausible example of what could happen under particular assumptions and conditions. Put simply: each scenario is a description of what the future could be, not a prediction of what the future will be (Peterson *et al.* 2003c). Scenarios are distinguished from other approaches to *future assessment*, such as forecasting and risk assessment, by being specifically intended for situations in which the factors shaping the future are highly uncertain and largely uncontrollable (Biggs *et al.* 2007). Generally, scenarios are intended to widen perspectives and illuminate key issues that might otherwise be missed or dismissed (Kok *et al.* 2007).

Scenario approaches have been widely used in the relevant literature. Prominent examples include scenarios on global change (Sala et al. 2000), vulnerability in Europe (Schröter *et al.* 2005), the world’s ecosystem services, i.e. the Millenium Ecosystem Assessment (UNEP 2005a), climate change, i.e. the work of the Intergovernmental Panel on Climate Change (e.g. IPCC 2007), positive visions and development alternatives for global land-use (Carpenter & Folke 2006) and the Global Environmental Outlook Scenarios. There are also rather local scenarios approaches as regards to conservation planning (Peterson *et al.* 2003c), for instance.[175]

In recent years, more attention has been paid to the value of incorporating participatory methods in scenario analyses. These *participatory scenarios* include a

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[174] It is important to make a distinction between scenarios in this sense and projections, forecasts and predictions, all of which relate more to the probability than possibility of future outcomes (Peterson *et al.* 2003c). Projections and forecasts work best for short-term forecasting in well-understood systems. Scenario planning on the other hand is most useful for dealing with uncertainty when we lack sufficient information about the probabilities that different events will occur, as in most environmental management situations in which uncertainty and surprise are common (Bohensky *et al.* 2006).

[175] For the history of scenario approaches and further literature confer Biggs *et al.* (2007) and Kok *et al.* (2007).

variety of stakeholders and are meant to be easily understood by scientists, policy makers and lay people (Wollenberg *et al.* 2000; UNEP 2005a; Kok *et al.* 2007). The choice of where to focus participatory exercises is primarily determined by the particular objective of the scenario. When the aim is to engage local-level stakeholders in processes initiated at higher levels and to increase the relevance of such initiatives, as in the MEA scenarios, the focus will tend to be on engaging local-level stakeholders. Where the aim is primarily to influence national-level decision making, the focus will be on engaging stakeholders at this level. In some cases it will be desirable to engage stakeholders to a similar extent at several scales (Kok *et al.* 2007).

Scenarios range widely in their goals, content and development methods. Biggs *et al.* (2007) offer numerous *dimensions* along which the scenarios in the relevant literature vary and can be located. These include, among others, *purpose* – exploratory or decision support, *motivation* – scientific inquiry or policy support, *approach* – quantitative (e.g. formal models, statistical forecasting, trend-impact analysis) or qualitative (e.g. visioning, intuitive logic, storytelling), *number of focal scale* – single scale or multiple scales, and *links between scales* – loosely linked (perspectives, uncertainties and drivers from each scale partially inform scenario exercises at other scales) or tightly coupled (perspectives, uncertainties and drivers from each scale strongly inform the scenario exercises at other scales). Broadly speaking, scenarios may be designed either primarily for *exploratory purposes and scientific inquiry*, thus, to understand alternative development trajectories and the impacts and interactions of the key forces driving change (as for example the scenarios used for the Millenium Ecosystem Assessment) or for *decision support*, i.e. scenarios that are more narrowly focussed on developing or testing specific policies.

What is the relevance inherent in the scenario approach to estimate ecological resilience? With respect to ecological resilience, scenarios can be used to explicate several storylines of possible disturbance regimes that ecosystems may face. Thus, in the following this section proposes an approach to examine the general ecological resilience for *specified* systems but against *several* disturbance regimes. In particular, it will be suggested that the ecological resilience is high if the ecosystem remains in the desirable states (or regime) across several likely disturbance regimes. This corresponds to the concept of “general resilience” *sensu* Walker and Salt (2006).

Peterson *et al.* (2003c) suggest a five-step approach to scenario building for conservation issues and their approach can be used likewise for resilience statements. Step one (termed *identification of a focal issue*) and step two (dubbed *assessment*) of scenario building mean to identify the focal issue, i.e. to clarify the system of interest and the research question, and to specify the variables, disturbances and changes that are of interest in the particular study. This corresponds largely to the of-what part of the resilience analysis considered in section 3.4.2.1. In short, it is necessary to specify the ecosystem services that are of concern, the self-identity of the system considered and the specific disturbance regimes that are of interest.

Subsequently, step three (termed *identification of alternatives*) means to identify alternative ways that the system could evolve. These alternatives should be both plausible and relevant to the original research question. In the case of resilience analysis, both the key variables, i.e. the slow and fast variables in the threshold approach, the selected elements and interactions in the self-identity approach and the resilience-conducive properties in the resilience mechanisms approach, and the impacts of a range of potential future disturbances ought to be considered. Obviously, some alternatives are more plausible or likely than others. It is thus important to weigh the alternatives according to their likelihood of occurrence based on assumptions about the occurrence of future disturbances, such as land-use measures or patterns of global change. Note that the alternatives are always prone to some degree of uncertainty (Cumming *et al.* 2005). Importantly, the third step of the scenario approach results in a *set of plausible alternatives*.

Step four, dubbed *building scenarios*, builds a set of scenarios based on the understanding accumulated during the previous steps. Scenarios convert the key alternatives of step three into dynamic stories by adding a credible series of external forces and system's responses. Thus, scenarios should become brief narratives that link historical and present events with hypothetical future events. Within these storylines the internal assumptions of the scenario and the differences between stories must be clearly visible (Peterson *et al.* 2003a). To understand the effects of external drivers and system's responses scenario planers may use integrated dynamic systems models or interactive approaches among stakeholders (cf. for further literature Bohensky *et al.* 2006). Subsequently, the scenarios should be tested

for consistency by means of simulation models or theoretical plausibility, which is step five of the scenario approach, termed *testing scenarios*.
What type of scenarios ought to be used with respect to resilience analysis? In the following I will treat this question by means of the dimensions unfolded in Biggs *et al.* (2007) concerned above. The scenarios with respect to a resilience analysis clearly should be descriptive and exploratory aiming primarily for scientific inquiry. The scenarios should range between the two extremes "on process" and "on outcome" because both the development of storylines and the implications of storylines for decision making are of interest. Scenarios with respect to ecological resilience should be based on empirical knowledge and may use the full range of approaches proposed in the relevant literature, from "hard" system models to soft methods, such as visioning and storytelling, but the tendency should be to incorporate useful system models.[176] Thus, scenarios will be expert-driven and the level of uncertainty ought to be reduced as far as possible.
In particular, it is necessary to decide about the *number of focal scales* of the scenarios used for a resilience analysis. This is an important point because it means to decide what disturbances to be included and what disturbances not to be included in a resilience analysis. As the concept of disturbance includes facets of global change, e.g. climate change or invasive species, regional changes, such as habitat loss and fragmentation, and even local changes, such as the loss of a particular species, it is non-trivial to judge what disturbances should be included within an analysis about the general ecological resilience of a specific ecosystem. This leads to the discussion of multiscale scenarios.
Two features can be used to categorize and understand types of *multiscale scenarios*, i.e. scenario approaches that develop storylines at several scales, e.g. global and national, and are linked to another to some degree. Those are: (1) the number of scales at which scenarios are developed and (2) the connectedness between scales, i.e. the strength of the links between them (Biggs *et al.* 2007). Based on this categorization, Biggs *et al.* (2007) identify three types of *scenario exercises*: (1) single-scale scenario exercises, which are constructed at a single focal scale; (2) loosely linked scenarios constructed at two or more scales; and (3) cross-scale scenarios that are tightly coupled across two or more scales.[177] As regards to a resilience analysis, obviously, more scales make things more complicated but

[176] For an overview of the different approaches confer the references in Biggs *et al.* (2007) and Kok *et al.* (2007).
[177] Examples for each scenario exercise can be found in Table 2 in Biggs *et al.* (2007).

eventually the analysis more realistic. Thus, a resilience analysis should strive for scenarios at several scales, for example, global, continental, regional and local.[178]

But within these multiscale scenarios, scales may be linked in different ways (Biggs *et al.* 2007; Kok *et al.* 2007). Either the process of developing the scenarios can be connected, i.e. having the same team of scenario developers create the scenarios at each scale to running parallel processes in which scenarios are built using the same methods, or the elements and outcomes of the scenarios can be linked (Biggs *et al.* 2007). When linking the elements and outcomes of the scenarios is the chosen method (which may be the most useful option also for a resilience analysis), one may distinguish between *downscaling scenarios* and *upscaling scenarios* (Biggs *et al.* 2007; Kok *et al.* 2007). The former means to "translate" broader-scale scenarios (e.g. the IPCC scenarios) to finer scale situations, and the latter refers to the reverse.

In addition, links between scales can be achieved by means of four methods (Biggs *et al.* 2007). Those are: (1) driver trajectories at the global scale are used as boundary conditions to frame developments within the regional-scale scenarios. The regional scenarios are developed in a way that ensures that the outcomes of the regional scenarios do not conflict with those of the global scenarios. Often, an iterative cycle of downscaling and upscaling is used; (2) the completed global-scale storylines are translated into regional stories; (3) regional scenarios are developed with little or no reference to the global scenarios and then mapped onto the global scenarios; and (4) global scenarios are used to test the viability and effectiveness of regional policy options without developing complete regional scenarios. Concerning a resilience analysis, method 1 might be the best option, as existing broad-scale scenarios (e.g. IPCC, MEA) can be used to inform scenarios developed at a regional scale for specific ecosystems.

To sum up: exploratory, descriptive, expert-driven and multiscale scenarios exercises should be used to gain a more general estimation of ecological resilience for specific ecosystems. The resulting set of scenarios can be used to estimate the generality of statements on ecological resilience. In general, *systems have high general ecological resilience if they remain in the bounds of desirability over a range of several likely disturbance scenarios*. As Cumming *et al.* point out: "if our system is likely to maintain its identity across a broad range of scenarios, it is resilient" (Cumming *et al.* 2005: 13). The specific bounds of desirability correspond to (*a*) the desirable basin of

[178] Taking into account temporal scales would be also important.

attraction in the threshold approach, (*b*) the desirable self-identities in the self-identity approach or to (*c*) a certain amount of resilience mechanisms in the resilience mechanisms approach. That is, the scenario approach can be applied to each approach for the estimation of ecological resilience in order to gain a more general estimate of ecological resilience.

3.5 Environmental Management Approach

Innovative approaches to environmental management are of high relevance for achieving sustainable development. During the history of ecology, a wealth of approaches has emerged referring to e.g. ecosystem integrity (Lemons *et al.* 1997; Rapport 1999), ecosystem health (Rapport 1989; McShane 2004), biodiversity (van der Maarel 1997; UNEP 2005c), ecosystem services (UNEP 2005b; Rodriguez *et al.* 2006), vulnerability (Schröter *et al.* 2005; Adger 2006) or to “classical” nature conservation measures, such as the preservation and management of “wilderness” or specific populations (e.g. Plachter 1991).

This section explores the environmental management approach that is put forward within resilience research, namely adaptive co-management. At first, section 3.5.1 pictures one of the alternative environmental management approaches that is the antithesis to adaptive co-management, namely the conventional resource management approach. Subsequently, section 3.5.2 describes and analyzes the adaptive co-management approach proposed within resilience research. Finally, section 3.5.3 concludes with some remarks on the conceptual and theoretical vagueness of adaptive co-management.

3.5.1 Conventional Resource Management

In the following I describe the conventional resource management approach with reference to an earlier publication (Brand 2005: 142ff). Confer also section 4.2.4.2 and Kirchhoff *et al.* (*in review*) for the specific man-nature relationship of conventional resource management.

According to the resilience approach, much of natural resource management has been an effort to control nature in order to harvest its products, reduce its threats, and establish highly predictable outcomes for the short-term benefit of humanity (Holling & Meffe 1996). The resource manager tries to control a target resource (e.g. supply of fish and timber) by reducing the variability of this target resource (Berkes & Folke 1998).

This corresponds to a focus on the exploitation (r) and conservation (K) phases of the renewal adaptive cycle and an ignorance of the release (α) and reorganization (Ω) phases (Holling & Gunderson 2002; Berkes *et al.* 2003).[179] These so-called *command-and-control approaches* to environmental management imply a reduction in the range of structural and functional variation of natural systems, i.e. variation through time (e.g. small-scale disturbances) and spatial heterogeneity (e.g. ecological redundancy, ecological memory, mobile links, spatial patterning) are reduced. As Holling and Meffe point out:

> "We dampen extremes of ecosystem behavior or change species composition to attain a predictable flow of goods and services or to reduce destructive or undesirable behavior of those systems. For example, we control agricultural pests through herbicides and pesticides; we convert natural, multi-species, variable-aged forests into monoculture, single-aged plantations; we hunt and kill predators to produce a larger, more reliable supply of game species; we suppress fires and pest outbreaks in forests to ensure a steady lumber supply; we clear forests for pasture development and steady cattle production, and so forth" (Holling & Meffe 1996: 329).

Such efforts attempt to replace natural ecological controls, which are largely unknown to us and highly variable, with engineered constructs and manipulations that on the surface seem entirely within our control. The purpose is to turn an unpredictable and

[179] For the description of the adaptive cycle confer section 3.2.1.1.

"inefficient" natural system into one that produces products in a predictable and economically efficient way (Holling & Meffe 1996). And Berkes and Holling warn:

> "The very success of management, effective in the short term, 'freezes' the ecosystem at a certain stage of natural change by actively blocking out environmental variability and feedbacks that govern change. Instead of allowing smaller perturbations to act on the system, management causes the accumulation of perturbations, inviting larger and less predictable feedbacks at a level and scale that threaten the functional performance of the whole ecosystem, and thereby also the flow of resources and services that it generates" (Berkes & Folke 1998: 11f).

According to the resilience approach, the result is a gradual loss of ecological resilience of the ecosystem, as resilience mechanisms such as ecological redundancy or ecological memory are reduced. In this respect, Gunderson (2003) identifies, first, the addition of key substances into the ecosystem (e.g. phosphorus into lakes), second, the removal of key resources or sources of ecological resilience (such as soil in tropical forests, drought-tolerant plant species in rangelands), and, third, the manipulation of keystone ecological processes by human perturbation (e.g. alteration of the fire-regime) as pathways that can lead to the loss of ecological resilience in ecosystems. Gunderson *et al.* (2002) adds, fourth, the homogenizing of temporal and spatial variability.

To use the metaphor of the stability landscape, the basin of attraction shrinks leaving the given regime more vulnerable to disturbance.[180] A disturbance event that previously could have been absorbed by the system becomes the trigger that causes the ecosystem to shift to another regime often with loss of essential ecosystem processes such as productivity (Folke *et al.* 2002).

What is the empirical evidence for this loss of resilience within environmental management? Holling *et al.* (2002a) point to the collapse of some fisheries, the vulnerability to drought of semiarid rangelands as well as to the increased vulnerability through flood control measures and irrigation developments. In addition, Jackson *et al.* (2001) hold that the historical human impact on and recent exploitation of coastal ecosystems have lead to a decrease in ecological resilience. "Early changes increased the sensitivity of coastal marine ecosystems to subsequent

[180] For a description of the concept of "stability landscape" confer section 3.2.3.3.4.1.

disturbance and thus preconditioned the collapse we are witnessing" (Jackson *et al.* 2001: 635).

With reference to such examples, Holling and Meffe propose that such crises and surprises "are the *inevitable* consequences of a command-and-control approach to renewable resource management" (Holling & Meffe 1996: 330) and coin the resulting collapses and crisis the *pathology of natural resource management* (cf. also Holling 1995).

According to Holling (2003), this pathology has the following features: (1) The new policies and development initially succeed in reversing the crisis or in enhancing growth. (2) Implementing agencies initially are responsive to the ecological, economic and social forces, but evolve to become narrow, rigid and myopic. They become captured by economic dependents and the perceived needs for their own survival. (3) Economic sectors affected by the resources grow and become increasingly dependent on subsidies. (4) The relevant ecosystems gradually lose ecological resilience to become fragile and vulnerable and more homogeneous, as diversity and spatial variability are reduced. (5) Crisis and vulnerabilities begin to become more likely and evident and the public begin to loose trust in governance. Crisis, conflict, and gridlock emerge whenever a single target, such as efficiency of production, and piecemeal policy is encouraged, a single scale (typically on the short term and the local) is focused, or there is no realization that all policies are experimental (Holling 1995, 2003).

To sum up, conventional resource management has been successful in producing yields and economic growth in the short term. It has not been very successful in safeguarding the dynamic capacity of ecosystems or in managing ecological and social systems for resilience and sustainability (Folke *et al.* 2002).

3.5.2 Adaptive Co-Management

This section delineates the adaptive co-management approach. At first, section 3.5.2.1 pictures the two components, i.e. adaptive management and collaborative management. Subsequently, section 3.5.2.2 shows that there is a focus within adaptive co-management on coupled social-ecological systems, while section 3.5.2.3 critizes the extension of ecological theory to other types of systems (e.g. social systems). Finally, section 3.5.2.4 reviews the variety of management strategies put forward in the relevant literature on adaptive co-management.

3.5.2.1 The Two Aspects of Managing for Resilience

Resilience theory channels in a specific approach to environmental management[181] termed *adaptive co-management* (e.g. Armitage *et al.* 2008). As the term suggests, this approach is based on the concepts of *adaptive management* introduced by Walters and Hilborn (Walters & Hilborn 1978) and *collaborative management* or *co-management* developed by Berkes *et al.* (1991).

Adaptive management[182] is a widely promoted concept within environmental management. Yet few concepts in environmental management are also as widely misunderstood. As Gregory *et al.* (2006) note, many ecological planning, restoration and recovery initiatives that are promoted under the banner of adaptive management exhibit few of the characteristics generally considered to be essential in the seminal and conceptualizing literature (e.g. Holling 1978; Walters & Holling 1990). Also, according to Plummer and Armitage (2007), a clear need exists to develop a common conceptual and terminological basis for adaptive management. The following sections aim at delineating a comprehensive picture of the essential components and characteristics of adaptive management.

Adaptive management is viewed as a systematic approach to improving the management process and accommodating change by learning from the outcomes of a set of environmental management policies and practices (Holling 1978). Environmental management is viewed to be faced by inevitable change and

[181] The term "management" can be understood as "the right to regulate internal use patterns and tranform the resource by making improvement" (Ostrom and Schlager 1996: 131; quoted in Carlsson & Berkes 2005).

[182] For a good overview of the concept of adaptive management confer Gregory *et al.* (2006); for some examples confer Hughes *et al.* (2007b).

surprises[183] in ecosystem dynamics. Knowledge about the complexity and interconnectedness of ecosystems was always incomplete (Peterson *et al.* 2003b). Hence, an appropriate management approach ought to be adaptive and include a means of learning about ecosystem dynamics (Olsson *et al.* 2004). According to the resilience approach, the generally stated goal of adaptive managament is to improve managers' knowledge about a set of well-defined ecological objectives through the implementation of carefully designed, quasi-experimental management interventions and monitoring programs (Gregory *et al.* 2006).

Two primary types of adaptive management have been defined with respect to the circumstances of a given management problem, "passive" and "active" (Walters & Holling 1990; Gregory *et al.* 2006). In *passive adaptive management*, managers typically use historical data, from the specific area under consideration or from areas considered to be ecologically comparable, to develop a "best guess" hypothesis and to implement a preferred course of action. Outcomes are monitored and new information is used to update the historical data set and, if necessary, the hypotheses and management action.

Under *active adaptive management*, in comparison, managers typically seek to define competing hypotheses about the impact of management activities on ecosystem processes and, in turn, design management experiments to test them. In this way, systems are deliberately tested through management interventions, often with several alternative types of management activities attempted in sequence or in parallel so as to observe and compare results.

The term "collaborative management" or "co-management" has been introduced by Berkes *et al.* and defined as "the sharing of power and responsibility between the government and local resource users" (Berkes *et al.* 1991: 12). Since this publication numerous definitions have been put forward.[184] According to Carlsson and Berkes (2005) however, the differing definitions and conceptualizations of *co-management* in the literature have some common underpinnings: (a) they explicitly associate the concept of co-management with natural resource management; (b) they regard co-management as some kind of partnership between public and private actors; and (c) they stress that co-management is not a fixed state but a process that takes place

[183] Following Berkes and Folke (1998), a suprise denotes the condition when perceived reality departs *qualitatively* from expectation.
[184] For an overview of the definitions for the concept of co-management confer Carlsson and Berkes (2005).

along a continuum.[185] Ideally, co-management may result in allocation of tasks, exchange of resources, linking different types and levels of organization, reduction of transaction costs, risk sharing, conflict resolution mechanisms and power sharing (Carlsson & Berkes 2005).

The two concepts of adaptive management and co-management are merged in the concept of *adaptive co-management*. As Olsson *et al.* point out: "[a]daptive comanagement combines the *dynamic learning* characteristic of adaptive management (...) with the linkage characteristic of cooperative management (...) and with collaborative management" (Olsson *et al.* 2004: 75). In order to present a more comprehensive picture, the following sections explicate some of the main elements of adaptive co-management.

3.5.2.2 Focus on Social-Ecological Systems

The adaptive co-management-approach puts a focus on the examination of coupled human and natural system, termed *social-ecological systems* (Berkes & Folke 1998).[186] Indeed, the concept of resilience is increasingly interpreted as a way of thinking, a perspective or even paradigm for analyzing social-ecological systems (Folke *et al.* 2002; Anderies *et al.* 2006; Folke 2006).[187] Much research aims at a general theory for the resilience of whole social-ecological systems (Anderies *et al.* 2006). Yet what is a social-ecological system or better: how is it conceptualized?

In a recent article, Liu *et al.* (2007) presented their view of coupled human and natural systems (termed CHANS). Liu *et al.* (2007) define CHANS as systems in which human and natural components interact reciprocally across diverse organizational levels. In order to conceptualize the term "interactions" the authors point to reciprocal effects, feedbacks, indirect effects, emergent properties and spatial/ temporal couplings. Indeed, many scholars contributing to the Resilience

[185] It would be interesting to explore the similarities of the concept of adaptive co-management to the concept of *transdisciplinarity* (Hirsch-Hadorn *et al.* 2006; Scholz *et al.* 2006).

[186] Other terms to signify the coupling of a human and a natural realm are for example "human-ecological system" (Raskin 2008) or "earth system" (Schellnhuber *et al.* 2005).

[187] In fact, from about the 1990s on the whole resilience approach (e.g. definition, theory building, operationalization) focussed on social-ecological systems and formulated their terms, definitions and measures for social-ecological systems. This thesis does not follow this switch from purely ecological systems to coupled social-ecological systems, because it is beyond my competence to examine the resilience of social or economic systems. In addition, I am sceptical as regards to the systems theoretical comprehension of the resilience approach (cf. section 3.5.2.3). In this section however, I do treat social-ecological systems as part of my examination of the adaptive co-management approach.

Alliance think there is a close connection between the human and the natural realm.[188] They even view the nature-culture split as arbitrary and artificial (Berkes & Holling 2002; Westley *et al.* 2002; Berkes *et al.* 2003). It is assumed that coupled social-ecologcial systems are more than just a combination of social and ecological systems. Rather, social-ecological systems are considered to be *coevolutionary units* of social and ecological systems (Folke *et al.* 2003; Liu *et al.* 2007).

In this respect, resilience scholars build on the work of the ecological economist Richard Norgaard. As Norgaard points out:

> "Through the coevolutionary process of development social systems increasingly reflected characteristics of the human influenced ecosystems they inhabited, while ecosystems reflected characteristics of the social systems which affected how individuals interacted with the ecosystems" (Norgaard 1988: 617).

That means, by the occurrence of the reciprocal effects on the one side the inner structure of the social system is formed by the natural habitat while on the other side the natural habitat is modified by the influence of the social system. The term "co-evolution" hereby does not refer to the strict biological meaning, i.e. the interaction of species that brings genetic modifications in both species; but rather the term means in a much broader sense a reciprocal interaction, in which the constitution of the one system determines the character of the other system, and *vice versa*.

This implies that the adaptive co-management-approach assumes a specific *man-nature relationship*. The man-nature relationship of the resilience approach is in its essence structurally analogue to the cultural theory of J. G. Herder (1744-1803). It can thus be interpreted as a scientific reformulation of a counter-enlightenment, conservative ideal of regional idiosyncrasy (cf. more details in section 4.2.4). This dependency of the concept of adaptive co-management on a cultural idea helps to clarify its scope and its applicability.

In my view, a clear conceptualization of the term "social-ecological system" beyond "co-evolution talk" and a reflection on the specific man-nature relationship proposed within the adaptive co-management approach is a far off horizon. This is an apparent weakness of the adaptive co-management approach.

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[188] Some authors have tried to model social-ecological systems (Carpenter *et al.* 1999; Peterson *et al.* 2003b; Schluter & Pahl-Wostl 2007).

3.5.2.3 Application of Ecological Theory to Social Systems

As we have considered in the previous section, the adaptive co-management-approach does not examine ecological systems only, but also social, economic, institutional and, in particular, coupled social-ecological systems. Yet the resilience approach does not only *examine* social and social-ecological systems. Rather, it applies the theory gained from ecological systems to social and social-ecological systems. This is in my view an important position of points that is made within resilience research. Indeed, resilience research takes literally all of its concepts or theory drawn from ecological systems (e.g. adaptive cycle, panarchy, alternative stable states, ecological thresholds, ecological redundancy, ecological memory, confer section 3.2) for achieving understanding in other types of systems.

Consider the following examples. Allison and Hobbs (2004) examine the Western Australian agricultural region – as an example of a large social-ecological system – by means of the adaptive cycle (cf. section 3.2.1.1) The authors describe the historical changes of the region as two iterations of the adaptive cycle and propose that the Western Australian agricultural region is currently in its backloop (release and reorganization phases). Similarly, Kinzig *et al.* (2006) apply the concept of ecological thresholds (cf. section 3.2.3.3) to social and economic systems. The authors hold that social and economic systems can also exhibit alternative stable states and are thus prone to sudden regime shifts.

These two examples provide only a small basis for a general conclusion.[189] Yet the point I want to make here is that the resilience approach and the adaptive co-management hold that it is possible to understand each system type with the same conceptual approach. It is assumed that ecological, social and social-ecological systems can principally be described and analyzed by means of the same conceptual tools. This indicates that resilience research when extending ecological theory to social systems refers to general systems theory.

Tentative Excursus: Resilience Research and General Systems Theory

To my knowledge, resilience research does not explicitly signify itself as systems-theoretical. This section examine the thesis whether resilience research can be

[189] But confer Gunderson and Holling (2002), Berkes *et al.* (2003) and Walker *et al.* (2006) for more examples.

interpreted as a systems-theoretical approach. In order to justify this thesis, in the following I propose some tentative arguments.[190]

The "*Programm einer organismischen Einheitswissenschaft*" (program of an organismic unity of sciences) proposed by Ludwig von Bertalanffy in the 1930s is considered as the beginning of the modern, general systems theory. Even though Bertalanffy aimed for a systems theory that is not restricted to the biological realm, general systems theory remained conncected to the notions of organic biology until the 1940s (Müller 1996). Hereby, Bertalanffy understands the organism as a system that is organised in a hierarchical way and that is in a dynamic equilibrium (Voigt & Weil 2006).

In the 1950s, Bertalanffy defines a generalized concept of "system" as *every complex of interacting elements* (Voigt & Weil 2006). Based on several upcoming scientific disciplines, especially information theory and cybernetics, the terms of Bertalanffy have been extended to other types of systems, e.g. social or technical systems. This "general systems theory" is characterized by a high degree of abstraction. It aims at an innovative approach to theory building that overcomes the divide between the natural sciences and the humanities. Mathematical methods are applied to each type of system, even to those systems that have formerly been understood by hermeneutic methods (Voigt & Weil 2006). One of the most prominent and prevalent approaches to systems theory proposed in the social sciences is provided by the "Theory of Social Systems" by Niklas Luhmann (e.g. Luhmann 1984). It must be noted that the work of Luhmann is contentious and is discussed controversial within the social sciences (Habermas & Luhmann 1976).

General systems theory attempts to overcome the boundaries between different scientific disciplines. It is assumed that (a) the idea of a "whole that is more than the sum of it parts" makes sense and that this "whole" can be examined by scientific methods; (b) "systems" exist; (c) these systems share formal commonalities that allow us to describe, examine and understand them; and that (d) principally every system can be examined by the means of the same theoretical framework, namely general systems theory (Müller 1996; Voigt & Weil 2006). These pivotal characteristics of general systems theory will be important when I interpret resilience research as a systems-theoretical approach in the following.

[190] These arguments are tentative, as I am not very familiar with general systems theory. Confer the work of Deborah Hoheisel (URL: http://www.dbu.de/stipendien_20008/955_db.html) for deeper examinations.

As considered above, resilience research extends the theory gained for ecological systems (e.g. adaptive cycle, panarchy, alternative stable states, ecological thresholds, ecological redundancy, ecological memory) to other types of systems, such as social, economic or socio-ecological systems (Gunderson & Holling 2002; Allison & Hobbs 2004; Kinzig *et al.* 2006). This is a non-trivial position of points. It is not self-evident that the theory gained for ecological systems holds for other types of systems as well. Yet exactly this assumption that resilience theory can be applied to other types of systems is a basic proposition of resilience research. As Walker *et al.* in a recent fundamental theory paper point out: "[t]he ecological and social domains of social-ecological systems can be addressed in a common conceptual, theoretical, and modeling framework" (Walker *et al.* 2006: 6). Thus, the theoretical aspirations and explanatory claims of resilience theory go far beyond ecological systems. Resilience research claims applicability to purely social domains as well (Reusswig 2007). I suggest that this proposition indicates that the resilience approach assumes the validity of general systems theory.

A further indication for the systems-theoretical character of resilience research can be gained via its similarities with other approaches to landscape planning that explicitly signify themselves as systems-theoretical. Indeed, there are some approaches to landscape ecology or landscape planning that (a) are similar to the resilience approach regarding its holistic (and organic) systems notion and the counter-enlightenment theory of the man-nature relationship, and (b) explicitly refer to general systems theory.[191]

For instance, Naveh (1995; 2001) follows a holistic notion of landscapes as concrete, tangible entities. A landscape shoud be conceived as "a natural, self-organizing Gestalt system" (Naveh 1995: 45). Naveh (2001) suggests that "a holistic theory of landscapes cannot be considered in isolation. It has to be based on hierarchical systems view of the world, rooted in general systems theory (...) and in its recent holistic and transdisciplinary insights in organized complexity, self-organization and co-evolution in nature and in human society". According to Naveh, this includes a paradigm shift from the parts to the wholes and from entirely reductionistic and mechanistic approaches to more holistic and organismic ones. In landscapes "like in an organism (or a melody) all their parts are related to each other by the general state of the whole" (Naveh 2001: 273). This notion of landscapes proposed by Naveh

[191] For the analysis and interpretation of the systems notion and the man-nature relationship of the resilience approach cf. section 4.2.

is highly similar to the organic, holistic systems notion proposed within resilience research, as considered in section 4.2.3.1.

In addition, Naveh (2001) views landscapes as complex nature-culture interaction systems, termed “total human ecosystem”. The total human ecosystem integrates humans and their total environment at the highest co-evolutionary level of the global ecological holarchy (Naveh 2001: 275). It “perceives humans and their ecological, cultural, social, political and economic dimensions as an integral part of this highest co-evolutionary geo-bio-anthropo level of the ecological hierarchy *above* the ecosystem level” (ibid: 275), thus creating a postindustrial symbiosis between nature and human society. Apparently, this is again closely related to the theory of the man-nature relationship proposed within resilience research in the form of the coevolution of ecological and social systems within so-called socio-ecological systems (cf. section 4.2.4).

Hence, the approach to landscape ecology proposed by Naveh (1995; 2001) is highly similar to the resilience approach. In addition, Naveh (2001) explicitly refers to general systems theory as the theoretical basis of his approach. This is a further indication - not a proof - that resilience research may be based on general systems theory.

The extension of resilience theory to other types of systems can well be critized. Indeed, the proposition that resilience theory holds for social systems as well is neither being justified by several good reasons nor is it rooted in a theoretical framework. It is just a proposition. Yet the claim that resilience theory holds for social systems as well may well be contentious. As Reusswig (2007: 122) states with respect to the adaptive cycle: “[t]here is social dynamics not covered by the four phases, and there is history not following the cyclical mode of phase interconnectivity. This is not to say that resilience theory is ‘wrong’. The authors of the volume are absolutely right by terming their theoretical insights ‘propositions’, i.e. empirical fallible statements or generalizations from a limited range of cases, waiting for further testing. Precisely for that reason the claim that ‘everything’ (ecosystems, managed ecosystems, social systems) works the way resilience theory is stating simply is an overstatement”. Reusswig (2007: 122) continues: “[m]y recommendation to resilience theory thus would simply be the following: Stick to your ‘propositions’ claim, and abandon the ‘we explain everything’ one.”

In my view, this criticism of resilience research makes a good point. The extension of resilience theory to every type of system is hardly justified. In addition, I infer from the tentative analysis made above that it might be possible to analyze the resilience approach with respect to the merits and threats of general systems theory. More particular, it is possible to critize the extension of resilience theory from ecological systems to other types of systems by referring to general systems theory. Thus, I propose that the applicability of resilience theory to other types of systems is dependent on the validity of general systems theory. To discuss this issue in more detail is well beyond this thesis.

3.5.2.4 Measures of Adaptive Co-Management

In the previous sections I have considered some of the essential characteristics of adaptive co-managment. Altogether it has become clear that adaptive co-management is an approach that tries to combine learning processes with some form of sharing of power and responsibility between stakeholders. In addition, adaptive co-management puts a focus on the examination of coupled social-ecological systems. Yet I also touched upon the limited scope and applicability of the approach due to its systems-theoretical *modus operandi* and its specific and partial comprehension of the man-nature-relationship. With the weaknesses of the approach in mind, the following sections delineate the concrete mamagement measures that follow from applying the adaptive co-management approach to environmental problems.

Folke *et al.* (2002) define adaptive co-management as a process by which institutional arrangments and ecological knowledge are tested and revised in a dynamic, ongoing, self-organized process of learning-by-doing. The aim of adaptive co-management is to examine ways of building resilience to enhance the capacity to deal with change and surprise (Berkes *et al.* 2003). Note that in the context of adaptive co-management the term “resilience” is conceptualized for social-ecological systems and defined as the capacity of social-ecological systems to absorb recurrent disturbances so as to retain essential structures, processes and feedbacks (Adger *et al.* 2005). Even though the adaptive co-management approach stresses the investigation of coupled social-ecological systems, measures for specific dimensions can be distinguished.

3.5.2.4.1 Ecological Dimension

In the ecological dimension of adaptive co-management resilience scholars take a closer look at the concepts of ecological knowledge, biodiversity and small-scale disturbances and their relation to management measures (cf. also Brand 2005: 146ff). I will describe these three points in some detail in the following.

First, according to resilience research ecological knowledge is being provided in a complementary fashion by (a) Western-style science, e.g. biodiversity research or resilience theory, and (b) *traditional ecological knowledge*. The latter is defined as a cumulative body of knowledge and beliefs, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings with one another and with their environment (Berkes and Folke 1998). There is an interesting point here. Berkes and Holling (2002) suggest that traditional practices have certain similarities and parallels to the theory of complex systems, with an emphasis on nonlinear relationships, threshold effects, multiple equilibria, the existence of several stability domains, cross-scale linkages in time and space, disturbance and surprise. Indeed, these practices seem to focus on the back-loop, i.e. the release (Ω) and reorganization (α) phase of the adaptive cycle (cf. section 3.2.1.1) and thus nurture the capacity for renewal and reorganization of the specific system.[192]

There are various examples for management measures based on traditional ecological knowledge (Berkes & Folke 1998). The *caiçaras* of the Brazilian Amazon, Icelandic fishermen and coastal communities in Maine monitor the state of their resources. Some traditional societies perform total protection of certain species, protection of vulnerable life-history stages of a variety of species (societies in South India) or the protection of habitat, e.g. through sacred forests and groves (tribal state of Mizoram in northeastern India). Canadian Amerindian hunters restrict the harvest of game temporarily. Many traditional systems apply multiple species management - such as integrated farming, rotation (Chisasibi Cree hunters) and cultivation systems - use landscape patchiness (Sahelian herders), respond to disturbances and surprises at various scales and nurture sources for renewal. Ecological knowledge is thus regarded as an important goal.

[192] The relevance of traditional ecological knowledge within resilience reearch comes as no surprise, if one intepretes adaptive co-management as a counter-enlightenment, conservative theory of the man-nature relationship. For more details on the cultural relativity of resilience research confer section 4.2.

Second, the maintenance of biodiversity is viewed as a further essential objective for adaptive co-management. As considered in section 3.3 the amount of ecological resilience can be viewed as an interplay of several interacting factors, such as population diversity, functional important types of species, ecological redundancy, response diversity, critical functional groups, biological legacies, source habitats and connectivity.

Third and finally, allowing disturbances at a small temporal and spatial scale is viewed as another important objective. Resilience scholars hold that one of the key features of an adaptive co-management approach to strengthen ecological resilience is to consider small-scale, natural disturbances as *intrinsic* part in the internal dynamics of ecosystems (cf. more details in section 3.3.2.3.1) (Folke *et al.* 1998; Colding *et al.* 2003). In order to ensure the ecological resilience of ecosystems at larger scales (e.g. region), appropriate management approaches encourage and speed-up the renewal and destruction of systems at smaller scales. Otherwise small-scale disturbances can accumulate and cascade up driving whole landscapes or regions into undesirable states or basins of attraction. The aim of adaptive management is therefore to prevent the build-up of large-scale crisis (Folke *et al.* 1998). This type of management is termed *backloop management*, as it indirectly considers the release-reorganization phases of the adaptive cycle (Colding *et al.* 2003). As Colding *et al.* point out:

> "[i]n backloop management, natural disturbances become an integrated part of manipulating and modifying the natural resource base, and managers actively respond to episodic or rare events using flexible institutions and management practices that reduce risk that large-scale ecological crisis will occur" (Colding *et al.* 2003: 164).

In forest ecology for instance, there is consensus that one of the best ways to preserve biodiversity in managed forests is to mimic the natural disturbance regime (Frelich & Reich 1998).

Local resource users may actively create small-scale disturbances within the landscape. Many traditional societies nurture sources of ecosystem renewal by creating small-scale disturbances (Folke *et al.* 1998). For instance, traditional agro-forestry practices such as shifting cultivation create forest gaps and enable people to produce crops or enhance the supply of wild foods without disrupting natural renewal processes, African herders use pulse grazing by migratory cattle to prevent the shift from semi-arid grasslands to an unproductive regime (Berkes & Holling 2002), while Amerindians of Northern Alberta, Canada, and Australian aborigines use fire on a patchy scale to improve the feeding habitat for game and to prevent the invasion of shrub species. Similarly, fire management in contemporary forest and protected area management uses controlled burning of grass and deadwood which reduces the spread of accidental, large-scale fires by preventing the slow build-up of fuel (Colding *et al.* 2003). By mimicking fine-scale natural disturbances, these practices help avoid the accumulation of disturbance that move across scales further up in the panarchy (Berkes & Holling 2002). Berkes *et al.* (2003) propose the general principle that resource and environmental management that suppresses disturbance and biodiversity will be "unsustainable".[193]

3.5.2.4.2 Social, Institutional and Organizational Dimension

According to Olsson *et al.* (2004), the social, institutional and organizational dimensions of managing for the resilience of social-ecological systems should be approached as carefully as the ecological dimension. Scholars who work on the social dimensions of adaptive co-management identify a myriad of elements that are considered as essential for building social-ecological resilience.[194]

In my view however, the treatment of the social dimension within adaptive co-management represents a mere rag rug, that means, a relatively unrelated and inconsistent aggregation of suggestions. Hereby, Lebel *et al.* (2006) accurately view the problems of measurement (i.e. operationalization) and causality regarding the concept of social-ecological resilience. Up to now, there are no clear measures or pathways formulated in the relevant literature. I suggest that this is due to the immense conceptual confusion on the term resilience in general and in particular on the term social-ecological resilience (cf. section 4.1) (Brand & Jax 2007). Nevertheless I will expand on the essential elements of the social dimension in the following for the sake of completeness.

The following elements are identified as being conducive for social-ecological resilience : (i) visionary leadership, as individual actors serve as key players in institution building and organizational change in relation to ecosystem dynamics and

[193] Again, the particular concept of sustainability is highly unclear within resilience research.

[194] For an overview of the social dimension of adaptive co-management confer Olsson *et al.* (2004), Folke *et al.* (2005) and Lebel *et al.* (2006).

facilitate horizontal and vertical linkages in the adaptive comanagament process", (ii) trust, as trust lubricates collaboration, (iii) enabling legislation that creates social space for ecosystem management, (iv) funds for responding to environmental change and remedial action, (v) monitoring and responding to environmental feedback, (vi) information flow and social network building, (vii) combining various sources of information, (viii) sense-making, (ix) arenas of collaborative learning, (x) organizational learning, (xi) polycentric institutional arrangements, (xii) social memory and (xiii) bridging organizations (Olsson *et al.* 2004; Folke *et al.* 2005).

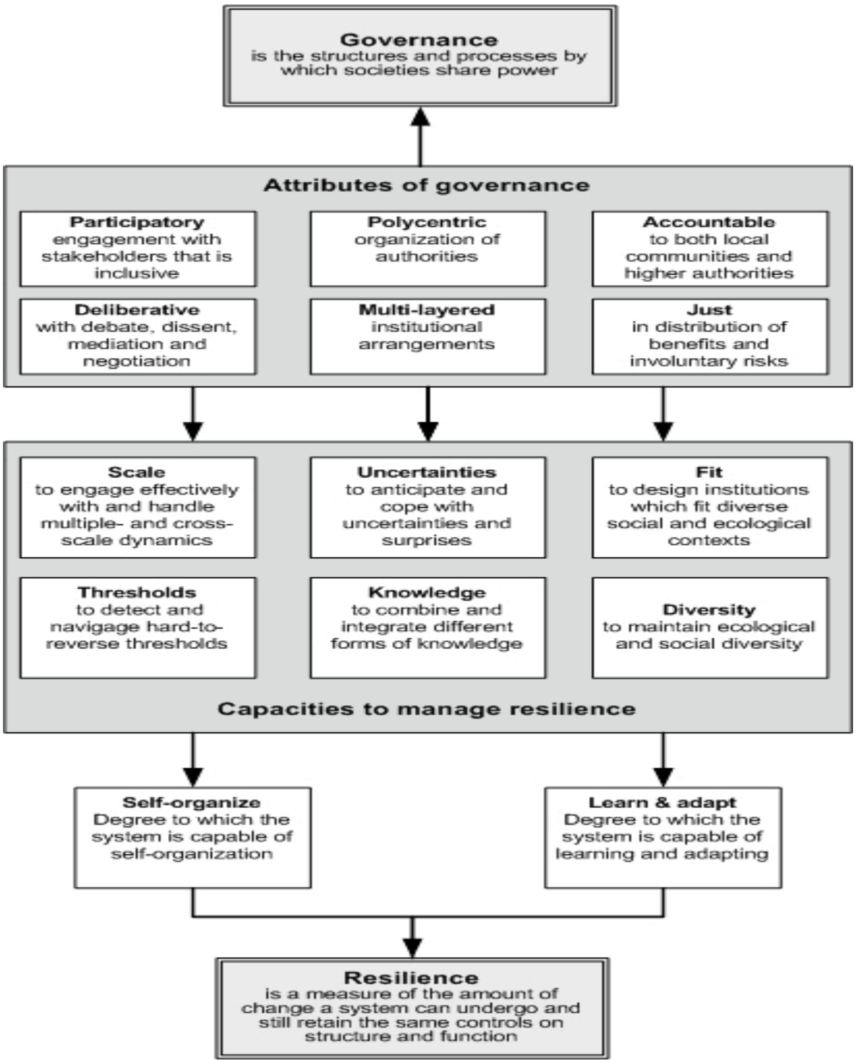

Figure 18: The link between governance attributes and social-ecological resilience
from Lebel *et al.* 2006: 3

Folke *et al.* (2005) conclude with the subsequent, conceptually vague recommendations: build knowledge and understanding of resource and ecosystem dynamics; feed ecological knowledge into adaptive management practices; support flexible institutions and multilevel governance systems; and deal with external perturbations, uncertainty and surprise. Still complementary to that, Lebel *et al.* (2006) identify the attributes of governance that promote the resilience of social-ecological systems: (i) participation and deliberation; (ii) polycentric institutions and (iii) accountable and just authorities. *Figure 18* illustrates the suggestions of Lebel *et al.* (2006) for the link between governance attributes and social-ecological resilience.
In addition, Anderies *et al.* (2004) analyze social-ecological systems from an institutional perspective. As a starting point, the authors propose a framwork that includes (*a*) defining the social-ecological system, (*b*) highlighting the key drivers, such as strategic interactions, operational rules and collective-choice processes, and (*c*) the specification of robustness[195] in social-ecological systems. The authors remain vague on the specific characteristics and measures of robustness.

3.5.2.4.3 Measures for Whole Social-Ecological Systems

Several publications explicitly refer to the resilience of coupled social-ecological systems. In correspondence to the conceptual confusion on the terms "resilience" and "social ecological resilience", the suggestions remain vague and tentative. For instance, Berkes and Seixas (2005) and Berkes (2007) identify the following essential elements for building social-ecological resilience: (i) learning to live with change and uncertainty; (ii) nurturing diversity for reorganization and renewal; (iii) combining different kinds of knowledge; and (iv) creating opportunity for self-organization. Complementery to that, Berkes and Seixas (2005) list other relevant aspects: learning from crisis, responding to change, nurturing ecological memory, monitoring the environment, building capacity for self-organization and conflict management. Furthermore, Hughes *et al.* (2005) suggest several key components of resilience for marine regions. These include leadership and insight, sustained mobilization of national and international aid, cultural and ecological diversity, development of multi-scale social networks, and the resolution of local civil unrest.

[195] Anderies *et al.* (2004) define *robustness* as "the maintenance of some desired system characteristics despite fluctuations in the behavior of its component parts or its environment". This comes close to the definition of ecological resilience.

3.5.3 Concluding Remarks

The measures to increase the social-ecological resilience in a social-ecological system (SES) described in the previous sections result in *adaptive capacity*, which is defined rather vague as the capacity of a SES to respond to and shape change (Folke *et al.* 2002). As Gunderson put it:

> "Resilience in the ecosystem sense provides SESs with the ability to persist in the face of shocks and disturbances. Maintaining a capacity for renewal in a dynamic environment provides an ecological buffer that protects the system from the failure of management actions that are taken based upon incomplete understanding, and therefore allows managers to affordably learn and change" (Gunderson 2003: 34).

And Carpenter *et al.* add:

> "[a]ny institution that gathers better information on slow variables, puts more weight on future returns, narrows the distribution of uncertainties, maintains social flexibility for adaptive response, and maintains the resilience of ecosystems to withstand novel perturbations has the potential to ameliorate the risk of collapse" (Carpenter *et al.* 2002: 193).

Following Raskin (2008), adaptive capacity is the *sine qua non* of any long-lived SES.

I want to stress here, however, that the literature on "adaptive co-management for building resilience in social-ecological systems" is rather vague and an accumulation of a variety of ideas.[196] This comes as no suprise, as the central terms "adaptive co-management", "resilience" and "social-ecological system" are vague and lack a sound conceptualization. As a consequence, the extension of the terms "adaptive co-management" and "social-ecological resilience" becomes very wide and the terms loose specific meaning. This, in turn, leads to an arbitrariness in the use of the terms. In my view, this is a great weakness in the conceptualization of the concept of resilience and adaptive co-management (cf. also section 4.1.4) (Brand & Jax 2007). Not to mention that there is a lack within resilience research to consider pivotal insights, concepts and theories gained within the social sciences and the humanities,

[196] In addition, adaptive co-management rests on ethical theories. Yet the relation of ethical theories to adaptive co-management is widely unexplored (Fennell *et al.* 2008).

such as general systems theory (Luhmann), discourse theory (Habermas), ethical theories or the concept of “transition management” (Kemp & Martens 2007).

Despite the weaknesses, the adaptive co-management approach can be applied to many environmental management situations characterized by a high degree of uncertainty. I have applied this approach to small-scale fisheries in developing countries in section 5.3.4.2.

4 Ecological Resilience: Some Conceptual Issues

The “anatomy” of the concept of resilience has been described in the previous chapter including definition, background theory, resilience mechanisms, operationalization and management-approach. These explanations provide the basis for the subsequent descriptions. This chapter takes a closer look at the definition and conceptual structure of the concept of resilience (section 4.1) and the cultural premises and therefore validity domain of resilience research (section 4.2).

4.1 Focussing the Meaning(s) of Resilience

This section reviews the concept of resilience regarding its definition and conceptual structure. It is based on an article developed with my senior supervisor Kurt Jax and published in the journal "Ecology & Society" (Brand & Jax 2007). In the following I will quote the original article in its full length and will include some new ideas and descriptions when appropriate.

4.1.1 Introduction

The concept of “resilience” is one of the most important research topics in the context of achieving sustainability (Perrings *et al.* 1995; Kates *et al.* 2001; Foley *et al.* 2005). First introduced as a descriptive ecological term (Holling 1973), resilience has been frequently redefined and extended by heuristic, metaphorical or normative dimensions (Holling 2001; Ott & Döring 2004; Pickett *et al.* 2004; Hughes *et al.* 2005). Meanwhile, the concept is used by various scientific disciplines as an approach to analyze ecological as well as social-ecological systems (Anderies *et al.* 2006; Folke 2006). As such, it promotes research efforts across disciplines and between science and policy.

However, both conceptual clarity and practical relevance are critically in danger. The original *descriptive* and *ecological* meaning of resilience is diluted as the term is used ambiguously and in a very wide extension. This is due to the blending of descriptive aspects, i.e. specifications of what *is* the case, and normative aspects, i.e. prescriptions what *ought to be* the case or is desirable as such. As a result, difficulties to operationalize and apply the concept of resilience within ecological science prevail. This, in turn, impedes progress and maturity of resilience theory (Pickett *et al.* 1994: 57ff). The success of the concept in stimulating research across disciplines on the one side and the dilution of the descriptive core on the other raises the fundamental question what conceptual structure we want resilience to have.

This section is divided into four parts. Section 4.1.2 offers a typology to structure the numerous definitions of resilience proposed within sustainability science. Using this typology as a background, the section 4.1.3 investigates in more detail a descriptive, ecological concept of resilience viewed from both a formal and an operational perspective. Subsequently, the section 4.1.4 examines the use of resilience as a rather vague boundary object and points to some chances and pitfalls. Section 4.1.5 concludes with final thoughts on the recent conceptual development and a fruitful conceptual structure of resilience.

4.1.2 A Typology for Definitions of Resilience

In what follows we suggest a typology for the variety of definitions of resilience used in sustainability science. The typology is based on the analysis of key papers published in the last 35 years (Janssen *et al.* 2006; Janssen 2007). It provides the background for discussing the conceptual development of resilience.

Before turning to the definitions in detail some words on the terminology used are in order. First of all, two basically distinct meanings of "resilience" must be distinguished. The first one refers to dynamics close to equilibrium and is defined as the time required for a system to return to an equilibrium point following a disturbance event. It has been coined "engineering resilience" (Holling 1996) and is largely identical to the stability property "elasticity" (Grimm & Wissel 1997). The second meaning of resilience refers to dynamics far from any equilibrium steady state and is defined as the amount of disturbance that a system can absorb before changing to another stable regime, which is controlled by a different set of variables and characterized by a different structure. It is this second kind of resilience - dubbed ecological resilience - to which we refer in this text.

The result of our analysis is displayed in *Table 8.* It shows three categories, ten classes and correspondingly ten definitions of "resilience". The three categories reflect whether the definition is in accordance with either a *genuinely-descriptive concept* (category I), a *hybrid concept*, where descriptive and normative connotations are inter-mingled (category II) or a *genuinely-normative concept* (category III). Thus, our scheme in the first place emphasizes the *degree of normativity* included in the different definitions that fit under the overall category of "resilience" as characterized above. However, we also found it useful and necessary to distinguish between purely ecological definitions (*class 1 - 4*) and those which are (also) used in the context of other fields, such as economy and sociology (*class 5 – 10*).

Table 8: The levels of meaning of ecological resilience

Categories & classes	*Definitions*	*References*
(I) DESCRIPTIVE CONCEPT		
(Ia) ECOLOGICAL SCIENCE		
1) Original-ecological	Measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables	Holling 1973: 14
2) Extended-ecological	The magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour & The capacity of a system to experience shocks while retaining essentially the same function, structure, feedbacks, and therefore identity	Gunderson & Holling 2002: 4 Walker *et al.* 2006: 2
2a) Three characteristics	capacities i) to absorb disturbances, ii) for self-organization and iii) for learning and adaptation	Walker *et al.* 2002
2b) Four aspects	1) latitude (width of the domain), 2) resistance (height of the domain), 3) precariousness 4) cross-scale relations	Folke *et al.* 2004: 573
3) Systemic-heuristic	Quantitative property that changes throughout ecosystem dynamics and occurs on each level of an ecosystem's hierarchy	Holling 2001
4) Operational	Resilience *of* what *to* what? & The ability of the system to maintain its identity in the face of internal change and external shocks and disturbances	Carpenter *et al.* 2001 Cumming *et al*. 2005
(Ib) SOCIAL SCIENCES		
5) Sociological	The ability of groups or communities to cope with external stresses and disturbances as a result of social, political and environmental change	Adger 2000: 347
6) Ecological-economic	Transition probability between states as a function of the consumption and production activities of decision makers & The ability of the system to withstand either market or environmental shocks without loosing the capacity to allocate resources efficiently	Brock *et al.* 2002: 273 Perrings 2006: 418
(II) HYBRID CONCEPT		
7) Ecosystem-services-related	The underlying capacity of an ecosystem to maintain desired ecosystem services in the face of a fluctuating environment and human use	Folke *et al.* 2002: 14

8) Social-ecological system		
8a) Social-ecological	The capacity of linked social-ecological systems to absorb recurrent disturbances (…) so as to retain essential structures, processes and feedbacks	Adger *et al.* 2005: 1036
8b) *Resilience-approach*	A perspective or approach to analyze social-ecological systems	Folke 2006
(III) NORMATIVE CONCEPT		
9) Metaphoric	Flexibility over the long term	Pickett *et al.* 2004: 381
10) Sustainability-related	Maintenance of natural capital in the long run	Ott & Döring 2004: 213f

In the following each definition of resilience is explained in more detail with respect to its category and class, respectively. Note that the proclaimed titles do not correspond to the particular references.

Category I: Descriptive concept

Sub-category Ia: Ecological Science

Class 1: Original-ecological definition.

In his seminal paper Holling (1973) defines resilience as a "measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables" (Holling 1973: 14). In this original-ecological meaning resilience focuses on the persistence of populations (or communities) at the ecosystem level and corresponds to both the overall area and the height of the lowest point of a population's domain of attraction. A relative measure is a population's probability of extinction.

Class 2: Extended-ecological definition.

Subsequent work published from the late 1980s (Holling 1986; Walker *et al.* 1999; Gunderson 2000; Gunderson & Holling 2002; Walker *et al.* 2004) is strongly influenced by theory on complex adaptive systems (e.g. Levin 1998) including the cross-scale morphology of ecosystems (Holling 1992a). According to the extended keystone hypothesis originally proposed by Holling (1992a), the hierarchical structure of ecosystems is primarily regulated by a small set of ecosystem processes each operating over different scale ranges. Important changes in ecosystem dynamics can be understood by analyzing a few, typically no more than five, key variables (Walker *et al.* 2006).[197] In this interpretation, the scientific focus is on the critical structure and processes of an ecosystem. Individual species can be replaced if the critical structure and key processes persist (Walker *et al.* 1999; Elmqvist *et al.* 2003; Nyström 2006). In this extended-ecological meaning, resilience is defined as "the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour" (Gunderson & Holling 2002: 4) or "the capacity of a system to experience shocks while retaining essentially the same function, structure, feedbacks, and therefore identity" (Walker *et al.* 2006: 2).

Class 2a: Three characteristics.

Some authors interpret the extended-ecological meaning as comprising three characteristics. Those are: (1) the amount of change a system can undergo and still remain within the same domain of attraction, i.e. to retain the same controls on structure and processes, (2) the degree to which the system is capable of self-organization and (3) the degree to which the system expresses capacity for learning and adaptation (Carpenter *et al.* 2001; Walker *et al.* 2002; Folke 2006).

Class 2b: Four aspects.

One line of research emphasizes the concept of *alternative stable regimes* (Scheffer & Carpenter 2003; Folke *et al.* 2004; Walker & Meyers 2004). Note that the term "regime" is preferred to avoid the static connotations of the term "state" and to describe the actual *dynamic* situation of a specified ecosystem (Scheffer & Carpenter 2003). Formally, alternative stable regimes exist within *alternative basins of attraction* (Walker *et al.* 2004). Four aspects of a basin of attraction are crucial.[198] Those are: (1) *latitude* or the maximum amount the system can be changed before losing its ability to recover (the width of the basin), (2) *resistance*, which matches the ease or difficulty of changing the system (the topology of the basin), (3) *precariousness*, i.e. the current trajectory of the system and proximity to a limit or threshold and (4) *cross-scale relations* or how the above three aspects are influenced by the dynamics of the

[197] Cf. section 3.2.2.
[198] Cf. section 3.2.3.3.4.1.

systems at scales above and below the scale of interest (Folke *et al.* 2004; Walker *et al.* 2004).

Class 3: Systemic-heuristic definition.

Some scholars have worked on the presuppositions of the concept of resilience, which include a heuristic for the dynamics of productive, self-organized systems, the *panarchy*. This meta-model (Cumming & Collier 2005) of ecosystem dynamics consists of four-phase adaptive cycles (r-, K-, Ω- and α-phases) that occur on each level of a system's hierarchy.[199] Against this background, resilience represents a quantitative property that changes throughout the adaptive cycle and principally occurs on each level of a system's hierarchy (Holling 2001; Gunderson & Holling 2002).

Class 4: Operational definition.

To apply the concept of resilience to empirical cases, it is critical to specify resilience *of* what *to* what (Carpenter *et al.* 2001). This operational definition constitutes the first step to make resilience concrete. Further operational steps suggest to focus on the concept of identity and define resilience as "the ability of the system to maintain its identity in the face of internal change and external shocks and disturbances" (Cumming *et al.* 2005).

Sub-category Ib: Social Sciences

Class 5: Sociological definition.

Some scientists apply the concept of resilience to social systems. Social resilience is defined as "the ability of groups or communities to cope with external stresses and disturbances as a result of social, political and environmental change" (Adger 2000: 347).

Class 6: Ecologic-economical definition.

In addition, the concept of resilience is used to analyze economy-environment systems (Perrings & Walker 1997; Perrings & Stern 2000; Brock *et al.* 2002; Perrings 2006). Resilience matches the "transition probability between states as a function of

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[199] Cf. section 3.2.1.1.

the consumption and production activities of decision makers" (Brock *et al.* 2002: 273) or "the ability of the system to withstand either market or environmental shocks without loosing the capacity to allocate resources efficiently" (Perrings 2006: 418).

Category II: Hybrid concept

Class 7: Ecosystem-services-related definition.

In this hybrid sense, resilience corresponds to the underlying capacity of an ecosystem to maintain desirable ecosystem services in the face of human use and a fluctuating environment (Folke *et al.* 2002). Studies focus on *desirable* ecosystem services of an ecological system, e.g. food production, water purification or aesthetic enjoyment (UNEP 2005a).

Class 8: Social-ecological system.

Many scientists state that it is critical to apply the concept of resilience to coupled social-ecological systems, as it may be a fundamental error of environmental policy to separate the human system from the natural system and treat them as independent (Folke *et al.* 2002; Anderies *et al.* 2006; Walker *et al.* 2006). The nature-culture split is seen as arbitrary and artificial; humans are regarded as part of the ecosystem (Westley *et al.* 2002; Berkes *et al.* 2003).[200]

Class 8a: Social-ecological definition.

Social-ecological resilience is defined as "the capacity of social-ecological systems to absorb recurrent disturbances (…) so as to retain essential structures, processes and feedbacks" (Adger *et al.* 2005: 1036). In this approach, a system analysis tends to incorporate specific values, e.g. cultural diversity or international aid. Consequently, there is an increase in the degree of normativity, i.e. resilience gets more and more desirable as such.

Class 8b: Resilience-approach.

Recently, resilience has been increasingly conceived as a perspective, as a way of thinking to analyze linked social-ecological systems (Folke 2006). No clear definition

[200] Cf. section 3.5.2.2.

is suggested. Rather, resilience is conceived as a collection of ideas about how to interpret complex systems (Anderies *et al.* 2006).

Category III: Normative concept

Class 9: Metaphorical definition.

In a metaphoric interpretation, the concept of resilience means “flexibility over the long term” (Pickett *et al.* 2004: 381) and is viewed as desirable as such.

Class 10: Sustainability-related definition.

Resilience has been suggested as to be one of the guidelines for a conception of strong sustainability (Ott & Döring 2004).[201] Hereby the term refers to the maintenance of natural capital in the long-term in order to provide ecosystem services that provide instrumental as well as eudaimonistic values for human society.

These ten[202] definitions together represent the intension of the term “resilience”. Even though they are all related to the original, descriptive concept of resilience, as introduced by Holling (1973), the term has been transformed considerably. The conceptual development of resilience has been recently reviewed by Folke (2006), who made a distinction between an early interpretation of resilience, which focuses on the robustness of systems to withstand shocks while maintaining function (*ecosystem* or *ecological resilience*, *social resilience*), and a subsequent interpretation, which refers more to the interplay of disturbance and reorganization within a system as well as to transformability, learning and innovation (*social-ecological resilience*). While Folke (2006) points to the change in the specific meaning of resilience our own interpretation of the conceptual development of resilience highlights the distinct use of the concept of resilience within the spectrum of scientific disciplines. Thus, the subsequent sections contrast (*a*) a clearly specified concept of resilience that is merely used in ecology with (*b*) a vague and malleable concept of resilience that is used as a communication tool across different scientific disciplines and between science and practice.

[201] Cf. section 5.1.4.1.

[202] An eleventh meaning of resilience has recently been put forward by Marschke and Berkes (2006). They consider the concept of “well-being” as a substitute for resilience. Yet no clear definition is provided.

4.1.3 Resilience as a Descriptive Ecological Concept

This section describes a descriptive, ecological concept of resilience in more detail. By definition a descriptive concept of resilience excludes normative dimensions. Resilience may be viewed as either desirable or undesirable in a specific case; this depends on the state *of concern*. This means, a degraded savannah or a polluted lake can be highly resilient but at the same time undesirable from an anthropocentric perspective (Carpenter *et al.* 2001; Carpenter & Cottingham 2002; Walker *et al.* 2002).

In a descriptive sense, the concept of resilience points to a non-equilibrium view on ecological systems (Wallington *et al.* 2005), that is, it assumes the existence of alternative stable regimes. For example, a savannah may exhibit either a locally stable grassy regime or a locally stable woody regime depending on the value of some driving factors, such as rainfall, grazing pressure and fire events (Walker 2002). There is strong evidence that many ecosystem types can exist in alternative stable regimes, for instance lakes, coral reefs, deserts, rangelands, woodlands and forests (Folke *et al.* 2004; Walker & Meyers 2004). Yet the weight of empirical evidence shows that the *relative frequency* of the occurrence of alternative stable regimes across systems is higher for systems controlled by environmental adversity, e.g. deserts, arctic tundra or savannahs, than those controlled by competitive adversity, e.g. forests or coral reefs (Didham & Norton 2006) (confer for further details section 3.2.3).

A mathematical model of this behaviour termed “bistability” is provided by phase plane and bifurcation diagrams proposed by Ludwig *et al.* (1997; 2002). Formally a system exhibits alternative basins of attraction when a fast state variable (e.g. annual grasses, macrophytes) responds to changes in a slow variable (e.g. long-lived organisms, nutrient storages) by a backwards folding curve, as shown in *Figure 19*. Because of the backward fold, two stable basins overlap, separated by an unstable one over a given range of the slow variable (Scheffer *et al.* 2001; Scheffer & Carpenter 2003; Schröder *et al.* 2005).

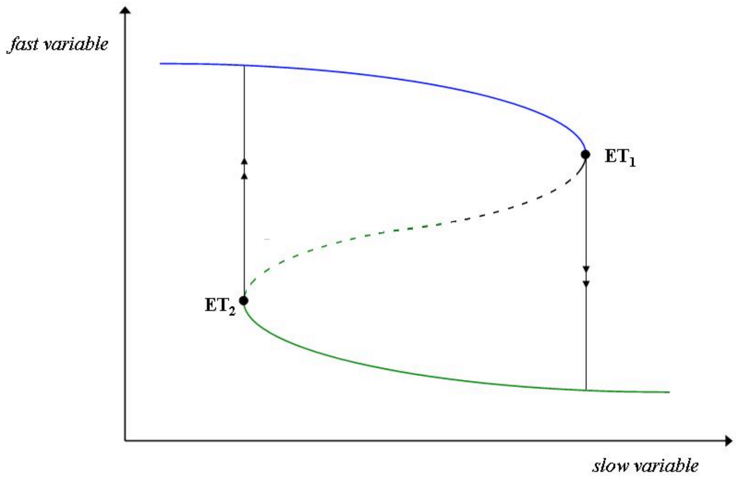

Figure 19: Bifurcation diagram of a system described by a fast variable and a slow variable: the stable regimes are given by the blue and green solid lines and the boundary of the basins of attraction (unstable state) by the dashed line. ET_1 and ET_2 represent ecological threshold points
modified from Scheffer & Carpenter 2003

When the system is in a regime on the upper blue branch of the folded curve in *Figure 19,* it can not pass to the lower green branch smoothly. Instead, when the slow variable changes sufficiently to pass the critical value, i.e. the ecological threshold (ET_1), a catastrophic transition to the lower branch occurs, either caused by only an incremental change in conditions or due to a bigger disturbance. To induce a switch back to the upper branch it is not sufficient to restore the slow variable to the value before the collapse. Instead, one needs to go back further, beyond the other switch point (ET_2), where the system recovers by shifting back to the upper branch – a pattern known as *hysteresis* (Scheffer & Carpenter 2003; Briske *et al.* 2006; Groffman *et al.* 2006). In contrast to a narrow equilibrium view, this indicates the importance of the boundaries of a basin of attraction and the ease or difficulty with which a system could be moved out of this basin (Holling 1973; Gunderson & Holling 2002).

For example, shallow lakes can exhibit two stable regimes with respect to nutrient load, i.e. a clear-water regime with aquatic plants and a turbid regime without vegetation. If the lake is in the clear-water regime, an increase of the nutrient level will lead to a gradual and moderate rise in turbidity until the critical turbidity for plant survival is reached. At this point, vegetation collapses and the lake shifts to the turbid regime. Reduction of nutrients after this catastrophic transition does not result in a return of plants immediately. However, the backward switch happens at much lower nutrient level than the forward switch. Thus, often reduction of the nutrient level to values at which the lake used to be clear and vegetated will not lead to restoration of that state (Dent *et al.* 2002).

Ecological resilience thus represents a measure for “stability” applied to ecological systems. Yet “stability” represents a meta-concept that covers a variety of stability properties, and correspondingly, stability concepts (Loreau *et al.* 2002). Grimm *et al.* (1992) count 163 definitions from 70 different stability concepts and more than 40 measures in the relevant ecological literature. Yet many of these concepts in fact signify the same thing. Grimm and Wissel (1997) identify only three fundamentally different properties while all other stability concepts in the relevant ecological literature can be defined in terms of these three stability properties: (1) “staying essentially unchanged” dubbed *constancy*, (2) “returning to a reference state (or dynamic) after a temporary disturbance” termed *resilience*, and (3) “persistence through time of an ecological system” labelled *persistence*. Three other aspects of the three fundamental properties are named so frequently in the relevant literature that they must be accepted as individual concepts, even though they can be defined in terms of the three fundamental concepts (Grimm & Wissel 1997): (4) “staying essentially unchanged despite the presence of disturbances” or *resistance* (an interpretation of property 1), (5) “speed of return to the reference state (or dynamic) after a temporary disturbance” dubbed *elasticity* (an aspect of property 2), and (6) “the whole of states from which the reference state (or dynamic) can be reached again after a temporary disturbance” termed *domain of attraction* (a further aspect of property 2).

Similar to Grimm and Wissel (1997), Hansson and Helgesson (2003) identify three basic stability properties in the relevant literature of both natural and social sciences, while using a different terminology compared to that of Grimm and Wissel (1997). The first property identifies (1) a system that remains, during a particular period of time, in a specific proper subset of a set of states which is dubbed *constancy*. This property describes what actually happens, not a tendency or what could have happened, had the circumstances been different. The second is represented by the (2) “tendency of a system to remain unchanged, or nearly unchanged, when exposed to perturbations" or *robustness*. Finally, the third property is (3) the "tendency of a system to recover or return to (or close to) its original state after a perturbation" termed *resilience*.[203]

What is the relation of the concept of ecological resilience to these stability properties identified in Grimm and Wissel (1997) and Hansson and Helgesson (2003)? As considered in section 3.1, I define ecological resilience as *the capacity of an ecosystem to resist disturbance and still maintain a specified state.* In this interpretation, the concept comes close to the stability concepts "resistance" (Grimm & Wissel 1997) or "robustness" (Hansson & Helgesson 2003).[204] This may underestimate other important characteristics of ecological resilience, such as the capacities for renewal, reorganization and development (Folke 2006). Yet this definition is in my view useful and workable to be used for the measurement of ecological resilience in real-world ecosystems. The descriptive ecological definitions described above (*class 1 – 4* in *Table 8*) differ with respect to the criteria they provide as means to determine if a system is resilient and to what degree. In this section we focus on the *extended-ecological definition* of resilience in order to point to the concept of slow controlling variables, which can be used to operationalize resilience, and thus to the importance of a quantitative and measurable approach to resilience.

Indeed, a crucial question for scientific progress is: are there any possibilities to estimate or measure the resilience of an ecosystem? Any operational interpretation of resilience means to specify resilience *to* what and *of* what (cf. *class 4* in *Table 8*) and channels into a comprehensive *resilience analysis*. This also means to inquire which of the criteria for resilience described in definition *classes 1 – 4* are in fact meant as criteria that must be measured to assess and/ or quantify the resilience of an ecosystem.[205]

The *to-what part* of the analysis explicates to what exactly a certain regime of an ecosystem should be resilient to. This corresponds to specifying the *disturbance regime*, e.g. the kind of disturbances, their frequency and intensity (Pickett & White 1985a: 6ff; White & Jentsch 2001), which may include both human disturbances, e.g. pollution pulses or habitat fragmentation, and natural disturbances, e.g. hurricanes or floods, as well as possible multiplicative effects (Vinebrooke *et al.* 2004).

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[203] Certainly, there is a variety of earlier classifications of stability concepts in the relevant literature (Orians 1975; Pimm 1984). Yet this thesis is based on the more recent classifications, as I view them as well-founded.

[204] The question whether the term "ecological resilience" includes other stability concepts apart from "resistance" or "robustness" is analyzed in some detail in Brand (2005: 113ff).

[205] For the operationalization of resilience cf. section 3.4.

The subsequent *of-what part* explicates the specific regime that is meant to be resilient to the identified disturbance regime. This part of the analysis is comprised of several steps. *Step one* means to assess which ecosystem services of the regime are of primary concern on which spatial and temporal scale (Walker *et al.* 2002). *Step two* explicates the (self-)identity of the selected regime delimited in step one. This includes to explicate the precise boundary of the regime, the set of variables of interest, the expected internal degree of relationships as well as the component resolution (Jax 2006). Step one and step two are dependent on societal values or normative judgements and should therefore incorporate environmental assessment procedures and participative deliberations (Jax & Rozzi 2004).

Apparently, the overall aim is to assess if the ecosystem is and remains within the regime of concern, which has been identified in the previous steps. Thus, *step three* specifies the variables and mechanisms that control the specific position of an ecosystem within state space. There are two options. The first option is to investigate empirically the value of the slow variables of a regime and plot it in a bifurcation diagram, as considered in *Figure 1*. The amount of resilience is then measured as the distance between the current value of the slow variable and the critical value (Carpenter *et al.* 2001; Peterson *et al.* 2003b), which is termed *precariousness* (Walker *et al.* 2004). It may be possible to predict the position of an ecological threshold either by studying return time and standard deviations of a fast variable (Wissel 1984; Carpenter & Brock 2006) or by the repeated calculation of the Fisher information of a regime (Mayer *et al.* 2006).

The second option of step three refers to the amount of *resilience mechanisms* inherent in the desirable regime. Resilience mechanisms include (*a*) critical functional groups and functional important species, such as top predators or keystone species (Bellwood *et al.* 2004; Folke *et al.* 2004), (*b*) ecological redundancy and response diversity within functional groups (Nyström 2006), and (*c*) the existence of a matrix of support areas at the landscape scale that provide potential colonists to compensate for the loss of species at the local scale (Bengtsson *et al.* 2003). The amount of resilience mechanisms may be a measure to assess the relative resilience of the desirable regime to the given disturbances (Allen *et al.* 2005) (confer for a detailed treatment of the operationalization of ecological resilience section 3.4).

The message of this section is that in a descriptive interpretation resilience can be a clearly specified and delimited stability concept. It is in this sense that resilience represents a quantitative and measurable concept that can be used for achieving progress in ecological science.

4.1.4 Resilience as a Boundary Object

In contrast to the use as a descriptive concept, resilience is increasingly viewed in a rather vague and malleable meaning. In the 1990s several scholars discovered the concept as an important tool to measure sustainability (Arrow *et al.* 1995; Perrings *et al.* 1995). Since then resilience has been used by various scientific disciplines, for instance economics (Farber 1995; Batabyal 1998; Brock *et al.* 2002; Perrings 2006), political science (Olsson *et al.* 2006), sociology (Adger 2000) or planning (Pickett *et al.* 2004) and each discipline has provided specific definitions (cf. *class 5 – 10* in Table 1). Moreover, resilience has been related to other scientific concepts such as carrying capacity (Seidl & Tisdell 1999), critical natural capital (Deutsch *et al.* 2003), strong sustainability (Ott & Döring 2004), globalization (Armitage & Johnson 2006), justice (Adger 2003) and adaptive co-management (Olsson *et al.* 2004).[206]

In particular, resilience is increasingly interpreted in a broader meaning across disciplines as a way of thinking, a perspective or even paradigm for analyzing social-ecological systems (Folke *et al.* 2002; Folke 2006; Walker *et al.* 2006; Walker & Salt 2006). Some authors expand theories or concepts drawn from ecological systems, e.g. alternative stable regimes, panarchy or ecological redundancy, to examine social, political and institutional systems (Allison & Hobbs 2004).[207] Much research aims at a general theory for the resilience of whole social-ecological systems (cf. *class 8* in *Table 8*). It is in this sense that resilience incorporates the capacity of social-ecological systems to cope with, adapt to, and shape change and learn to live with uncertainty and surprise (Folke 2003, 2006).

Thus, we suggest that resilience has become a *boundary object*. Within the field of *Science and Technology Studies*, this signifies a term that facilitates communication across disciplinary borders by creating shared vocabulary although the understanding of the parties would differ regarding the precise meaning of the term in question (Star & Griesemer 1989). Boundary objects are able to co-ordinate different groups without a consensus about their aims and interests. If they are both open to interpretation and valuable for various scientific disciplines or social groups, boundary

[206] Note that there is an interesting body of work within ecological economics on the modelling of economy-environment systems (Perrings 1998; Perrings & Stern 2000; Perrings 2006). Unfortunately, it is beyond this thesis to examine this literature in some detail.

[207] The application of concepts and theory gained for ecological systems to other types of systems, such as social or social-ecological systems can be explained by the system-theoretical comprehension of resilience research. As desribed in section 3.5.2.3, the resilience approach can be intepreted as a system-theoretical approach

objects can be highly useful as a communication tool in order to bridge scientific disciplines and the gap between science and policy (Eser 2002; Cash *et al.* 2003). Indeed, it is this vagueness and malleability, i.e. the potential variety of interpretations or applications of the term, that makes boundary objects politically successful (Eser 2002). For example, the boundary object “sustainability” has been highly successful in providing the common ground for ecologists and economists, which were formerly thought contrary, to engage together for the needs of future generations. In addition, the concept has helped to reconcile contrasting interests of industrial and developing countries.[208]

But there is a fundamental drawback to this. Boundary objects can in fact be a hindrance to scientific progress. For example, the meaning of the term “sustainability” is highly diluted and unclear. The three-pillar conception of sustainability (development in economic, social *and* ecological systems) has been reduced to a listing of any societal objectives that agents happen to think important. That means, the extension of the term has become extremely wide. This is due to the fact that the catchword “sustainable development” enables different scientific disciplines or social groups to justify *their* particular interest with respect to an accepted and ethically legitimated, societal goal (Grunwald 2004b). It may thus even hide conflicts and power relations when different persons agree on the need for sustainability when in fact meaning different things by it. Therefore, sustainability is generally conceived as arbitrary or as an illusion and within sustainability science there is confusion on how to operationalize and apply the concept (Grunwald 2004a). To foster conceptual clarity and practical relevance some authors have suggested a clear and specified theory of sustainability, which is characterized by both a *narrow extension* and a *clear intension* of the term (Ott & Döring 2004).

These insights indicate that, metaphorically spoken, boundary objects are Janus-faced, i.e. they are inherently ambivalent. They may have positive *and* negative aspects in terms of scientific progress and political success. What does that mean for a scientific concept of resilience?

[208] The concept of resilience can thus be regarded as “environmental innovation”, as it aids bridging scientific disciplines in innovative ways. It may complement other concepts within sustainability science, such as “biodiversity” and “ecosystem services” (Kurt Jax, *personal communication*).

4.1.5 Synthesis

In this section we synthesize the points made in the previous sections and discuss some implications for a fruitful conceptual structure of "resilience". Resilience is a two-faced concept. On the one hand, the concept is used as a *descriptive, ecological concept* (e.g. Walker 2002; Nyström 2006; Nyström *et al.* 2008), whereas on the other hand, it represents a *boundary object* with a rather wide and vague meaning (e.g. Gunderson & Holling 2002; Walker *et al.* 2006). As a result, the original, ecological concept of resilience first defined by Holling (1973) has been transformed considerably. This becomes apparent in several points.

First, *the specific meaning of resilience gets diluted and increasingly unclear*. This is due to the use of the concept (*a*) with *many different intensions* and (*b*) with *a very wide extension*. For example, Hughes *et al.* (2005) suggest several key components of resilience for marine regions. These include leadership and insight, sustained mobilization of national and international aid, cultural and ecological diversity, development of multi-scale social networks, and the resolution of local civil unrest. Apparently, Hughes *et al.* (2005) apply both the *social-ecological definition* and the *metaphoric definition* of resilience (cf. *class 8a + 9* in Table 1) in order to link an ecological-descriptive meaning of resilience to governance structures, economics and society. As a result, however, the concept of resilience includes very much (from international aid and leadership to ecological diversity) and it is for this reason why the meaning of resilience gets diluted and unclear, as for logical reasons any concept that encompasses very much (*wide extension*) must lose specific meaning (*clear intension*) (Ott 2003). Indeed, regarding the interpretation of resilience put forward by Hughes *et al.* (2005) it gets difficult to decide whether a certain state is resilient or not or to specify the particular degree of resilience inherent in a certain state.

Second, *a broad concept of resilience often includes normative dimensions*. Following the interpretation of Hughes *et al.* (2005) resilience represents a *hybrid concept* containing a blending of descriptive and normative aspects, as international aid, cultural diversity and the resolution of local civil unrest represent instrumental and eudaimonistic values.[209] The fact that a broad concept of resilience includes

[209] In a recent publication, the term „resilience" becomes entirely normative. Marschke and Berkes (2006) consider "well-being" as a substitute for the resilience of livelihoods in Cambodian fishing communities. I am very sceptical about this usage of resilience. If there is no difference between the two terms "resilience" and "well-being", the concept of resilience loses its meaning.

normative dimensions is not surprising. We see other boundary objects floating between descriptive and normative meanings, as in the case of "biodiversity" (biodiversity in the specific scientific sense of diversity at the level of genes, species and ecosystems *vs.* biodiversity in the sense of the ominous value of life on earth) (Eser 2002). But the important point is that these normative aspects within a broad concept of resilience ought to be made explicit and, wherever possible, justified ethically (U. Eser and T. Potthast, *personal communication*).

Third, *the term "resilience" is used ambiguously as divergent conceptions of resilience are proposed*. There are at least ten different approaches to "resilience". Each approach emphasizes different aspects of resilience with respect to the specific interest (the ecological aspect is stressed by ecologists, the political and institutional aspects are stressed by sociologists, etc.). Thus, the term resilience is used ambiguously for fundamentally different intensions (cf. *class 1 – 10* in Table 1). The direct consequence are trade-offs between social and environmental objectives *within* a conception of resilience, which may be difficult to handle.

Fourth, *the original ecological dimension of resilience is about to vanish*. Our impression is that recent studies increasingly stress the social, political and institutional dimensions of resilience or address whole social-ecological systems, while genuinely *ecological* studies of resilience get rare (but cf. Bellwood *et al.* 2004; Nyström *et al.* 2008).

Finally, *resilience is increasingly conceived as a perspective, rather than a clear and well-defined concept*. Recently, resilience has been conceived either as a way of thinking, as an approach to address social processes, such as social learning, leadership and adaptive governance (cf. *class 8b* in *Table 8*) or as a metaphor for the flexibility of a social-ecological system over the long term (cf. *class 9* in *Table 8*). According to Anderies *et al.* (2006) "resilience" is better described as a collection of ideas about how to interpret complex systems. As a result, the meaning of resilience gets increasingly vague and unspecified.

How to evaluate this conceptual development of resilience? We suggest on the one side that both *conceptual clarity* and *practical relevance* of "resilience" are critically at stake. A scientific concept of resilience must have a clear and specified meaning that is constantly used in the same way. In particular, it must be possible (*a*) to specify the particular objects the concept refers to, (*b*) to decide whether particular states in nature are resilient or non-resilient and it should be possible (*c*) to assess the degree of resilience of a certain state (cf. Grunwald 2004a, b; Jax 2006). In fact, the quality of the term "resilience" is strongly dependent on the ability to *exclude* phenomena that do not meet this term, as both operationalization and application with respect to environmental management are strongly dependent on a clear and delimited meaning of the term (Pickett *et al.* 1994: 57ff).

On the other side, however, we propose that the increased vagueness and malleability of "resilience" is highly valuable because it is for this reason that the concept is able to foster communication across disciplines and between science and practice (cf. Eser 2002). Therefore, it is not the suggestion to eradicate this vagueness and ambiguousness entirely but to *grasp the ambivalent character of boundary objects and, hence, of a wide and vague use of "resilience"*.

To counterbalance the positive and negative aspects of the conceptual development of resilience we, thus, argue for division of labor in a scientific sense. Resilience conceived as a *descriptive concept* should be a clear, well-defined and specified concept that provides the basis for operationalization and application within ecological science. For the sake of clarity this meaning may be dubbed *ecological resilience/ ecosystem resilience* (for ecological systems) or just *resilience* (if applied to systems other than ecological, e.g. climatic systems). In contrast, resilience conceived as a *boundary object* should be designed in a manner so as to foster interdisciplinary work. In this sense, resilience constitutes a vague and malleable concept that is used as an trans-disciplinary approach to analyze social-ecological systems. For greater clarity this meaning may be termed *social-ecological resilience* (as in Folke 2006).

4.2 Adaptive Management, Cultural Theory and Rationalist Cosmology: Uncovering Connections

This section points to some of the cultural presumptions of resilience research. It is the result of the collaboration with Dr. Thomas Kirchhoff and Deborah Hoheisel, both working at the Technische Universität München. Some of the results of this section will be published in the journal *Ecology & Society*.

After a short introduction in section 4.2.1, section 4.2.2 justifies the methodology of our analysis within the philosophy of science, which is mainly based on the doctoral thesis of Thomas Kirchhoff (Kirchhoff 2007). Subsequently, we analyze the cultural presumptions of resilience research regarding its systems notion in section 4.2.3 and its theory of the man-nature relationship in section 4.2.4. Section 4.2.5 concludes with the findings of this section.

4.2.1 Introduction

Regarding bibliometric statistics, the concept of resilience ranges among the rising stars within sustainability science (Janssen 2007). The term has been introduced in the 1970s as a stability concept for ecological systems (Holling 1973). Resilience research views “nature” as an interacting hierarchy of complex, adaptive and self-organizing ecosystems that can exhibit alternative stable states (Holling 1986). Recently, this systems-theoretical approach has been extended to social, political, institutional and coupled social-ecological systems (e.g. Gunderson & Holling 2002; Folke 2006). Based on this perception of complex systems, the “resilience approach” champions a particular conception of environmental management focussing on social-ecological systems. This “adaptive management approach” is being proposed as the necessary and innovative way to treat or interact with “nature”, as this approach alone would take into account the development principles of ecosystem dynamics and would thus assure a sustainable use of natural assets (Holling & Meffe 1996; Olsson *et al.* 2004). Hereby, resilience research presupposes a particular notion of ecological systems and a specific theory of the man-nature relationship demanding universal validity.

This demand is being investigated in this section. We show that both the systems notion and the theory of the man-nature relationship advocated by the “resilience approach” are one-sided, in that they are based on particular cultural presumptions that lead to the masking of alternative, well-founded approaches from the very beginning. Thus, we challenge the universal validity of the resilience approach, i.e. the claim that “nature” and the man-nature relationship must necessarily be conceived according to the resilience approach. By contrast, we suggest that there are alternative, reasonable approaches, that may also contribute to sustainable forms of resource use and environmental management.

This one-sidedness of the resilience approach is being elucidated *not* by showing that there are ecological or social-ecological systems that are constituted in a different way than the resilience approach holds they are. Generally speaking, we do *not* aim at finding empirical arguments in favor or against the resilience approach. To uncover its partiality we take an entirely different path. As a “secondary-order observer” (Luhmann 1992: 68ff), we ask how the mode of empirical first-order observation characteristic for the resilience approach emerges. We thus aim at contributing to explain the accomplishment of the systems notion and the theory of the man-nature relationship proposed within resilience research and not to evaluate it by confronting it with the results of empirical work.

More precisely, based on anti-positivistic theories of science (e.g. Popper, Foucault, Kuhn, Lakatos), we delineate the philosophical presumptions of the resilience approach by means of an analysis within the history of ideas. We show that the resilience approach is in its essence structurally analogue to (*a*) a particular type of holistic cosmology, that we characterize by the example of Leibniz' monadology and to (*b*) a particular counter-enlightenment theory of the man-nature relationship that we specify by recurring to Herder. The aim of this section is thus to uncover the cultural basis and bias of the resilience approach and of the adaptive management approach for explicating its theoretical partiality and limitations. The claimed objective necessity (i.e. imperative) of adaptive management turns out to be a subjective tenet (i.e. maxim), whose validity is dependent on cultural ideas which are not universally valid.

4.2.2 Justification of the Analysis' Method

In this section we challenge the universal validity claimed by the resilience approach and the thereof based adaptive management approach. Hereby, we do *not* empirically evaluate theses approaches, rather we show that they are based on and determined by cultural presumptions that (*a*) cause a one-sidedness in empirical examination and (*b*) mask alternative, well-founded approaches from the beginning. To justify this thesis we try to uncover isomorphies, i.e. structural analogies even though the subject matter differs (Gould 1990), of the resilience approach on the one side and specific cultural and philosophical ideas on the other side.

This procedure may rise the following questions: (i) why does it make sense at all to examine the cultural presumptions of a scientifically based approach to environmental management?; (ii) why is it adequate to treat these isomorphies as an indication for a culturally determined one-sidedness? By answering these questions, we unfold and justify the methodology of our analysis in the following.

Put simply, our reply to both questions refers to the following thesis: any "reality" is - even if the "objective reality" of natural science is concerned - partly (but not exclusively) the result of a cultural construction of nature. For biology, the starting point of this thesis are the empirical facts that (i) the history of biology is characterized by long-standing controversies, which apparently cannot be resolved in favor of a sole theory by falsifying all alternative theories; (ii) such controversies can be highlighted in almost every biological subdiscipline; and (iii) these disputes show considerable parallels. Indeed, for a long time rather individualistic and rather organismic theories compete not only within community ecology (Trepl 1987; Jax 2002, 2006) but also in population ecology (Schwerdtfeger 1979; den Boer & Reddingius 1996), genetics (de Winter 1997; Skipper 2002) and evolutionary theory (Brandon & Burian 1984; Stegmann 2005) (for a comprehensive review cf. Mayr 1082; Looijen 1998; Kirchhoff 2007).

These historical facts can be at first explained by a bundle of diverging philosophies of science that altogether doubt that scientific theories are ever decisively refuted (Quine 1951; Lakatos 1970; Hacking 1992). A further explanation is provided by particular theories of science, which argue against empiristic theories (Nagel 1961) and thus against ontological realism. According to these anti-empiristic theories any scientific theory is being formulated within a theoretical framework that includes several characteristics. Those are: (i) the framework consists of conventional presumptions about the general constitution of the study objects that are not empirically justified; (ii) the framework determines what kind of questions are (not) being posed and in what form they should (not) be answered; and (iii) the framework is being protected against falsification. This theoretical framework has been called “metaphysical presumptions” of a “*disciplinary matrix” (Kuhn 1970), “hard core of research programs” (Lakatos 1970) or “styles of scientific thinking” (Crombie 1994). Following these scholars, non-empirical presumptions are a non-eliminable component of science. That means, more or less, that the context of justification of a scientific theory is dependent on its context of discovery. Instead of pointing to (alleged) misapprehensions in the perception of a metaphysical-ontologically interpreted reality (Bacon, Popper), any concept of reality - complementary to its empirical evaluation - is thus to be investigated for its (philosophical) presumptions (Hacking 1999; Demeritt 2002).

These non-empirical presumptions do not primarily represent peculiarities of an individual researcher, but intersubjective and cultural interpretative models coined “historical a priori” (Foucault 1970/2004) and “cultural predispositions” (Groh & Groh 1996) within the philosophy of science. In the end, these interpretative frameworks are based on perceptions of humans regarding herself/himself and the relation to other humans. As Merchant (1990: 69) notes: „A view of nature can be seen as a projection of human perceptions of self and society onto the cosmos“. The overall message of this is that competing cultural, interpretative models determine competing scientific concepts of nature (Röd 1991; Lenk 1995; Wu & Loucks 1995; Hacking 1999). Regarding community ecology and specific approaches to environmental management (such as the resilience approach), an anti-empiristic theory of science has an important implication: competing theories of ecological units mirror competing cultural ideas on the relation of “the part” and “the whole”, that are based on competing ideas of man, concepts of individuality and theories of human societies (Trepl 1993; Kirchhoff 2007) (cf. section 4.2.3).

We aim at uncovering the specific cultural ideas that underlie the systems notion and the theory of the man-nature relationship proposed within resilience research (perspective of a “secondary-order observer”). We illustrate this link within the history of ideas by showing isomorphies of the resilience approach and the adaptive management approach on the one side and particular influential, cultural ideas on the other side. These cultural ideas are delineated with reference to the cosmology of Leibniz (section 4.2.3) and the cultural theory of Herder (section 4.2.4) as these authors have expressed the interpretative models relevant for this article with a particular resoluteness. Note that the cultural ideas could be found in the work of numerous other authors, yet these ideas were efficacious as a form of *Zeitgeist* even if they would have never been formulated as a philosophical theory. It is therefore inadequate to object against our procedure that the proponents of the resilience approach have never read anything of Leibniz or Herder (for the longevity of ideas cf. Lovejoy 1936; Blumenberg 1966; Berlin 1979).

To be sure: yes, we point to the determinating effect of cultural ideas on the theory development in ecology and environmental management, but this does not mean that we deny that nature *per se* restricts the possibilities to culturally construct nature. Yet the proposition that the existence of humans has natural bounds and that concepts of nature cannot be in conflict with these existential conditions, is in our view trivial. What we want to show is rather that any objectivity of “reality” is partially determined by culture, more precisely, that this determination is always two-fold, i.e. by “nature” *and* by “society”.

4.2.3 Theory of Ecological Systems

In section 4.2.2 we have offered the view that theories of ecological units are based on cultural ideas of individuality and human societies, respectively. In this section we apply this insight to the theory of ecological systems represented by the resilience approach. First, we describe this theory and make clear that it belongs to the organismic paradigm. Then we show that organismic theories are controversial and opposed by so-called individualistic theories. Finally, we outline the opposing cultural ideas which produce those conflicting ecological theories. We portray these ecological theories and the underlying cultural ideas only insofar as the structural analogies between them become evident.

4.2.3.1 Resilience Approach

The theory of ecological systems in the resilience approach includes a particular concept of (i) the organisation of ecosystems, (ii) their development, and (iii) their functional connections between each other and with systems of other hierarchical levels. In the following, we describe this concept of ecological systems by analyzing some key publications of the resilience approach, such as Holling (1986), Holling (2001), Holling and Gunderson (2002) and Folke (2006).[210]

(i) Ecosystems are defined as "communities of organisms in which internal interactions between the organisms determine behavior more than do external biological events" (Holling 1986: 297). Similar to Odum (1971), ecosystems are conceived as distinct subsystems of the earthly circular biogeochemical system. These ecosystems mediate external abiotic influences by strong internal biological interactions, cybernetically regulate the conditions for life, and insofar are "Gaia writ small" (Holling 1986). Also, ecosystems are understood as ontologically real units, because their components - like the organs of a single organism - are considered as being connected to one another by contributing to the self-preservation of the system. As Cumming and Collier (2005: 4) note: "being a component must be understood in terms of having a relevant role in the overall functioning of the system, not just being there as a constituent of the system".

[210] Cf. also section 3.2.1.

(ii) Regarding ecosystem development, the resilience approach at first follows again Odum's view: by a succession of definite stages a system of *r*-strategists, which opportunistically exploite ressources ("*exploitation function*"), gradually develops into a consolidated system of *K*-strategists ("*conservation function*") - in the course of which each the degree of organisation, the biomass of the system as well as the connectedness of its components increases. But the proponents of the resilience approach modify this classical theory of succession by refuting the mechanistic-deterministic interpretation of succession: Succession is a discontinuous, non-linear, 'organic' process; it depends on the autonomous, internal reactions of the ecosystem on the always changing and specific environmental conditions. Consequently, succession does not always lead to the same climax, as multiple outcomes typically are possible depending on accidents of history. An ecosystem if disturbed thus needs not to return to the original state.

The resilience approach argues that the classical equilibrium-centered view (Clements 1916; Patten 1975) is a misconception of nature (Holling 1986; Folke 2006). The equilibrium-centered view of nature, on which the concept of 'engineering resilience' is based, does not take into account that the self-organisation of ecosystems is guided by alternative "basins of attraction" so that nature is a "multi-stable state reality" (Folke 2006).[211] Consequently, in the resilience approach 'resilience' is defined as the ability of a system to absorb disturbances without leaving its basin of attraction.

Moreover, the resilience approach modifies the classical theory of succession by assuming two additional stages of succession. Corresponding to the degree of connectedness, during the succession resilience increases for some time, but then descreases. Sometime or other but necessarily, this loss of resilience leads to "*creative destruction*" and "*ecosystem renewal*": By the sudden destruction of many organisms, accumulated ressources are released, yet the system might be able to minimize their loss and to use them for its renewal. Thus, the ecosystems - by means of their four functions exploitation, conservation, creative destruction, and renewal - go through a cyclic development, termed the "adaptive cycle". Owing to the occurence of this "renewal cycle", ecosystems are able to adapt to permanently changing environmental conditions. Disturbances at a small spatial and temporal scale are regarded as a (necessary) part of this cycle and as a prerequisite for

[211] Cf. section 3.2.3.

resilience, for they initiate the adaptive renewal cycle before the systems organization has become too rigid for renewal and the need for adaptation to the changing environment has become too great.

(iii) The resilience approach assumes a nested hierarchy of ecological systems, reaching from the organs of a single organism to the whole biosphere. On the one hand, ecosystems are distinct systems, whereas on the other hand they are functional subsystems of the comprising system of the earth, the “panarchy”. This hierarchical organisation is regarded as a precondition for resilience, because it enables the system to maintain and adapt, i.e. to maintain continuity while changing (Peterson 2000; Holling 2001).

Altogether it has become clear that the ecosystem theory of the resilience approach belongs to the organismic paradigm, as it ascribes many properties to ecosystems that usually are regarded as being characteristics of (single) organisms. These characteristics are - shown in Köchy’s (2003) extensive analysis of organism concepts - (i) unity, individuality, and purposiveness (concerning the organism as a whole), (ii) interaction, hierarchy, process, and spontaneity (concerning the relations among its parts), and (iii) evolution, reproduction, and freedom (concerning the relation of an organism to its environment). Similarly, the resilience approach views ecosystems as (i) quite distinct unities, which (ii) organize and maintain themselves by the hierarchical interactions among its components, in which (iii) the organisation of the ecosystem varies according to the environmental conditions, without being determined by them. By assigning the resilience approach to the organismic paradigm we do not insinuate that it regards ecosystems as (super-)organisms (Wu & Loucks 1995).

4.2.3.2 Alternative Approaches

Since the beginning of ecology it is controversial (i) which are the causes of species composition and of correlations between the distribution of species respectively, and (ii) which ontological status can be ascribed to ecological units distinguished by ecologists. Put simply, there is a controversy between organismic and individualistic theories (cf. Trepl 1987; Jax 2002, 2006; Kirchhoff 2007; Voigt 2008).

Even with respect to ecosystems, organismic theories have been and still are disputed. For instance Engelberg and Boyarsky (1979) have argued against Odum’s organismic theory (Odum 1971), upon which the resilience approach is based (see above): Ecosystems are no cybernetic, self-organizing systems, because their components are not connected with one another by information-based feedback. Radically individualistic critics of organismic theories argue that the properties and the distribution of a species are not strongly affected by the presence or absence of other species. It is for this reason that species composition normally is continually shifting in time and space (Gleason 1926; Hengeveld & Walter 1999). In less radical individualistic theories, it is assumed that - besides immigration and abiotic environmental factors -interspecific interactions determine which species can coexist; but it is disputed whether or not these interactions thightly bound certain species to one another and so produce communites with a fairly fixed composition (Peus 1954; McIntosh 1995; Hubbell 2001).

4.2.3.3 Notes on the Cultural Background

In the third section we have demonstrated on the basis of theories of scientific knowledge that the opposing positions within the controversy about the character of ecological units result from opposing concepts of individuality proposed during our occidental cultural history. In this section we outline these concepts, which originally have been developed in answer to a theological-philosophical problem, not to a natural-scientific one. It is possible to discover these ideas in many western philosophies. Our cases in point are the nominalistic cosmology of William of Occam and the rationalistic monadology of Gottfried Wilhelm Leibniz. Their philosophies represent the opposing concepts with a particular resoluteness: Occam develops the individualistic concept of individuality, Leibniz the holistic or organismic concept of individuality (Lukes 1973), and their philosophies have had far-reaching influence (see below). We are not insinuating that they have had a *direct* influence on ecology.

Occam: The world as a fortuitous plurality of unique individuals

What is the nature of individuality and universality? For Occam and Leibniz - as for centuries in the entire Christian philosophy - the answer to this question depends on the answer to the following issue: what is the relation between god’s reason and god’s will?

Occam interprets god’s will as an absolute power. It is bounded by nothing, not even by his own reason (but by the principle of contradiction). Consequently, it must not be assumed that god has created the world as an (universal) order in which every creature occupies a certain place. It must not even be supposed that he, in creating single creatures, has been guided or bounded by universal ideas. Rather, it needs to be assumed that he has created every creature as an individual with a unique nature and regardless of which other creatures he had already created or would still create. Therefore, every creature is a *res absoluta* with regard to its properties and reason for being; the world is nothing but a plurality of individuals that is fortuitous with regard to extension and composition. As every individual represents nothing but itself, man, to assert himself, has to observe single facts and then (resting on accidental similarities among them) has to construct an order which is useful for the particular purpose (*nominalism*) (Blumenberg 1966).

Occams cosmology and epistemology are usually regarded as the basis of modern empirism. This in turn is tightly connected with political and economical liberalism, whose basic presumptions are the ideas of every man’s freewill, singularity, and equal opportunity for pursuing egoistic interests. Such cultural ideas have contributed, e.g. to the rise of Darwins theory of evolution (Ghiselin 1969; Schweber 1977; Gould 1990). The most important structural analogies between Occams philosophy and the ecological individualism is that they both regard the “individual”, i.e. the single creature and the single species, respectively, as being independent from one another with regard to their properties and occurrence in time and space. In addition, both regard the “universal”, i.e. universal concepts of order and ecological communities, respectively, to be nothing but useful constructions of the mind.

Leibniz: The world as a harmonious system of unique individuals

In contrast to Occam, Leibniz presupposes that god’s will and reason form a unity. Consequently, god’s creation has to be a reasonable, universal order. Yet correspondingly to Occam, Leibniz presupposes that god has wanted every creature in its uniqueness. Leibniz reconciles both presumptions by the idea that the universal *order*, which interlinks all creatures, is just based on the *uniqueness* of every creature, by which it is seperated from the others. In other words: the delimited, original, unique individuality of every creature or monad consists in nothing but its unique relations to all the other monads. The individuality of every monad is holistic because it is refering to the entirety of monads constituting the world; and the world is an organic system insofar as each monad is possible only due to the existence of all the other monads. This proposition by Leibniz has been a new and influential concept of individuality and universality within the history of ideas. Leibniz has elaborated this concept as a theory of perception (which is not a theory of causal relations) pursuant to which the world is a multistage nested hierarchy (Gurwitsch 1974; Holz 1992; Köchy 2004).

The monadology has been an important intellectual basis of the modern biological concept of the organism and of the cultural theory of Herder, which identifies as the goal of history that a diversity of unique, organic unities of a people or nation and its country is developed (see section 4.2.4). Herder's cultural theory, in turn, has given rise to the classical, likewise more or less superorganicistic cultural geography (Glacken 1973). This cultural geography (which has principally been developed by C. Ritter), has found monopolistic spread in the German-speaking area and has been absorbed everywhere in Europe (Martin 2005). In the US, this approach has been further developed by geographers like C.O. Sauer and R. Hartshorne (Duncan 1980; Olwig 1996; Martin 2005). These geographical theories, in turn, have been an important intellectual basis for ecological organicism in which unities of community and habitat replace those of people and country (Trepl 1997).

The most important structural analogy between Leibniz' philosophy and the ecological organicism described above is that both assume the existence of a nested hierarchy of systems whose components are interdependent in their existence and properties.

4.2.4 Ecosystem Management

In the previous section, we analyzed the resilience approach with regard to its perception of ecological systems. The results show that the resilience approach is one-sided, because it follows a specific cultural concept of individuality. Now we turn towards the practice of the resilience approach, the adaptive management approach. We will first characterize it, then point to opposing conventional land-use approaches and finally uncover the cultural premises of these opposing approaches. The objective is - as in the previous section - to show the one-sidedness of the resilience approach and to discuss in what form implicit and in most cases probably unconscious cultural presumptions result in biased recommendations for the practice of environmental management.

4.2.4.1 Adaptive Management

The following characterization of the adaptive management approach is done with the aim to later compare it with the man-nature-relationship that Herder proposed in his cultural theory. The characterization therefore does not lay claim to completeness, but picks out specific aspects of the theory of the man-nature-realtionship that is part of the adaptive management approach.[212]

This approach is based on the assumption of a close relationship between ecological and social systems. It is assumed that ecological, social and social-ecological systems can be described and analyzed with the same conceptual, theoretical, and modeling framework (Walker *et al.* 2006; Holling & Gunderson 2002), even though humans are told to have a special role in social-ecological systems (e.g. Walker *et al.* 2006). Furthermore, it is assumed that social-ecological systems are more than just a combination of social and ecological systems. Berkes *et al.* (2003b) even consider the separation of ecological and social systems in the end as artificial and arbitrary. Social-ecological systems are thought to be coevolutionary units comprised of social and ecological subsystems (Liu *et al.* 2007). This most probably means, that in the course of time through a mutual influence the inner structure of the social system is formed by the specific ecosystems of the environment while these ecosystems are

[212] For a description of the adaptive management approach cf. section 3.5.2.

formed by the social system (Norgaard 1988; Folke *et al.* 2003). It is assumed that on the one side concrete rules - namely the specific requirements for resilience - are stipulated by the natural ecosystems and have to be taken into account by humans; otherwise catastrophes will occur (Folke *et al.* 1998). On the other side humans have the possibility to change ecosystems - within the limits that are set by these rules - to reach a resilient state that is thought to be especially desirable or useful (RA 2007). Thus, humans are bound to the fact that ecological or social-ecological systems, respectively, can exhibit specific basins of attraction. Hereby, the resilience of the system increases towards the centre of these basins. It is possible to hold a system artificially at the edge of a basin of attraction, but the expenditure is high. Furthermore, the system is then highly prone to external disturbances, i.e. the risk of a shift to an undesirable state located within another basin of attraction is high. Such a behaviour is not efficient or "sustainable" and thus unreasonable.

In this context, the resilience approach points to the knowledge of traditional, pre-modern cultures (termed *traditional knowledge* or *local knowledge*). It is claimed that in contrary to the conventional resource management put forward by the industrial countries, many of these cultures have drawn the rules of a succesful adaptive management directly from nature. It is assumed that each culture has developed unique means to adapt itself to the specific natural conditions. By the active use of small disturbances that prevent a big crash and allow an ongoing renewal and adaptation of the system, the release phase of the adaptive cycle has been taken into account by those cultures (Berkes & Folke 1998).

The adaptive management approach stands in opposition to a dualistic distinction of "nature" and "society" and combines the idea of the bond of society to concrete nature (in the form of regional specific self-organising ecosystems) with the idea of the freedom of humans (in the form of utilitarian possibilities to change these ecosystems) in a theory of coevolutinary man-nature-units. Moreover, it combines the idea of a bond to concrete nature with the idea that an ongoing development must be possible for this nature.

4.2.4.2 Conventional Resource Management

The adaptive management approach is directed against approaches that we summarize here with the term "conventional resource management". This management tries to change the unpredictable and for human beings inefficient natural systems into predictable ones with a maximum and stable production of the desired product (Holling 1995). It is (i) based on a mechanistic and deterministic view of nature insofar as it is supposed that the manipulations of abiotic conditions and the control of biotic factors have predictable, i.e. more or less linear consequences. It is (ii) based on the paradigm of sustained yield, which holds that "renewable natural resources can produce continously under a given intensity of management" (Anderson 1995). It is (iii) the expression of a constructivist ideal insofar as it is considered to be possible to optimize existing ecosystems by manipulating them or even to built artificial ecosystems (cf. Holling & Meffe 1996). Behind all this lies the assumption that ecosystems are *not* organic wholes, i.e, not systems that 'react' on changes (through humans) to maintain themselves in a certain state.

In this approach "nature" is a nature that is infinitely malleable and amenable to human control and domination if only the right values and the right timing are chosen (Holling *et al.* 2002). Accordingly, the man-nature-relationship in conventional resource management is not viewed as a coevolutionary one, but as a controlling and subdueing one. Social and ecological systems are seen as separated, as uncoupled systems (cf. Folke *et al.* 2002). Humans are not a part of the (social-ecological) system, but stand outside it and try to control "nature".

4.2.4.3 Notes on the Cultural Background

Now we turn to the cultural basis of the adaptive management and the conventional ressource management approach - with the focal point on the former.

The enlightened ideas of the individual and of emancipation as the cultural basis of conventional resource management

The common ground of the different enlightenment theories is the assumption of the free will of each human beiing and of the existence of a common and universal reason of mankind. With the help of this reason mankind should and can create a social order that secures (a maximum of) freedom and equal rights for every individual. This order has a conventional character, because it is established by a social contract or just through habits and actions and it has a universal character (cosmopolitanism), because it is based on universal principles and intentionally independet from adaptation to nature. Based on the research into the *general* laws of nature the development of technology should make material progress and the emancipation of mankind from the natural conditions at a specific place on earth possible.

The idea of the emancipation from concrete nature is based on the assumption that the use of biological ressources and processes is not bound to the continued existence of the concrete ecological systems that have developed in the different regions on earth in the course of the natural history, but that useful ecosystems (in principle in the same way as machines and human societies) can be constructed in almost any way depending on the objective. This again is based on the assumption of an (essentially) individualistic interpretation of ecological systems. The cultural basis of this interpretation - and of liberalism (but not of all forms of enlightenment) - was described in section 4.2.3 by referring to the example of Occam's cosmology. It is obvious that the enlightened ideals of individual freedom and emancipation are the basis of the conventional ressource management that is critized by the resilience approach.

Herder's theory of organic people-country-units as cultural basis of the adaptive management approach

We will now show that the man-nature-relationship stated in the adaptive management approach is in its essential aspects equivalent to the one demanded by the counter-enlightenment. We characterize this view by the example of the cultural theory of Johann Gottfried Herder, who is said to be the founder of modern cultural geography (cf. section 4.2.3).

Herder assumes - refering to Leibniz' monadology - that nature is a self-developing organic totality, in which the same way of organisation and the same principle of development is characteristic for all living creatures: on the one hand, each creature is for itself an integral whole with an organic force that leads to the development of a characteristic form, which reflects the characteristic conditions of its environment; on the other hand, the development of each creature reflects the developments of the other creatures, it is dependent on them, i.e., each creature is part of a superordinated integral whole, likewise reflecting the characteristic conditions of its environment (Herder 1976 (1877-1913)); SW XIII, XVIII: 308). The same relationship between individual, superordinated individual and environment exists, though in a rational way, between human individuals and their cultural community (people, nation) (XVIII: 308 f.), between a people and its country ("Land", "Erdstrich") (see below) as well as between different cultures (XIII: 270, 345 f., 352). The history of culture or mankind respectively has to be the rational continuation of the history of nature, if it shall succeed (XIII: 163, XIV: 226). The development of a culture has to be a process with a two-fold determination in which (i) the people adapts itself (like an organism in the relationship with its environment) to the specific conditions of its country (Herder summarizes these with the term "climate"), (ii) the climate forms the sensuality and way of thinking of the people, i.e. its character, and (iii) the people shapes its country according to its original, environmentally modified, and specific character (VIII: 210, XIII, XIV: 38). That is, the people *detaches itself* from the direct constraints of the nature of his country just by *adapting* itself to it. So, cultural development leads to organic units of country and people each having an unique character (V: 505, XIV: 83-86, 227). For the succes of this development it is crucial that, on the one hand, the specific cultural traditions are considered, because they contain the knowledge about the possibilities of adaptation to the country, but, on the other hand, each generation does not follow the traditions blindly, but develops them further according to their own needs and talents (VI: 250, XIII: 347 f.). For the organisation of the state follows from this conception that it must neither be despotic-arbitrary (absolutism) nor secure equal opportunities in the pursuit of egoistic interests (nominalistic liberalism) nor follow alleged universal principles (rationalistic enlightenment); the state should rather enable and encourage the individuals to freely develop their different nature-given talents in the service of the community (XVII: 122, XVIII: 309). In this way, and only in this way, every people and each individual can realise its unique talents, i.e. perfect itself (and its country), i.e. be free.

With this cultural theory Herder opposes, like the enlightenment, in the name of individual freedom against despotism and fixed traditions. At the same time he contradicts essential ideas of the enlightenment: The history of mankind is not a linear progress, which - with the means of ahistoric universal principles of reason - finally leads to the same state all over the world; rather it is an organic process of growing, which - with the means of tradition and historic and specific reason respectively - leads to unique perfection in which a natural cycle of growing, prosperity and dying of cultures is necessary for the development of new cultures (for the entire interpretation cf. Gadamer 1942; Vierhaus 1973; Eisel 1991; Kirchhoff 2007: 489ff).

Now we can make explicit the isomorphies between the adaptive management approach and Herder's cultural theory: (i) Both assume that nature and society are in principle based on the same way of organisation. (ii) Both critize a dualistic distinction of "nature" and "society" or "nature" and "reason" respectively; both combine the idea of the bond of society to conrete nature - according to Herder the specific country or climate, according to the resilience approach the local and regional ecosystems - with the idea of the freedom of man to form and change his country or the ecosystems respectively. (iii) In both theories social and ecological systems are not only thought as interacting, but as coevolutionary and so forming a new, superordinated unit. (iv) According to Herder traditions are important, nevertheless they should not be followed strictly, but be developed further without loosing their origin. In a similar way, the resilience approach connects the idea of the bond of society to an evolved, natural ecosystem with the idea that this system or the corresponding social-ecological system respectively have to develop continuously. (v) In both theories development is a process that combines cyclic repetition (constancy) and continous development (change). (vi) Both theories assume that (succeeding) development is neither chaotic nor mechanistic, but organic and leads to discrete states: i.e. cultural uniqueness and systems with alternative bassins of attraction, respectively.

Beside these isomorphies some differences exist. At least one is to be mentioned here. Herder's theory contains the idea of (individual) perfection and thus a teleological element: Every culture and every man is to realise its unique predetermined possibilities and is to improve the divine creation through the cultivation of its country. Whereas in the adaptive management approach it is - in a purely utilitarian way - only important that the ecological systems keep their identity during the cultural development, no matter in which direction the socio-ecological systems develop. That is, the idea of personal and cultural uniqueness which can develop thanks to the particularities of concrete nature is transformed into the idea of functional constraints to humans by the principles of self-organization of ecological systems. This shift can be interpretated as follows: Herder's theory is naturalistic insofar as it bounds culture to nature, but it is theological insofar as it interprets the particuliarities of concret nature as divine means to encourage human self-perfection. If concrete nature can or shall no longer be regarded in this way and yet society shall be regarded as tied to it, then in concrete nature itself (i.e. without final reference to its transcendent creator) has to be proved an objective principle that imposes a restriction on social systems. In the resilience approach, as this principle functions the supposed organic organization of ecosystems in combination with the supposed existence of discontinuous alternative basins of attraction (cf. section 4.2.3.1). If and only if species are the result of coevolution and species (or at least groups of species) are mutually depending on each other in their existence (organicism), can ecosystems be thought to be (i) real, observer-independent systems to which (ii) humans have to adapt, if they want to use biotic and biogene ressources in a sustainable way (Eisel 1989; Trepl 1993). If ecosystems were only constructions (as usually assumed in the opposing conventional resource management), whether real or mental, humans would only have to consider the requirements for existence of the *single* species, if they use natural ecosystems or construct them in reality according to their goals.

Despite the described difference it can be summarized that the man-nature relationship that the adaptive management approach strives for and the one that is demanded by Herder in his cultural theory are in essential points isomorphic. In the resilience approach the cultural and political idea of the counter-enlightenment that society is bonded to concrete nature and thus the ideal of regional uniqueness is reformulated in a utilitarian way with a scientific, ecological argumentation by determining concrete nature as consisting of organic systems to which social systems are bounded.

4.2.5 Conclusion

The analysis of the systems notion and the theory of the man-nature-relationship proposed within resilience research shows two fundamental isomorphies with cultural theories. These cultural theories have been developed a long time ago within an entirely different context, such as theories of individuality, of society and cultural development. (i) The resilience approach champions a (non-teleological) organicism within community ecology, as ecosystems are specified by characteristics that are also valid for single organisms. This organicism is isomorph to Leibniz' monadology (even though the main difference is that organicism proposes the idea of a self-organising nature instead of the monadological idea of God's plan). (ii) The adaptive management approach assumes that the maintenance of the self-organization of ecosystems is a prerequisite for the sustainable use of environmental resources. The resilience approach thus advocates a man-nature-relationship that is in its essence structurally analogue to the cultural theory of Herder, which is based on Leibniz' monadology and has been formulated as a counter-reaction to the enlightenment ideal of the emancipation and domination of nature (even though the most important difference is that a theological theory based on the idea of uniqueness is replaced by an utilitarian-ecological argumentation suggested in resilience research). The resilience approach can thus be subsumed under the so-called "regionalism". These globalisation-opposed theories are developed under late-modern living conditions for pre-modern interpretations of reality (Werlen 1997: 296), as culture is viewed as essentially dependent on and determined by specific regional living conditions.

By uncovering these isomorphies we challenge the universal validity of the resilience approach, i.e. the claim that "nature" and the man-nature relationship is to be conceived as proposed within resilience research. The unfolded isomorphies are not a proof but a strong indication that this approach and the adaptive management approach are determined by presupposed cultural ideas. We thus propose that the resilience approach should be viewed as a culturally determined way of percepting and dealing with nature. It is one-sided, because cultural presumptions lead to the masking of alternative, well-founded approaches, e.g. the individualistic systems notion within community ecology. The adaptive management approach should not be viewed as an indispensable means for achieving sustainable land-use practices and thus as an universally valid, practical necessity (hypothetic imperative *sensu* Kant), but rather as a subjective tenet (maxim *sensu* Kant). Regarding the adaptive management approach, the problematic thing is that it introduces (in a tendentious naturalistic way) societal ties to nature as objective necessities, even though these ties - at recent ecological knowledge - can only be legitimized by the societal weighing of subjective tenets, and thus also of interests and risks. Whether it is appropriate to propose these necessities for strategic reasons in order to promote nature conservation measures, is another question.

To be sure: the cultural determination of scientific theories, which we uncovered for the resilience approach, is valid for alternative approaches as well, e.g. an individualistic approach in community ecology or the conventional approach to resource management. Our analysis of the cultural bias of the resilience approach is no pleading for the unrestricted exploitation of nature. Rather, what we propose is that one should in principal be sceptical if any necessity in the societal relations to nature is justified ecologically on the ground of the supposed objective constitution of nature *and* if this constitution is based on a particular theory in community ecology instead of referring to similarities following from competing theories.

Regarding the discussion about an adequate approach to environmental management, we suggest to examine more profoundly - complementary to the empirical evaluations of competing approaches - in how far non-explicated and widely unconscious cultural presumptions result in the partial interpretation of data and inadequate practical recommendations. What is required are thus the widely proposed transdisciplinary studies of social-ecological systems. Yet these studies should *not* be based on the partial paradigm of the *ontological* unity of nature and culture, as proposed in resilience research and similarly in landscape ecology (e.g. Naveh & Fröhlich 1996; Tress & Tress 2001).

5 Ecological Resilience & Sustainable Development

The third fundamental objective of this thesis is to investigate the relation of the concept of ecological resilience to a well-founded conception of sustainability. In my view, pivotal sustainability problems can be used as a touchstone for the practical relevance of the concept of ecological resilience as regards to sustainability.

As the extension of the topic of sustainable development is very wide, it is impossible to treat the whole relation of the concept of ecological resilience to sustainable development in this chapter. Rather, I focus on two central topics: the concept of critical natural capital and poverty reduction measures in developing countries. Why is the focus on these concepts? First, as I champion a conception of strong or at least intermediate sustainability (cf. section 5.1), I consider the maintenance of natural capital as an important objective of sustainable development (for proponents of "weak sustainability" this is not necessarily the case). In the face of current population growth and global change, the concept of *critical* natural capital becomes especially relevant, as it signifies that part of the natural capital that we are obliged to preserve for present and future generations in any case. This is the reason why I focus on the relation of the concept of ecological resilience and critical natural capital. Second, to my knowledge the eradication of poverty is a fundamental objective of *all* sustainability conceptions, as sustainability refers to the basic idea that present and future generations have the same right to find, on the average, equal opportunities for realising their concepts of a good human life (Ott 2003). Thus, the reduction of poverty in developing countries is a salient topic for illustrating the practical relevance of the concept of ecological resilience.

In the following, at first I clarify the concept of sustainable development (section 5.1), before I examine the relation of the concept of ecological resilience to the concept of critical natural capital (section 5.2) and poverty eradication (section 5.3).

5.1 Sustainable Development – What is It?

This section specifes what I mean by “sustainable development”. It is absolutely essential to specify the specific conception of sustainable development, as the term “sustainability” is interpreted highly ambiguously in the national and international scientific discourse as well as in the general public. I have delineated my notion of sustainability in a recent article that I wrote together with a colleague from the university of Greifswald, Julia Schultz, as well as with Jürgen Kopfmüller, ITAS Karlsruhe and Konrad Ott, university of Greifswald. This section represents a revised version of the article that is published in the *International Journal of Environment and Sustainable Development* (Schultz *et al.* 2008).

5.1.1 Introduction

Sustainable development is recognized the world over as a key challenge facing 21st century society (UNEP 2002; WBGU 2005; Komiyama & Takeuchi 2006; Clark 2007). Since the Brundlandt-report (WCED 1987) and subsequent international conferences in Rio de Janeiro and Johannesburg the idea of sustainability exhibits broad political appeal but has proven difficult to define in precise terms (Parris & Kates 2003b; Kates *et al.* 2005). The extraordinarily broad list of items to be sustained and to be developed, as identified in the relevant publications, indicates that the extension of 'sustainable development' has become extremely wide (Parris & Kates 2003a). As a consequence, the intension of the term is diluted and it has become highly unclear what 'sustainability' or 'sustainable development' actually mean or, viewed scientifically, *should* mean.[213]

Meanwhile, conceptual development in sustainability science falls short of maturity. Despite substantial effort to conceptualize sustainability in the form of goals, targets and indicators (Parris & Kates 2003b; Kates *et al.* 2005), its normative justification remains scrappy. Obviously, the assessment of progress toward sustainable development should be guided by a clear vision of sustainability. It is thus widely acknowledged that what is still missing in the debate is a profound theoretical and normative basis for the justification of sustainable development (Kates & Parris 2003; Clark *et al.* 2004; Keiner 2004; Komiyama & Takeuchi 2006). Therefore, the decisive question emerges whether or not the topic of sustainability is open to theory building (Ott 2001). Is it possible and useful to set up a well-founded 'Theory of Sustainable Development'?

This section suggests that the development of a 'Theory of Sustainable Development' is indeed possible, useful and urgently needed. To establish this proposition, we examine two of the salient approaches to sustainability proposed within the German discourse on sustainable development. We aim at calling attention to these well-founded sustainability conceptions, i.e. the 'Theory of Strong Sustainability' (*Greifswald*-approach) and the 'Integrative Sustainability Concept' (HGF-approach), which are widely acknowledged in the German debate but have not yet been of much impact in the international discourse on sustainability.

[213] We propose that "science" ought to provide criteria for distinguishing between well-founded and weak sustainability concepts. Hence the expression "should".

This section is organized as follows. Section 5.1.2 provides a systematic review of the German and the international discourse on sustainability. It shows that there is a lack of a profound theoretical and normative basis for the justification of sustainability. Section 5.1.3 asks why we do actually need a ‘Theory of Sustainable Development’ and offers several reasons for this claim. Subsequently, section 5.1.4 outlines the two salient sustainability conceptions proposed in the German discourse, i.e. the *Greifswald*-approach and the HGF-approach. Finally, section 5.1.5 concludes with the findings of this paper.

5.1.2 The International Discourse on Sustainability and its German Counterpart

Providing some background with regard to the content, this section reviews the international and the German discourse on sustainability. The main aim is to show the shortcomings in sustainability science in terms of conceptual and theoretical foundation.

5.1.2.1 The International Discourse on Sustainability

In the last 20 years, we have been confronted with several different discourses on sustainable development in the international debate, some of which are mutually exclusive (Redclift 2005). In the political realm, important documents mainly at the UN level, e.g. the declaration of Cocoyoc (1974) or the Brundtland report (WCED 1987), and international conferences in Stockholm (1972), Rio de Janeiro (1992) and Johannesburg (2002) mark important steps in the ongoing discourse on sustainability (UNEP 2002).

Meanwhile, several attempts emerged in the sciences to conceptualize the idea of sustainability. The 1990s brought some rather metaphorical illustrations of sustainability, such as (*a*) the 'three-pillar'-model, i.e. focusing at the development in each the ecological, the economic and the social dimension of society, (*b*) the 'prism' of sustainable development, which adds a fourth dimension (e.g. institutions or culture), and (*c*) the 'egg' of sustainability, which pictures sustainability as a mutually interdependent interaction between people and ecosystems (reviewed in Keiner 2004). Critics argue that these rather simple models for sustainability resulted in conceptual vagueness in the form of the ambiguity of the term, the plurality of purpose in characterizing and measuring sustainable development and the confusion of terminology, data and methods of measurement (Parris & Kates 2003a). Due to a deliberate interpretation of the different dimensions, the extension of the term sustainability has become extremely wide and its intension has been diluted. Rather as a clear and established conception, in the general public sustainable development appeared to be an arbitrary, ideological or illusory approach (Grunwald 2004b).

Also in the 1990s, another attempt to conceptualize sustainability came up: the debate of 'weak' sustainability vs. 'strong' sustainability (Neumayer 1999). This debate is based on different comprehensions as regards to the right structure of a fair bequest package, which signifies the specific goods and options that we are obliged to leave to future generations. It is assumed that these goods and options are provided by various types of capital, i.e. man-made capital, natural capital, cultivated natural capital (e.g. salmon farms, wineries), social capital (e.g. political institutions), human capital (e.g. skills, education), and knowledge capital. The decisive controversy is about whether or not or to what extent natural capital can be substituted with other types of capital.

The conception of *weak sustainability* assumes an infinite substitutability of all types of capital, as a consequence of a high substitution elasticity in the production function (Solow 1974). Weak sustainability advocates the position that (only) the sum of the various capital types must be kept constant for future generations' availability. Principally every loss of natural capital can be compensated by an increase in man-made, human or knowledge capital. The fair bequest package is conceptualized as non-declining utility over time, which is assessed in consumption options and measured by means of the genuine savings model (Atkinson *et al.* 1997).

In contrast, the conception of *strong sustainability* considers substitution to be seriously limited, as natural and man-made capital are not substitutes, but rather complements. According to strong sustainability, each type of capital is to be held constant separately in favour of future generations (Daly 1996). As natural capital gets scarce in many industrialized countries, strong sustainability consequently requires keeping natural capital intact over time, which is termed the 'constant natural capital rule' (CNCR) (Daly 1996).

Additionally, an intermediary position has been developed, based on the assumption that there is a complementary as well as a substitutive relationship between the capital stocks. The basic idea of this *intermediate sustainability* is to define the critical stocks for all parts of the natural capital, regardless the amount of the other capital stocks (SRU 2002; Ekins *et al.* 2003a). There is an ongoing debate about the superiority of one of the conceptions in the relevant literature.

Since the beginning of the 21st century, sustainability conceptions have been analyzed in the US-American discourse along three fundamental questions: 'what is to be sustained?', 'what is to be developed?' and 'for how long?' (NRC 1999; Parris & Kates 2003a; Kates *et al.* 2005). The existing sustainability conceptions and indicators sets, e.g. the United Nations Commission on Sustainable Development

Conception, the Wellbeing Index or the Environmental Sustainability Index, can then be classified with reference to these questions (Parris & Kates 2003a).

An interesting conception focuses on the characteristics of a sustainability transition, understood as meeting the needs of a stabilizing future world population while reducing hunger and poverty and maintaining the planet's life-support systems (NRC 1999). This problem-driven approach examines long-term trends in population, poverty or global environmental change and considers their effects on specific goals with respect to global problems, such as hunger, climate change and fresh water availability (Raskin *et al.* 2002; Kates & Parris 2003; Parris & Kates 2003a). In our view, this conception displays the typical merits and threats of a problem-driven approach. For example, there are no criteria to judge which global problems are of particular relevance and which are not (i.e. the specific subject realm sustainability refers to), or how to handle conflicts between different goals (i.e. the integrative aspect of sustainability). Such a conception also lacks a normative justification of sustainable development.

A further line of thought develops a research program for the advancement of Earth system understanding and global sustainability (Schellnhuber & Sahagian 2002; Clark *et al.* 2004; Kinzig *et al.* 2004; Schellnhuber *et al.* 2005). The authors list 23 crucial questions that need to be addressed for achieving global sustainability, which cover, for instance, the examination of the critical elements in the Earth system, typical functional patterns of environment – society interactions or the carrying capacity of the Earth (Schellnhuber & Sahagian 2002). A couple of methods for achieving global sustainability are being proposed, e.g. vulnerability analysis, guidance systems for sustainability or a social contract between society and science (Clark *et al.* 2004; Kinzig *et al.* 2004). This is surely a highly interesting and stimulating contribution, yet there is likewise a lack of theory building as regards to a sustainability conception. It is, for instance, questionable *why* sustainability should be about the specific research questions and not about other ones. Still a normative justification of sustainable development is a far off horizon.

Altogether it has become clear that even though there are some attempts to conceptualize sustainability in the US-American and international discourse, there is no profound normative basis for the justification of sustainable development so far.

5.1.2.2 The German Discourse

Ever since the UN-conference in Rio in 1992 sustainability has also taken hold in Germany with regard to both a ‘sustainable Germany’ as well as the contribution of Germany to a global sustainable development. This does not mean, however, that there has been consensus regarding the conceptual basis and the content of sustainability. Different actors such as economic organizations, the government, political parties, science, environmental actors, the North-South movement, unions or feminist groups set different priorities and favour different strategies ranging from achieving sustainability via technical progress to concepts of a regional subsistence economy (Brand & Jochum 2000).

The very first institution to deal with sustainability at a national level was the German Environmental Advisory Council (SRU). Its ‘Environment Report of 1994’ belongs to the central documents of the early German debate, as does a 1996 study entitled ‘Sustainable Germany’ by the internationally well-known Wuppertal Institute for Climate, Environment and Energy (Bund & Misereor 1996). Similarly important was a 1998 report of the Enquete-Commission of the German Parliament called ‘Protection of man and the environment’ that can be characterized as the first official document based on the “Three-pillar-approach”. In 2002 the German government adopted the German national sustainability strategy called ‘*Perspektiven für Deutschland*’ (Perspectives for Germany), which on the one hand was published rather late compared to many other countries, but was innovative in implementing an integrative view on the sustainability guiding principle, on the other hand.[214] The strategy highlights the four main principles of ‘intergenerational justice’, ‘quality of life’, ‘social cohesion’ and ‘international responsibility’, and is being followed up by the government on a two-year basis.

Similar to the international discourse there is a wide spectrum of opinions in Germany on how many dimensions sustainability should consist of, how they can be integrated and what substance they encompass. Increasingly, however, integrative, multidimensional approaches prevail. Furthermore, the question of how to define and achieve justice is widely being debated. Some approaches put a stronger focus on the intergenerational perspective, such as approaches of ‘strong’ vs. ‘weak’

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[214] For the German sustainability strategy confer URL: http://www.bundesregierung.de/nsc_true/Content/DE/__Anlagen/2006-2007/perspektiven-fuer-deutschland-langfassung,templateId=raw,property=publicationFile.pdf/perspektiven-fuer-deutschland-langfassung.

sustainability; other approaches give inter- and intragenerational issues equal weight. Note that even though the sustainability discourse has by now left the purely political and scientific sphere in Germany, sustainability as a term is to date not very established in the German public sphere nor is there a strong theoretical and normative core of sustainable development.

5.1.3 Do We Need a 'Theory of Sustainable Development'?

From the previous section it has become clear that there is a lack of a normative basis and theory building for the justification of sustainable development in the relevant literature. Yet is there a need for a specified 'Theory of Sustainable Development' (hereafter: T_{sust}) at all? We propose in this paper that there are indeed various reasons for setting up a well-founded theory.

First, a T_{sust} exhibits distinctive power. That means, it is possible (*a*) to distinguish sustainable from less or non-sustainable states in an unequivocal and transparent way; (*b*) to specify the 'subject realm' sustainability refers to; and (*c*) to operationalize sustainability for practical concerns (Grunwald 2004a, b). It is this distinctiveness that is viewed as critical for assessing whether or not there is progress toward sustainability. For instance, Clark *et al.* (2004) search for the general criteria and principles for distinguishing non-sustainable and sustainable futures and ask how progress toward sustainability could be reliably measured, and Schellnhuber *et al.* (2005) add that the immense challenge of sustainability science is to start seeking those criteria and principles that might define the sustainability of an Earth System future.

Second, a T_{sust} provides a well-founded basis for the justification of sustainable development. Faced with the numerous definitions, conceptions and indicator sets proposed for sustainability (Brand & Jochum 2000; Parris & Kates 2003a; Tremmel 2003; Kates *et al.* 2005), we argue that a critical-systematic path of theory building is the only fruitful avenue for achieving conceptual progress in sustainability science (Ott & Döring 2006). Only if we agree upon a stable normative core of a T_{sust}, we will be able to apply and adapt sustainability to specific policy fields (Parris & Kates 2003b). Indeed, if we skip fundamental normative questions, there is the danger that discussions at the level of application fall back to basic conceptual arguments that ought to be clarified in the first place.

Third, a scientific T_{sust} may perform as a 'rational corrective' to clarify the diffuse discourse on sustainable development that takes place in society (Grunwald 2004b). In the general public, the term 'sustainability' often appears to be vague and arbitrary. The agreement on what is a good sustainability theory within the scientific realm could curb the strategic and arbitrary use of the terms 'sustainability' and 'sustainable development' in the general public (Grunwald 2004a; Ott & Döring 2006). In our view, it is the responsibility of science to map a useful and reasonable comprehension of sustainability that politics and humankind will need to navigate toward a just and sustainable future (Kates & Parris 2003; Grunwald 2004b).

Thus, we propose that a T_{sust} is urgently needed in order to achieve theoretical progress in sustainability science. Assumed that we are in need for a T_{sust}, it may be still controversial, however, whether or not such a normative-scientific theory is in fact possible viewed from a philosophy of science perspective. As long as the criteria for a “theory” are not defined so as to only be fulfilled by empirical theories, sustainability can clearly be a theory. Neither the ‘*Theorie des kommunikativen Handelns*’ by Jürgen Habermas nor John Rawls’ ‘Theory of Justice’ has been denied the status of a theory. Even though it is true that such theories cannot give causal explanations and do not have high prognostic potential. There may be, at the border of science, systematizations, which make it hard to speak of “theory” and more appropriate to speak of “concepts” or “schemes”. More important, however, than the dispute about terms is, in our view, the question whether the argumentation of the sustainability conception is justified.

5.1.4 Two Salient Sustainability Conceptions within the German Discourse

We will now take a closer look at two salient attempts within the German discourse of setting up a sustainability conception or theory. With the thoughts of the previous section in mind we assume that a 'Theory of Sustainable Development' is highly needed and also possible in terms of theory of science. This section presents two well-founded conceptions to sustainability, the *Greifswald*-approach and the HGF-approach.

5.1.4.1 A 'Theory of Strong Sustainability': the Greifswald-Approach

The *Greifswald*-approach to sustainability has been developed by the ethicist Konrad Ott and the economist Ralf Döring (hereafter: O&D) at the university of Greifswald, Germany. This chapter is based on O&D's workings (Ott 2001, 2004; Ott & Döring 2004, 2006). It offers an outline of their 'Theory of Strong Sustainability'. Note that it has already been adopted by the German Environmental Advisory Council who advises the German Federal Government with respect to environmental affairs (SRU 2002).

The layers of sustainability discourse

For the sake of theory formation and clarity, O&D make a distinction between eight layers on which the scientific and ethical debate on sustainability occurs:

1. *Idea:* intra- and intergenerational justice
2. *Conception:* strong, intermediate or weak sustainability
3. *Rules:* management rules
4. *Guidelines:* resilience, sufficiency, efficiency
5. *Dimensions of Policy Making:* policy fields
6. *Objectives:* targets, time frames, set of instruments, indicators
7. *Special concepts and models*
8. *Implementation, Monitoring*

Each layer is associated with argumentative processes that lead to determinations with regard to the content. Referring to a structuralistic conception of theories (Stegmüller 1980), O&D state that layer 1 – 3 represent the theory's core, layer 4 encompasses 'bridging principles' (*Brückenprinzipien*) and layer 5 – 8 introduce applications. The connection between these levels is not deductive; rather results of higher layers give some orientation for lower layers. The determination of each layer with well-founded content, which is coherent (i.e. without contradictions) and consistent (i.e. closely linked), yields a 'Theory of Sustainability'.

It is far beyond this article to appreciate the *Greifswald*-approach to sustainability in an appropriate manner (but cf. Ott & Döring 2004). Rather this section focuses on the conceptual layers 1 – 2, which are described in some detail in the following.

5.1.4.1.1 Layer 1: Idea - Sustainability and Justice

There is a broad consensus in the international and German discourses that each sustainability theory should be about distributive justice as regards to present and future generations (Kopfmüller *et al.* 2001; Parris & Kates 2003a). It is essential, however, to decide whether sustainability implies an absolute or comparative standard. An *absolute standard* suggests that we are obliged to provide merely the basic needs present and future generations demand, whereas an *egalitarian-comparative standard* requires that present and future generations should be at least as well off as the contemporary ones living in industrial countries (Ott & Döring 2004). O&D advocate an absolute intragenerational and an egalitarian-comparative intergenerational standard (Ott & Döring 2004, 2006). As regards to present generations, they develop a pretentious absolute intragenerational standard by replacing the "basic-needs"-approach of the Brundtland-report with the capabilities-approach suggested by Sen (1986) and further developed by Nussbaum (2003). According to Nussbaum (2003), a good human life is based on various capabilities, such as the capability to have a good health or the capability for social interaction. To develop these human capabilities in an authentic, individual way is a legitimate moral claim (Nussbaum 2003). Viewed from the perspective of discourse ethics, recognition of this claim should be in the interest of each person while disagreement would assume genuine discrimination (Ott & Döring 2004). Complementary to the capability-approach, O&D suggest a comprehension of 'quality of life' based on the subjective contentment persons have regarding their own living conditions and mode of life (Vemuri & Costanza 2006). In such a conception, intragenerational justice is not only about human survival, but to provide the conditions for all humans to individually develop genuine capabilities and an authentic conception of a good life (Ott & Döring 2004).

Based on this absolute intragenerational standard, O&D suggest an egalitarian-comparative standard with respect to future generations. Refuting at first various no-obligation arguments within future ethics (Ott 2004), O&D make two independent arguments. With reference to Rawls' theory of justice (1973), the first argument demands to weave Rawls' 'veil of ignorance' in such a manner that the individuals exhibiting the 'original position' lack information on what generation they are belonging to. Representatives of each generation are present behind the 'veil of ignorance'. They choose about humanity's development paths without knowing at which position of the path they will find themselves in reality. Facing these circumstances, it would be rational to choose an egalitarian-comparative standard (Ott & Döring 2004, 2006).

The second argument for an egalitarian-comparative standard with respect to future generations refers to two important ethical concepts. First, the 'prohibition of primary discrimination' (*Verbot primärer Diskriminierung*) proposed by Tugendhat (1993) explicates the elementary moral intuition to treat equally what is equal and to treat unequally what is unequal. This 'principle of impartiality' holds that the prejudiced treatment of equal entities is apparently unjust.

Second, it is possible to consider societal principles, such as 'every person is to be treated equally in front of the law' or 'every vote counts equally in an election', as legitimate claims of equal treatment. This claim can be transferred to questions of distributive justice in the way that they would imply a 'presumption in favour of equality'. Hence, goods are to be distributed equally unless there are no good reasons in favour of an unequal treatment (e.g. particular merits or demands) (Ott and Döring 2004). This egalitarian conception is not evident in the philosophical discourse on justice and has to be justified against anti-egalitarian positions, championed by Krebs (2003), for instance.

Thus, the core idea of the second argument is that (*a*) the intuition that no generation is something exceptional in the chain of generations, connected with (*b*) the established 'prohibition of primary discrimination' suggested by Tugendhat (1993) and (*c*) the (controversial) ‘presumption in favour of equality’ altogether represent a sufficient premise base for an intergenerational egalitarian-comparative standard. Note that this standard should rest on a conception of quality of life (rather than living standard) in order to be reconcilable with ecological limits to growth (Ott & Döring 2006).

5.1.4.1.2 Layer 2: A Conception of Strong Sustainability

At layer 2, O&D refer to the debate of weak vs. strong sustainability, as described in the second section of this article. In accordance with Neumayer (1999), O&D state that there can be no scientific proof to falsify (in the classical sense of Popper) one of the two alternatives. Rather, a justification for the choice between weak and strong sustainability is a ‘prudent judgement’ and can be understood analogous to the rational choice between theories in the sciences. It is about making a based judgement for good reasons by assessing the arguments occurring on the conceptual level (Ott & Döring 2006).

O&D champion a conception of strong sustainability. Their judgement is based, among others, on the following arguments (cf. for a comprehensive treatment (SRU 2002, chapter 1; Ott & Döring 2004: 150ff):

First, the substitution of natural capital by man-made capital is seriously limited because of the multi-functionality occurring in many ecosystems. If man-made capital was a perfect representative, it would be possible to find a substitute for each ecological “function” (Jax 2005) of a lake or forest, for instance. This, however, is considered as impossible.

Second, the case of the Pacific island Nauru can be considered as a counterexample to weak sustainability (Gowdy & McDaniel 1999). Owing to the interest rates of the accumulated capital gained from the mining of phosphate resources, Nauru’s inhabitants afforded a high living standard for some decades. Yet the quality of life of Nauru’s people has rather decreased during this time, as 80% of the natural environment has been destroyed and life conditions have been characterized by a high rate of illnesses (e.g. diabetes, high blood pressure), alcoholism and decreasing life expectancy for men. Meanwhile, a huge fraction of the economic capital has been lost in business affairs. According to the measures of weak sustainability, Nauru is nevertheless the most sustainable place on earth. If the measure of weak

sustainability implies that countries with an impoverished environment, poor health conditions, widespread addiction to alcohol and a risky economical base are to be ranked as highly sustainable, there must be something at odds with the concept. If so, the Nauru-case counts as refutation of the weak sustainability approach.
Third, the 'false-negative/ false-positive criterion' suggests that we should choose that option bringing the morally most acceptable result if we erred in the empirical dimension. In the case of sustainability there are two hypotheses. H1 states that the widespread substitution of natural capital is harmless (*positive hypothesis*), which corresponds to weak sustainability. H2 proposes that H1 is wrong because of the potential consequences (*negative hypothesis*), which refers to strong sustainability. The important point here is that the consequences of a false-positive result are more disadvantageous than those of a false-negative result. Facing a false-negative result we "only" put more effort in nature conservation practices than would actually be necessary, whereas a false-positive result brings future shortages in natural capital and crisis. Therefore this criterion is a reason to reject H1 and to choose strong sustainability.
A conception of strong sustainability as proposed by O&D puts an emphasis on the maintenance of and on the investment in natural capital (Daly 1996; Ott & Döring 2004; Sachs & Reid 2006). Key objectives are thus to build-up resilient land-use systems, reduce current resource use rates, decrease the amount of climate change and to develop effective environmental institutions (Ott & Döring 2006).

5.1.4.1.3 A 'Theory of Strong Sustainability' – Basic Theoretical Structure

To sum up: The *Greifswald*-approach links sustainability to an intragenerational absolute standard in combination with an intergenerational egalitarian-comparative standard at layer 1 and a conception of strong sustainability at layer 2. From these conceptual determinations made at the first two levels, O&D derive some general rules at layer 3, e.g. the natural capital rule or the widely cited management rules of sustainability. At layer 4, the three bridging principles, i.e. resilience, sufficiency and efficiency, function as a hinge between theory and application. Based on this theoretical core, O&D turn at layer 5 to various policy fields, such as agriculture, fisheries and climate policy (Ott & Döring 2004). At layer 6, a specific objective for a certain policy field is being proposed, e.g. a 450 ppmv CO_2-concentration as a tolerable limit in climate policy. Finally, at layer 7 O&D suggest special concepts, e.g. the concept of ‘contraction & convergence’ in climate change, and some measures for monitoring at layer 8.

The basic theoretical structure of the ‘Theory of Strong Sustainability’ thus corresponds to the following filled layers:

1. *Idea:* Capabilities-approach, absolute intragenerational and egalitarian-comparative intergenerational standard with respect to quality of life
2. *Conception:* strong sustainability
3. *Rules:* constant natural capital rule, investment rules, management rules
4. *Guidelines:* resilience, sufficiency, efficiency
5. *Dimensions of Policy Making:* land-use systems, nature conservation, energy policy, environmental media (air, soil, water), mobility
6. *Objectives:*
   a) *Agriculture:* Maintenance of biodiversity, including agro-biodiversity, building of a system of conservation areas
   b) *Fisheries:* Community-based management following ‘safe biological limits’, reduction of negative ecological impacts of fisheries, sustained catch
   c) *Climate policy:* 450 ppmv CO_2-concentration as tolerable limit, delay in the increase of temperature to 0.2˚C per decade
7. *Special concepts and models*
8. *Implementation, Monitoring*

Far from being a catchphrase, ‘sustainability’ in this comprehension represents a clearly specified, scientific-normative theory that is able to stimulate a research program as regards to sustainability science.

5.1.4.2 The Integrative Concept of Sustainability: the HGF-Approach

The HGF-approach to sustainability has been developed by Jürgen Kopfmüller *et al.* (2001) at the *Institut für Technikfolgenabschätzung und Systemanalyse* (ITAS) in the Helmholtz Association (‘*Helmholtz Gemeinschaft*’) between 1997 and 1998. With its fifteen research centres the Helmholtz Association represents Germany's largest research institution. Also contributing in the development if the HGF-approach were the *Forschungszentrum Karlsruhe* (FZK) as well as various research institutes and groups of the *Forschungszentrum Jülich* (FZJ), the *Centre for Environmental Research* (UFZ) – all of them members of the Helmholtz Association - and the *Fraunhofer Gesellschaft.*

The aim of the HGF-approach is to offer a theoretically sound and practically workable basis for the operationalization of sustainability. Its main intention is to provide for distinctiveness as regards to the content and character of sustainability. Even though it does not claim to be a complete theory but rather a conception, in the context of this paper it contributes to what we understand as a better normative basis of sustainability.

On the basis of the Brundtland Report and the documents of the Rio-process three *constitutive elements* of sustainability are defined: inter- and intragenerational justice, global perspective and enlightened-anthropocentric approach. The focus on inter- and intragenerational justice in dealing with sustainability has far-reaching consequences. Sustainability is not "only" a matter of adjusting production methods and patterns of consumption as well as social institutions to "objective" limits (e.g. the carrying capacity of ecosystems or the availability of natural resources), as it is usually the case in an ecologically focussed understanding of sustainability. But rather it is a matter of the *internal organization of society under the idea of justice* – to be sure in consideration of the available knowledge, e.g. on the natural carrying capacity.

The global orientation in dealing with sustainability is supported by the following arguments: *(a) ethically,* sustainable development implies that all people (of present and future generations) have the moral right to have their basic needs and aspirations to a better life satisfied; *(b) in terms of problem,* many of the problems of sustainability such as climate change and loss of biodiversity are of global nature (even though they often are disproportionately distributed regionally) and, consequentially, require global answers; and finally *(c) in terms of action*, the coping with these global problems calls for common global endeavours, both concerning the identification and analysis of problems and their causes as well as concerning the development and implementation of the solutions.

The anthropocentric orientation is based on an "enlightened" understanding of anthropocentrism. Unlike approaches that grant the natural environment own "rights", this enlightened anthropocentric approach asserts the obligation to interact cautiously

with nature out of a well understood self-interest of mankind. The notion of man's self-interest, however, is not being equated with a short-sighted, exploitive consumption or wastage of nature, but rather refers to the long-term preservation of the various functions that nature fulfills for mankind. This encompasses not only the use of nature as resources or as sink for accumulated waste flows, but also the various cultural functions of nature such as the enabling of certain aesthetic experience. The reference to enlightened anthropocentrism confines the concept of sustainable development regarding the natural environment to questions of human use. Questions beyond that, such as questions of animal ethics, are being excluded – which does not make them irrelevant; they simply belong to a different "discourse".
The first step of the operationalization of these three constitutive elements consists of their "translation" into three *general goals of sustainable development*:

1. Assuring human existence
2. Maintaining the potentials for social production
3. Protecting the opportunities for development and action

Since the non-fulfillment of these three goals harms the constitutive elements, the goals are the condition precedent to sustainability. The second step of the operationalization of the three constitutive elements is to specify these general goals by identifying minimum conditions for sustainable development, namely the "What"-rules.
The "What"-rules substantiate the sustainability concept with respect to the various social sectors such as the handling of natural resources or equality of opportunity. Confer for an overview of the "What"-rules *Table 9*. In the context of the second goal of "Maintaining the social production potentials", the HGF-approach takes a "middle" stand on the question of "weak" vs. "strong" sustainability (see section 2 of this article). Rather as a question of "either-or", the issue of sustainability is understood as a question of the appropriate degree of substitution between the different forms of capital (natural capital, human capital etc.). According to the HGF-approach the substitution of man-made capital for natural capital is acceptable to a limited extent, insofar as the basic functions of nature (including the immaterial) are preserved.

Table 9: The what-rules of the HGF-approach[215]
Kopfmüller et al. 2001: 172

Goals / Rules	**1. Assuring human existence**	**2. Maintaining the potentials for social production**	**3. Protecting the opportunities for development and action**
	1.1 Protection of human health	2.1 Sustainable use of renewable resources	3.1 Equality of opportunity
	1.2. Provision of primary health care	2.2 Sustainable use of non-renewable resources	3.2 Participation in social decision-making processes
	1.3 Ensuring an existence independently	2.3 Sustainable use of the environment as a sink	3.3 Preservation of the cultural heritage and diversity
	1.4 Just distribution of options to use the environment	2.4 Avoidance of unjustifiable technical risks	3.4 Preservation of the cultural function of nature
	1.5 Balance of extreme differences in income and wealth	2.5 Sustainable development of man-made, human and knowledge capital	3.5 Preservation of the social resources

Beyond the "What"-rules, it is deemed crucial, under what general premises in society compliance with these substantial minimum conditions is possible at all. This leads to the *instrumental* conditions for sustainable development, the *"How"-rules*. These rules relate to the economic and political-institutional aspects of sustainable development and pertain to the following ten social fields: internalization of external social and ecological costs, adequate discounting, encumbrance, fair world economic framework, fostering of international cooperation, reactiveness of the society, reflexiveness of the society, steering capacity, self-organization and balance of forces.

The "What"- and "How"-rules together further unfold the normative aspects of sustainability in relation to the relevant sectors in society. They serve as test criteria, which can help to identify sustainable and unsustainable conditions and

[215] It is awkward - or better: circular - that the of-what rules of the HGF-approach include the term "sustainable". Yet the term "sustainable" is to be explained by these terms.

developments: when the rules are adhered to, the general goals can be achieved; that in turn leads to a positive evaluation as regards to sustainability. That is to say, when all rules are adhered to, the postulate of sustainable development is being realized, when just one rule is violated, there is no sustainability.

The rules, however, allow for a certain degree of consideration in order to be able to deal with possible conflicts of rules or goals. Any analysis always has to consider the entirety of rules; and for each rule at least a core part (‘*Kernbereich*’) always has to be fulfilled. Beyond this core part, however, they do not have to be adhered to in an all-or-nothing manner: justifications for one decision suggested by one rule can in cases of conflict be thwarted by justifications belonging to another rule.

Altogether the rules constitute the *normative* basis and orientation for learning processes in society as regards to sustainable development. They are not meant to embody the ”nice, good and true” in society, as integrative concepts of sustainability are often accused of. Rather the substantial rules are the *minimum standards* that all members of the global society, including the coming generations, have a right to have guaranteed. The integrative concept encompasses by no means the sum of all desirable political, social and economic goals, but solely a certain welfare base (‘*Wohlfahrtssockel*’). Beyond that base there can be various other legitimate and desirable individual and societal goals, whose fulfilment, however, cannot be constitutive for the concept of sustainability. In terms of time, the system of rules is not infinitely valid, but rather serves as a kind of ‘*morale provisoire*’, i.e. a well-founded compass regarding sustainable development, which does not preclude future advancements.

In order for the normative reflections to be filled with content and the rules not to stay solely appellative, a solid amount of empirical knowledge is necessary. This joining of the normative reflections with empirical knowledge takes part on the level of *indicators*. For each rule there are set indicators, which adequately represent the issues this respective rule refers to. This way the rules are further substantiated as well as contextualized.

Ever since its original development the HGF-concept has been applied to various cases in Germany and beyond, such as to the development of a municipal reporting system for sustainable development in the German cities of Leipzig and Halle, to regional questions of agriculture and food industry, to the question of education for sustainable development and to the question of the generation of energy from grassland (Coenen & Grunwald 2003; Kopfmüller 2006). It is currently being applied to the subject of global megacities.

5.1.5 Conclusion

Despite various attempts to establish sustainability, theoretical development within sustainability science remains piecemeal. In accordance with some key publications in the German and international discourse (Kopfmüller *et al.* 2001; Kates & Parris 2003; Clark *et al.* 2004; Keiner 2004; Ott & Döring 2004; Komiyama & Takeuchi 2006), this article identifies a lack of a well-founded normative basis for the justification of sustainable development.

In order to fill this gap within sustainability science, we offer two conceptions proposed in the German discourse, namely the *Greifswald*-approach (Ott & Döring 2004) and the HGF-approach (Kopfmüller *et al.* 2001; Kopfmüller 2006). Both conceptions comprehend sustainability as a multi-level concept that is comprised of a well-founded normative core, i.e. idea, conception and management rules, in combination with a couple of layers of application, i.e. dimensions of policy making, objectives and indicators. Such a multi-level concept provides the means for setting up a well-founded 'Theory of Sustainable Development'.

It has also become clear that both concepts hold many merits, each in different fields. Even though both highly value the normative foundations of sustainability, the HGF-approach is more of a pragmatic-analytical approach, while the *Greifswald*-spproach attaches more value to ethical justifications. The *Greifswald*-approach takes a more explicit stand on the question of strong vs. weak sustainability, while the HGF-approach leaves much room for various positioning in this respect. In addition, the *Greifswald*-approach focuses strongly on environmental objectives, whereas the HGF-approach places more emphasis on social questions and justice.

Especially their application to real life cases makes clear, however, that also numerous commonalities and converging tendencies between the two exist. This indicates that there is a considerable potential for further advancement of the two approaches within their respective internal logic (Ott 2006). Such advancement and expansion have not been realized yet and this represents a fruitful avenue for further research.

We suggest that the topic of sustainability is open to theory building. Only a critical-systematic approach to theory building provides a well-founded normative basis for the justification of sustainable development. A strong normative core in turn paves the way for an established and thus non-arbitrary application of sustainability to specific policy fields. It also brings distinctiveness in the form of the separation of sustainable from less or non-sustainable states, the specification of the subjects sustainability refers to and measures to operationalize sustainability for practical concerns (Grunwald 2004a, b). It allows the assessment whether or not there is progress toward sustainability in a specific case study.

We further suggest that it is in the responsibility of science to conduct a discourse-rational choice about the best sustainability conceptions we have (Grunwald 2004a). Such an enlightened comprehension of sustainability puts forward a scientific claim to rightness or truth and may inform politics or the societal discourse in concerns of sustainable development (Ott & Döring 2006). The scientific agreement on the best sustainability conceptions may confine the prevalent strategic and arbitrary use of the terms ‘sustainability’ and ‘sustainable development’ in the general public. A normative ‘Theory of Sustainable Development’ is urgently needed for achieving progress in sustainability science.

This thesis thus assumes a conception of strong sustainability (*Greifswald*-approach) or at least of intermediate sustainability (HGF-approach) in the following. The maintenance of natural capital is therefore an important objective for achieving sustainable development.

5.2 Ecological Resilience and Critical Natural Capital

This section reveals the relevance of the concept of ecological resilience for a conception of critical natural capital. I have delineated my thoughts on that topic in a recent article that is published in the journal *Ecological Economics* (Brand 2009). The following is thus a revised version of this article.

5.2.1 Introduction

Sustainable development represents one of the key challenges of the 21st century (Sachs 2005; Clark 2007). Even though there is a wide political consensus on the principal idea of sustainability, scientific agreement regarding the key question 'what to sustain?' (Dobson 1996; NRC 1999; Kates *et al.* 2005) is a far off horizon. It is still controversial what types of capital, i.e. natural capital (e.g. ecosystems, air, water), cultivated natural capital (e.g. salmon farms, wineries), man-made capital (e.g infrastructure), social capital (e.g. political institutions), human capital (e.g. skills, education) and knowledge capital ought to be preserved in favour of current and future generations (Costanza *et al.* 2007).

At this conceptual level of sustainability science, basically two positions fight for validity (Neumayer 1999; Ott & Döring 2004). *Weak sustainability* holds that utility (or well-being) ought to be maintained over intergenerational time scales. In this conception, natural capital and man-made capital are viewed as substitutes within specific production processes. Consequently, the stock of the natural capital can be depleted, unless the utility over time is declining (Pezzey 1992; Norton & Toman 1997). In contrast, *strong sustainability* states that natural capital and man-made capital must be viewed as complementary. We are obliged to keep each type of capital intact over time. Thus, the whole stock of natural capital ought to be preserved for current and future generations in the long run (Daly 1996) (confer for a detailed treatment of "sustainability" section 5.1).

The concept of critical natural capital emerged as a trade-off between these two extreme positions. It signifies the part of the natural capital that performs important and irreplaceable environmental functions, i.e. ecosystem services (Jax 2005) that cannot be substituted by other types of capital (de Groot *et al.* 2003; Dietz & Neumayer 2007). Paradigmatic examples include essential ecosystem services, such as freshwater resources, climate regulation and fertile soils (UNEP 2005b). It is this importance for the quality of life and the survival of humans that makes critical natural capital such an important objective of sustainability. Critical natural capital represents thus the part of the natural environment that ought to be maintained in any circumstances in favour of present and future generations. In addition, the identification and management of critical natural capital is a promising tool for a sound approach to environmental policy (Ekins *et al.* 2003a). The quest for a clear conceptualization of ‘critical natural capital’ is hence worth the effort.

Yet conceptual confusion is immense, as numerous scientific disciplines and societal groups bring their own perspective in valuing nature. It is indeed highly unclear what makes natural capital ‘important’, ‘irreplaceable’, and therefore ‘critical’. In other words, it is controversial which measure would be appropriate to reflect or mirror ‘criticality’. Is it the ecological importance we ascribe to certain habitats due to a high degree of species richness or “naturalness”? Is it the economic value that some ecosystem services bring for human society? Or is it the socio-cultural relevance of the “landscape”? The important point here is that we urgently need well-founded criteria to assess the specific criticality of natural capital stocks (MacDonald *et al.* 1999; Ekins *et al.* 2003a).

This article examines the link between the concept of ecological resilience and critical natural capital. I state that the empirical estimation of ecological resilience can help a great deal in assessing the “ecological criticality” of specific parts of renewable natural capital. More specifically, I propose that the amount of resilience can be used to estimate the degree of threat certain ecosystems are prone to. The concept of ecological resilience therefore adds a further important criterion to build a comprehensive conception of criticality.

The article is organized as follows. The first section offers a short description of the concept of ecological resilience with an emphasis on questions regarding the conceptualization and measurement of ecological resilience. Subsequently, the second section revisits the concept of critical natural capital and formulates a comprehensive approach to criticality. Based on these conceptual reflections, the third section examines the link between the concept of ecological resilience and a conception of critical natural capital. Finally, the fourth section concludes with the findings of this paper.

5.2.2 The Concept of Ecological Resilience

Among the scientific concepts currently used in sustainability science, ‘resilience’ is one of the most prevalent and topical (Kates *et al.* 2001; Foley *et al.* 2005). First of all, two distinct meanings of the term must be distinguished. The first one refers to dynamics close to equilibrium and is defined as the time required for a system to return to an equilibrium point following a disturbance event. It has been coined “engineering resilience” (Holling 1996) and is largely identical to the stability property “elasticity” (Grimm & Wissel 1997). The second meaning of resilience refers to dynamics far from any equilibrium steady state and is defined as the capacity to absorb shocks and still maintain “function” (cf. for the ambiguous term ‘function’ Jax 2005). This meaning has been termed “ecological resilience” (Gunderson & Pritchard 2002; Anderies *et al.* 2006) and it is this second kind of resilience to which I refer in this text.

The concept of ecological resilience emerged in ecology in the 1960 – 1970s but has been adopted since then by numerous scientific disciplines, e.g. sociology, economy or political science (Folke 2006). It is currently used either as a descriptive concept that is applied primarily to ecological systems, i.e. ecological resilience (cf. Bellwood *et al.* 2003; Nyström 2006), or as a boundary object, a term that facilitates communication across disciplinary borders, i.e. social-ecological resilience (cf. Brand & Jax 2007). In the latter interpretation, the concept is viewed as an innovative perspective to analyze coupled social-ecological systems (Walker *et al.* 2006; Walker & Salt 2006).

This article focuses on the former descriptive meaning of the term. Ecological resilience is defined as *the capacity of an ecosystem to resist disturbance and still maintain a specified state*. In this definition the concept gets close to the stability concept ‘resistance’, as identified by Grimm and Wissel (1997). This may underestimate other important characteristics of ecological resilience, such as the capacities for renewal, reorganization and development (Folke 2006). Yet this definition is in my view useful and workable to be used for the measurement of ecological resilience in real-world ecosystems.

How to measure ecological resilience? Carpenter *et al*. (2005) recently noticed that ecological resilience cannot be measured directly. Rather it must be estimated by means of resilience surrogates, i.e. indirect proxies that are derived from theory used in indicating resilience (for resilience theory cf. Walker *et al.* 2006). Surrogates for ecological resilience refer to the concept of resilience mechanisms, e.g. ecological redundancy, response diversity or ecological memory (Nyström 2006), the concept of maintained system identity (Cumming *et al.* 2005), probabilistic resilience and percolation theory (Peterson 2002b) or to approaches using the concept of alternative stable states and ecological thresholds (Scheffer *et al.* 2001; Bennett *et al.* 2005a). Before I will expand in some detail on the latter approach it must be noted that each estimation of ecological resilience is based on a comprehensive resilience analysis, which includes the identification of the specific disturbance regime and a societal choice of the desired ecosystem services. Confer for a detailed review of a resilience analysis Carpenter *et al.* (2001), Walker *et al.* (2002) and Resilience Alliance (2007) (confer for a detailed treatment of the operationalization of ecological resilience section 3.4 and for the threshold approach section 3.4.2.3.1).[216]

The threshold (T-) approach to resilience surrogates is used widely in the relevant literature (Carpenter *et al.* 2001; Peterson *et al.* 2003b; Bennett *et al.* 2005a). Despite its actual prominence it is based on two controversial assumptions. The first assumption holds that ecosystems can shift non-linearly between alternative stable states that are separated by ecological thresholds. For example, coral reefs can show an algae-dominated or a coral-dominated state while savannahs may exhibit a grassy or woody state. Yet this is true for many ecosystem types (Folke *et al.* 2004; Walker & Meyers 2004) but not all (Schröder *et al.* 2005). Indeed, the weight of empirical evidence shows that the relative frequency of the occurrence of alternative stable states across systems is higher for systems controlled by environmental adversity, e.g. deserts, arctic tundra or savannahs, than those controlled rather by competitive interactions, e.g. forests or coral reefs (Didham & Norton 2006). The T-approach may thus be limited to those systems controlled by environmental adversity.

The second assumption states that ecosystem dynamics can be understood by analyzing a few key variables, which is termed the 'rule of hand'. Key variables are subdivided into fast variables and slow variables according to the turnover rates in space and time (Rinaldi & Scheffer 2000; Walker *et al.* 2006). The important (and controversial) assumption here is that the slow variables are viewed to 'control' the

[216] I focus here on the threshold approach, as this approach is most illustrative and established within resilience research. Also, this approach refers to the "precariousness" of a system (confer below). Yet there are other approaches to operationalize ecological resilience (cf. section 3.4.2.3).

whole ecosystem in determining the system's position within a stability landscape (Walker *et al.* 2004). The value of the slow variable, e.g. the abundance of woody plants in rangelands or the phosphate concentrations in a shallow lake, is thus regarded as the most relevant factor for the maintenance of ecological resilience (Gunderson & Walters 2002).

However, the concept of 'rule of hand' is limited, as it is dependent on a certain educated guess regarding the variables and parameters that are to be included in the model (Schmitz 2000). A further shortcoming is that this approach must ignore the individual variability of organisms (Grimm 1999). The 'rule of hand'-approach is not false but rather partial and may be complemented by bottom-up approaches (Grimm 1999), e.g. individual-based models (Grimm & Railsback 2005). From the perspective of individual-based modellers, it is highly controversial, for instance, whether the key variables and the controlling slow variables actually can be identified (V. Grimm, *personal communication*).

The threshold approach for the estimation of ecological resilience is thus restricted to ecological situations in which both assumptions hold: ecosystems must exhibit alternative stable states and it must be possible to identify the key controlling variables. When these necessary preconditions are met, the threshold approach tracks ecological resilience by means of a bifurcation diagram. As illustrated in *Figure 20,* this diagram plots the equilibria of an ecosystem on axes of a fast variable and a slow variable. In the case of a shallow lake the fast variable is represented by the abundance of submerged plants while the slow controlling variable corresponds to the phosphate concentrations in the sediment of the lake. The plot then shows upper and lower sets of stable regions (the solid lines in *Figure 20*) separated by two ecological thresholds (ET_1 and ET_2) and an unstable set of equilibria (the dashed line). The important point here is that the value of the slow variable (CV_{sv}) is regarded to control an ecosystem's position in state space, and thus, to be responsible for the maintenance of ecological resilience of the whole system. Therefore, the degree of the ecosystem's ecological resilience is estimated as the distance from the current value of the slow variable (CV_{sv}) to the value of the ecological threshold (ET_1) (Rinaldi & Scheffer 2000; Peterson *et al.* 2003b). In other words: in this approach, the slow variable performs as a resilience surrogate, as it is this variable that controls the position of the whole ecosystem within state space. Note that this methodology of estimating ecological resilience focuses on the precariousness of the system, i.e. the current trajectory of the system and proximity to a limit or threshold. Clearly, there might be other important facets of ecological resilience, such as resistance or latitude (Walker *et al.* 2004).

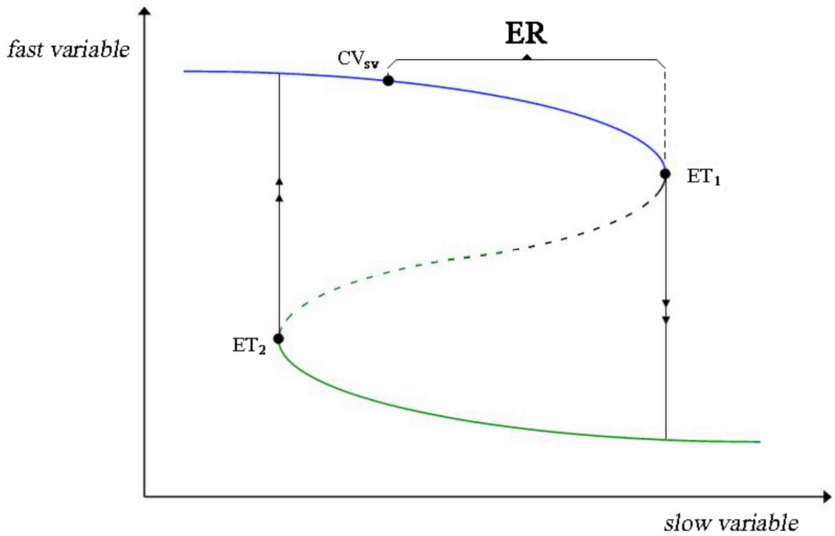

Figure 20: Bifurcation diagram of a system described by a fast and a slow variable: the stable regimes are given by the solid lines and the unstable states by the dashed line. ET_1 and ET_2 represent ecological threshold points and CV_{SV} signifies the current value of the slow variable. Ecological resilience (ER) is measured as the distance from CV_{SV} to ET_1.
modified from Scheffer and Carpenter (2003) and Brand and Jax (2007)

Thus, the estimation of ecological resilience means to examine both the current value and the threshold value of the slow variable. Principally, the former is easy to measure, such as the phosphate concentrations in lake sediment, for instance. What appears to be more difficult is to predict the location of the ecological threshold point (or zone). This further difficulty of the threshold approach is hardly acknowledged but evident: to estimate the ecological resilience by means of the 'slow variable surrogate' it is necessary to predict the position of the ecological threshold as regards to the slow variable (maybe besides across-site comparisons between 'classes' of ecosystems). This is a difficulty because predictions in ecology are hard to achieve. Yet there are at least three (perhaps interrelated) methods for predicting the position of ecological thresholds. Those are: (1) the extrapolation of empirically estimated return times of controlling variables well distant from the threshold (Wissel 1984); (2) the examination of standard deviations of fast variables in the vicinity of thresholds (Carpenter & Brock 2006); and (3) the repeated calculation of the Fisher Information, i.e. a statistical measure of indeterminacy of a specific ecosystem (Mayer *et al.* 2006). These sound methods have not been of much impact in the relevant literature but it would be interesting to examine their characteristics and interrelations more profoundly.

Note that this early type of threshold approach referring to the ‘slow variable surrogate’ and bifurcation diagrams has recently been developed further by Bennett *et al*.’s (2005) alternative approach where they also take into account time (i.e. ‘how fast is the slow variable moving toward or away the ecological threshold’) and whether ecological thresholds are dynamic or static. A comprehensive discussion of Bennett *et al*.’s (2005) sound approach is beyond the realm of this article.

The message of this section is that ecological resilience represents a relatively clearly defined stability concept that is embedded in a rich resilience theory. It is principally possible to empirically estimate surrogates for ecological resilience. This will be important in the third section when I connect the concept of ecological resilience to the concept of critical natural capital. In the next section we will at first have a closer look at the specific conceptualization of the term ‘critical natural capital’.

5.2.3 Critical Natural Capital: Some Conceptual Remarks

The concept of critical natural capital has been developed by Turner (1993) following capital theory in economics (cf. for the term ‘capital’ Ekins *et al.* 2003). It gained some attention with the recent EU-project CRITINC (Ekins *et al.* 2003a; Ekins 2003) and has been applied to several EU-countries, such as France (Douguet & O'Connor 2003) and the UK (Ekins & Simon 2003). The following delineations in this section must be viewed as complementary to the sound work of the CRITINC project.

The concept of critical natural capital is obviously based on the concept of natural capital, often understood as any stock of natural resources or environmental assets that provides a flow of useful goods or services, now and in the future (Pearce & Turner 1990; MacDonald *et al.* 1999; de Groot *et al.* 2003). The term ‘natural capital’ has been criticized for its reductionistic and utilitarian connotations (Chiesura & de Groot 2003), praised for its terminological strengths (Dobson 1996) and continues to stimulate a debate about its accurate conceptual intension and extension (Ott & Döring 2004; de Groot 2006). It is used widely to signify a myriad of components (e.g. resources, biodiversity, fertile soil, ozone layer), properties (e.g. ecological resilience, ecosystem health, integrity) and dispositions (e.g. regulative or assimilative capacities). Natural capital is thus a multidimensional meta-concept for a plurality of interrelated and heterogeneous stocks that perform various functions and services for human society (Chiesura & de Groot 2003; Ott & Döring 2004; Aronson *et al.* 2006a).

Regarding the multidimensional character of natural capital, it is not surprising to find conceptual confusion about ‘critical natural capital’ (Turner 1993; MacDonald *et al.* 1999). This is due to the existence of different domains under which natural capital can be considered as critical, as different disciplines bring different conceptual frameworks to value ecosystems (Chiesura & de Groot 2003). The decisive question is: ‘critical for what and for whom?’ (de Groot *et al.* 2003). Considering all the relevant literature at least six domains may be distinguished under which natural capital is evaluated as critical:

1) socio-cultural: natural capital becomes important, crucial or vital for a particular social group, as it provides the socio-cultural context for human society in terms of non-materialistic needs, e.g. health, recreation, scientific and educational information, cultural identity, source of spiritual experience or aesthetic enjoyment (Chiesura & de Groot 2003; Kazal *et al.* 2006).

2) environmental: natural capital is ecologically valued for its importance in terms of naturalness, biodiversity, irreversibility or uniqueness (de Groot *et al.* 2003; de Groot 2006), for instance.

3) sustainability: this domain refers to the debate of weak vs. strong sustainability described in the introduction of this paper. Natural capital is viewed as critical as regards to human well-being if it is non-substitutable with other types of capital (Turner 1993; Neumayer 1999; Dietz & Neumayer 2007). Good examples are life-securing ecosystem services, such as the provision of food, raw materials or drinking water.

4) ethical: a loss of natural capital can be morally disadvantageous in that moral values are being violated (Dietz & Neumayer 2007). For example, from the standpoint of sentientism the preservation of higher developed animals, e.g. bears, beavers or casuaries (a bird), would be *prima facie* regarded as critical (Haider & Jax 2007).

5) economic: the loss of natural capital can also bring about very high economic costs. These costs can be validated by the full spectrum of monetary valuation (de Groot 2006).

6) human survival: natural capital becomes obviously critical when without it human life would not be possible (Dobson 1998). Examples are climate regulation, flood regulation or fertile soils.

Consequently, definitions of critical natural capital are manifold, as specific domains of criticality are being stressed. Some definitions refer to one domain of criticality only. For instance, Turner's (Turner 1993: 11) definition "[t]he constraint [of critical natural capital] will be required to maintain populations/resource stocks within bounds thought to be consistent with ecosystem stability and resilience" is ecological, whereas the definition put forward by Douguet and O'Connor's (Douguet & O'Connor

2003: 237): “natural capital which is responsible for important environmental functions and which cannot be substituted in the provision of these functions by manufactured capital” stresses the sustainability domain. Alternative definitions try to include a higher amount of criticality domains, as for instance, the definition proposed by Dietz and Neumayer (Dietz & Neumayer 2007: 619): “we may ‘ring-fence’ as critical any natural capital that is strictly non-substitutable (also by other forms of natural capital), the loss of which would be irreversible, would entail very large costs due to its vital role for human welfare or would be unethical”.

Each of these definitions refers to specific domains of criticality only and can thus be criticized as partial and incomplete. There is the need for a more comprehensive approach to criticality. In this article I consider natural capital to be critical if it applies to at least one of the six domains of criticality, i.e. the socio-cultural, ecological, sustainability, ethical, economic or the human survival domain. In this conception criticality comes in degrees. Criticality is dependent on *(a)* the amount of significance within one domain of criticality (e.g. the socio-cultural importance, the ethical value, the economic costs), *(b)* the amount of domains under which the natural capital is valued (i.e. the more domains the more critical) and *(c)* the weighing of the different domains. Hence, different parts of natural capital can have various degrees of criticality. It is also important to note that criticality is to some degree context-specific (de Groot *et al.* 2003), as it is related to certain standards of living and human values that may change over time.

Altogether it has become clear that the concept of critical natural capital is by no means rooted solely in the natural sciences but also and much more in the full array of social sciences and the humanities. In the subsequent section I will have a closer look at the relation of the concepts of ecological resilience and critical natural capital.

5.2.4 Ecological Resilience and Critical Natural Capital

What is the relation between ecological resilience and critical natural capital? Previous work determined several links of ecological resilience to criticality. For instance, Serrão *et al.* (1996) point to environmental criticality, i.e. a state of nature in which the extent of environmental degradation passes a threshold beyond which current levels of social welfare may not be supported. Even though no clear measure is being proposed Serrão *et al.* (1996) suggest that the estimation of environmental criticality requires information about ecological resilience.

Alternatively, Deutsch *et al.* (2003) acknowledge ecosystem performance as a criterion for criticality. By 'ecosystem performance' they mean 'the dynamic, often non-linear interrelations between populations and communities of plants, animals and microorganisms and their energetic, hydrological and biogeochemical environment'. This ecosystem property is perceived as an underlying pre-requisite for human well-being because it generates and sustains the flow of ecosystem services. This notion is closely related to several other terms used in the literature on natural capital, e.g. functions-of nature (de Groot *et al.* 2003), regulation functions (de Groot *et al.* 2002), life-support functions (Ekins *et al.* 2003b) or regulating services (UNEP 2005b). The important point here is that Deutsch *et al.* (2003) link ecosystem performance to resilience theory, and in particular to the concepts of pulse disturbance, alternative stable states, slow variables and biodiversity. Hence, the relation of ecological resilience and critical natural capital has been the subject of scientific debate since several years.

In the subsequent delineations I will follow De Groot *et al*.'s (2003) approach to criticality. In order to stress the ecological aspect within a conception of critical natural capital, its multi-dimensionality can be boiled down to two criteria: importance and degree of threat (cf. *Figure 21*). As De Groot *et al.* (2003) argue, it is at first appropriate to conceptualize criticality with reference to the importance society ascribes to natural capital. Yet human activities can bring changes in the natural capital and these changes, in turn, can affect its ecological, socio-cultural or economic importance, for instance. Therefore, another criterion for determining criticality is found: 'degree of threat'. De Groot *et al.* (2003) argue that the degree of threat ecosystems are exposed to is assessed based on changes in quantity and quality of the remaining natural capital. Ecosystem quantity simply refers to the percentage a region is covered by a certain ecosystem type and can be determined by means of land cover databases. In contrast, ecosystem quality is being related to the concepts of integrity and vulnerability and estimated by changes in species richness or pressure on ecosystems.

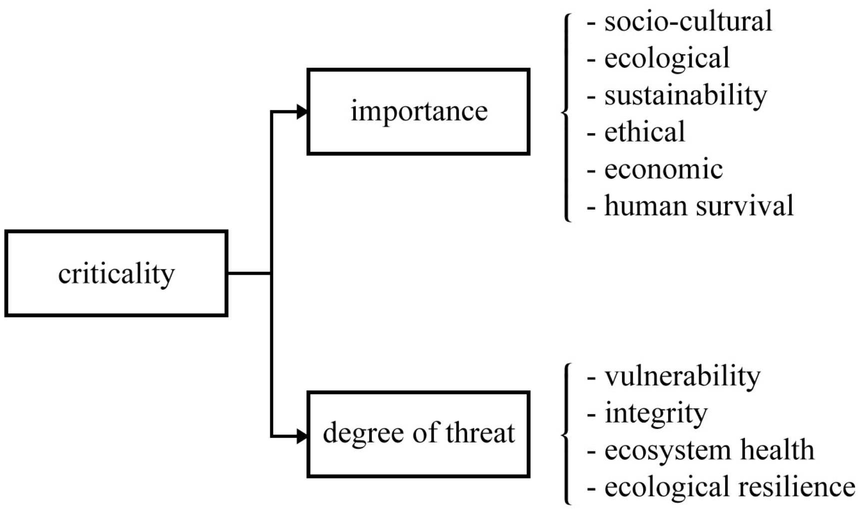

Figure 21: A conception of critical natural capital. The 'criticality' results from the two criteria 'importance' and 'degree of threat'
modified from De Groot *et al.* (2003)

This article follows the approach to criticality suggested by De Groot *et al.* (2003) but proposes an alternative method to assess the degree of threat based on the concept of ecological resilience. This section thus develops an approach to 'ecological criticality' and puts an emphasis on the ecological domain of criticality. It is certainly not the suggestion to neglect neither the relevance of other domains of criticality, e.g. the socio-cultural, ethical, economic importance nor the criterion of ecological importance, which includes e.g. uniqueness, naturalness or biodiversity. What I propose is rather that the concept of ecological resilience provides a useful means for estimating the degree of threat ecosystems may face.

My argument goes as follows. Ecological resilience is defined in its ecosystem services-related meaning (Brand & Jax 2007) as the underlying capacity of an ecosystem to maintain desirable ecosystem services in the face of human use and a fluctuating environment (Carpenter *et al.* 2001; Folke *et al.* 2002). The loss of ecological resilience thus indicates whether ecosystems are prone to shifts to undesirable ecosystem states that cease to deliver the ecosystem services people value in a specific case (Folke *et al.* 2004). I infer from this first that an ecosystem's amount of ecological resilience is directly linked to the degree of threat this ecosystem may face. Second, I propose that in order to quantify the degree of threat an ecosystem is exposed to, it is necessary to estimate the ecological resilience of an ecosystem in a specific case.

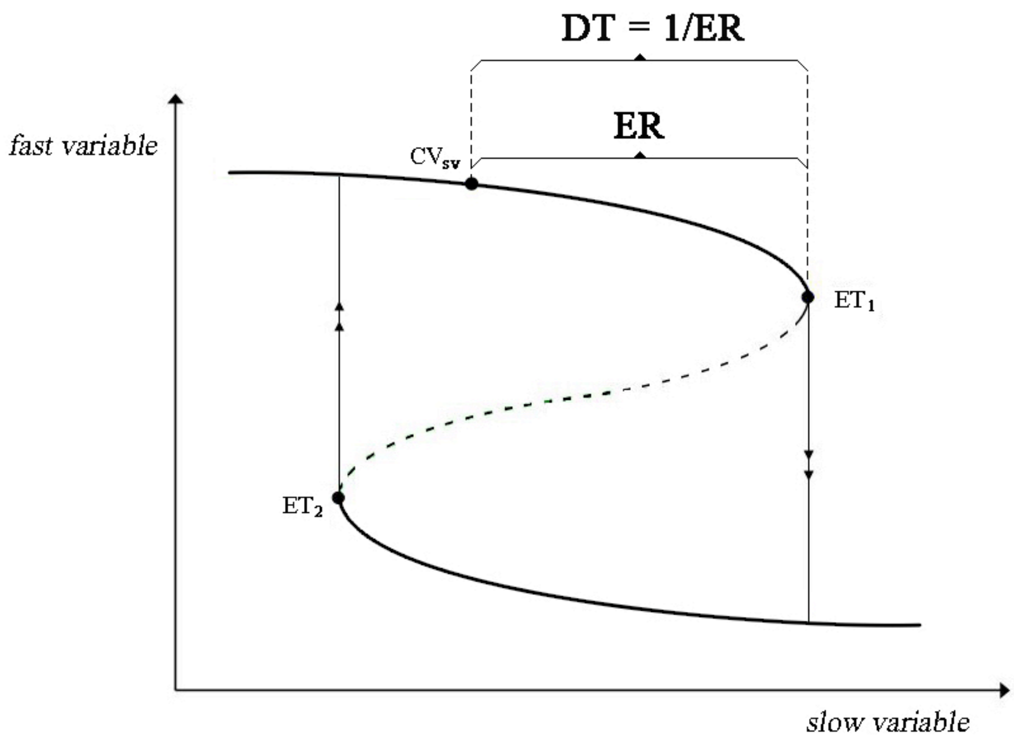

Figure 22: Bifurcation diagram of a system described by a fast and a slow variable: the stable regimes are given by the solid lines and the unstable states by the dashed line. ET_1 and ET_2 represent ecological threshold points and CV_{SV} signifies the current value of the slow variable. Ecological resilience (ER) is measured as the distance from CV_{SV} to ET_1. The degree of threat (DT) is inversely related to ER
modified from Scheffer and Carpenter (2003) and Brand and Jax (2007)

Consider again the bifurcation diagram in *Figure 22*. By using the threshold approach, a surrogate for ecological resilience can be estimated as the distance from the current value of the slow variable (CV_{sv}) to the (predicted) value of the ecological threshold (ET_1). The important point here is that ecological resilience (ER) is directly linked to the degree of threat (DT) the ecosystem is exposed to. Strictly speaking, the amount of ER is inversely related to DT, that means ER = 1/ DT. If the amount of ER is low (i.e. the system is close to the ecological threshold ET_1) the DT would be high, and if the amount of ER is high (i.e. the system is far from the ecological threshold ET_1), the DT would be low. Clearly, the inverse relationship of ecological resilience and the degree of threat applies to each surrogate of ecological resilience. That means, it is also possible to estimate the degree of threat by means of other resilience surrogates, such as resilience mechanisms (Nyström 2006), maintained system identity (Cumming *et al.* 2005) or percolation theory (Peterson 2002b).

Consider a simple model of a shallow lake ecosystem as an example. Shallow lakes can exhibit two stable regimes with respect to nutrient load, i.e. a clear-water regime with aquatic plants and a turbid regime without vegetation. If the lake is in the clear-water regime, an increase of the nutrient level will lead to a gradual and moderate rise in turbidity until the critical turbidity for plant survival is reached. At this point, vegetation collapses and the lake shifts to the turbid regime. Following the threshold approach, ecological resilience corresponds to the distance of the current value of nutrient concentrations to the critical level. Thus, if the specific shallow lake had a very low nutrient level, the ecological resilience would be high. As a consequence, its degree of threat would be low. This example is certainly oversimplifying (Scheffer & van Nes 2007). Yet I want to illustrate here that ecological resilience theory is principally rich enough to spell out the degree of threat ecosystems are prone to.

It is important to note that the amount of ecological resilience can only be used as a criterion of criticality for specific dimensions of natural capital, as natural capital is a multi-facet concept that includes divergent environmental media, such as air, water or habitats. According to the classifications proposed by de Groot *et al.* (2002) and Ekins *et al.* (2003b), ecological resilience is thus applied exclusively to natural capital that *(a)* is renewable, *(b)* refers to the basic type 'habitat' (i.e. ecosystems, flora and fauna) and *(c)* refers to life-support functions, i.e. the capacity to sustain "ecosystem health" and "function". Paradigmatic examples are represented by any ecosystem type, e.g. shallow lake, savannah or boreal forest.

Using the concept of ecological resilience can thus help to specify the 'ecological criticality' of specific ecosystem types. This approach to ecological criticality is based on the ideas proposed by Serrão *et al.* (1996), Deutsch *et al.* (2003) as well as De Groot *et al*. (2003) and is not suggesting to replace these ideas in any way. Rather, the resilience-approach should be used complementary to the concept of vulnerability (and integrity or ecosystem health) in order to estimate the degree of threat specific ecosystems face. At this point it becomes again apparent that ecological resilience and vulnerability can be used as complementary measures of “stability” (Gallopin 2006). The concept of ecological resilience therefore adds a further criterion to a comprehensive conception of critical natural capital.

5.2.5 Conclusion

This article concludes that the concept of ecological resilience can help a great deal to specify the 'ecological criticality' of specific parts of natural capital (Deutsch *et al.* 2003). The empirical estimation of ecological resilience represents a measure to estimate the degree of threat an ecosystem is exposed to. More specifically, I propose that an ecosystem's degree of ecological resilience is inversely related to its degree of threat. Clearly, the quantification of the degree of threat requires the empirical estimation of ecological resilience. This is possible by means of several methods, such as the threshold approach (Carpenter *et al.* 2001; Peterson *et al.* 2003b), the concept of resilience mechanisms (Nyström 2006), probabilistic resilience and percolation theory (Peterson 2002b) or the concept of maintained system identity (Cumming *et al.* 2005). The concept of ecological resilience may be used complementary to other approaches that mirror the degree of threat ecosystems face, such as integrity or vulnerability (de Groot *et al.* 2003). Thus, the degree of ecological resilience can be used as a further criterion for the criticality of natural capital. This measure does not replace other criteria for criticality, such as the socio-cultural relevance, the economic value or the ecological importance, but rather completes them in order to build a comprehensive conception of critical natural capital.

This article suggests that the estimation of ecological criticality, and thus ecological resilience, is important for the maintenance of valuable ecosystem services and the sustainable use of renewable natural capital (cf. also UNEP 2005b). In a like vein, Mäler (2008) states that the ecological resilience of a system should be regarded as an important capital stock for achieving sustainability, while Aronson *et al.* (2006a; 2006b) assert that dwindling natural capital effectively limits economic growth globally and argue for ecological restoration measures as a counter-strategy. Yet the importance of natural capital for achieving sustainability is not self-evident. The relevance we ascribe to natural capital in general and a system's ecological resilience in particular depends on the specific conception of 'sustainable development' we advocate. Apparently, a conception of strong sustainability would set a higher value on the management of critical natural capital and to concepts such as ecological resilience than a conception of weak sustainability would do (Brand 2005: 22). The resolution of the persisting controversy of weak vs. strong sustainability (Neumayer 1999) is a far off horizon, yet would bring immense theoretical progress to sustainability science and, in my view, ought to be one of the fundamental theoretical foundations of ecological economics. There is no doubt that the ‘discourse-rational choice about the right sustainability conception’ must be based on a variety of criteria (Ott & Döring 2004, 2007; Schultz *et al.* 2008) and that research on ecological resilience can only provide some tentative arguments for the importance of natural capital, and thus in favour of a conception of strong sustainability. Yet in this way, resilience research (Folke 2006; Walker *et al.* 2006) may help to challenge one of the central controversies in sustainability science and to build an important theoretical foundation for ecological economics.

Five years after the CRITINC-research project (Ekins 2003) a clear conceptualization of critical natural capital is hence worth the effort. Such a concept may inform environmental policy and management to identify the natural capital that ought to be preserved in any circumstances in favour of current and future generations. It is thus an important step in our quest for sustainable development.

5.3 Ecological Resilience and Poverty Eradication

This section explores the relevance of resilience research for poverty reduction measures in developing countries. After a short introduction (section 5.3.1), I will shortly sketch my understanding of the environment-poverty nexus (section 5.3.2). Subsequently, I will present a mini-review of the papers published in the last ten years on the resilience-poverty link (section 5.3.3), before I will give some recommendations for future research (section 5.3.4). I will conclude with the findings of this section (section 5.3.5).

5.3.1 Introduction

Poverty reduction within developing countries is one of the grand challenges of sustainability science (Sachs 2005; Cabrera *et al.* 2008). The first goal of the United Nations Millennium Declaration published in 2000 calls for halving extreme poverty and hunger by 2015. Apparently, scientific studies on poverty alleviation published in the last decades have focussed on public policy, investments, institutions or geopolitics and have largely neglected the poverty-environment nexus (Dasgupta 2003a, 2007).

Recently however, the importance of natural capital, defined as any stock of natural resources or environmental assets that provides a flow of useful goods or services, now and in the future (Pearce & Turner 1990; de Groot *et al.* 2003), has been acknowledged by scholars with diverging scientific background. Economists that can be grouped as proponents of weak sustainability argue for the importance of natural capital for achieving poverty alleviation (Sachs & Reid 2006; Dasgupta 2007; Kates & Dasgupta 2007). Likewise, advocates of strong sustainability suggest that natural capital has a strong relevance for well-being and the quality of life (Ott & Döring 2004; Vemuri & Costanza 2006; Costanza *et al.* 2007). In a similar vein, ecologists and environmentally-focussed scholars point to the close link of ecological knowledge, ecosystem services and natural capital for fighting poverty in developing countries (Adams *et al.* 2004; Roe & Elliott 2004; UNEP 2005b; Aronson *et al.* 2006a; DeClerck *et al.* 2006; Mertz *et al.* 2007). The positive effects of managing natural capital for poverty eradication are thus widely established.

As a promising tool for the management of the natural capital, the "resilience approach" has lately been put forward (Gunderson & Holling 2002; Folke 2006; Walker *et al.* 2006). Despite its prominence within sustainability science (Janssen 2007), there has been little systematic effort on the link of resilience research and poverty alleviation in developing countries. To provide an in-depth investigation of this nexus is the aim of this section.

5.3.2 A Note on the Environment-Poverty Nexus

Poverty in developing countries (sometimes termed “extreme poverty”) is generally defined as the condition of a person living on less than one US dollar a day (Dasgupta 2003b). Of course, the idea of a poverty line can be criticized and there is a variety of ideas to conceptualize poverty in terms of indices (UNDP 2003) or capabilities (Sen 1986). Explaining poverty is also complicated, as causes and pathways may include geopolitics and the heritage of colonialism, the failures of development aid, high population growth, poor governance, ineffective institutions, lack of economic growth and the degradation of natural assets (Kates & Dasgupta 2007). Apparently, the sound management of natural capital is only one component among a bundle of strategies for fighting poverty in developing countries. As Lawton (2007) puts it: “ecology alone is not the answer”.

Yet most of the poor countries are “biomass-based economies”, in that the share of agriculture in GDP (i.e. gross domestic product) is about 25% and more than 70% of people live in rural areas. That means, many of the people in poor countries draw their production directly from nature and thus the maintenance of natural capital can have a profound effect on poverty reduction (UNEP 2005b; Dasgupta 2007). For instance, there are close causal linkages between reducing hunger and the sustainable management of natural resources and ecosystems (Adams *et al.* 2004; Roe & Elliott 2004; DeClerck *et al.* 2006). It is important to note however, that the relation of maintaining natural capital and reducing poverty is not straightforward. Studies of the environment-poverty nexus should take into account technical know-how, proper resource pricing and access to resources, but also the deeper socio-political changes (e.g. land reforms) or changes in cultural values (Lélé 1991).

Altogether it has become clear that an innovative approach to environmental management can make a pivotal contribution to the reduction of poverty in developing countries. The next sections explore the “resilience approach” to environmental management and its potential effects for poverty alleviation.

5.3.3 A Mini-Review of Resilience Research for Poverty Reduction

This section reviews the articles published in the last ten years on the topic of resilience research for poverty reduction in developing countries. On the 14th february 2008, I used the Web of Science ISI search engine and the phrase “poverty AND resilience” in the search field “topic”. The literature search resulted in a list of 164 articles, from which 125 articles were discarded, as they refer to psychological studies, and also further 6 articles, because they were either unavailable or only available in Spanish. The remaining 33 papers have not been evaluated in detail but merely scanned for the points listed in *Table 10*.

Table 10: Criteria used to scan papers in mini-review

A	Journal	In which journal the article has been published?
B	Publication Year	In which year the article has been published?
C	Resilience Concept	Which resilience concept(s) are being used in the text?
D	Definition	Is a definition being provided for the term “resilience”?
E	Operationalization	In what way resilience is estimated, indicated or measured?
F	Poverty-link	What is the link to poverty reduction?
G	Management strategies	What are the concrete measures suggested for alleviating poverty?
H	Region	To what region the study refers?

The results of my analysis show that only quite recently there has been a sharp increase in publication efforts on the resilience-poverty nexus. In addition, the data shows that the 33 publications stem from as many as 26 journals. Apparently, the resilience-poverty nexus is examined by a variety of perspectives and scientific disciplines, ranging from the natural sciences (e.g. *International Geology Review, Geoforum, Hydrology and Earth System Sciences*), agricultural science (*Australian Journal of Agricultural Research, Outlook on Agriculture*), health sciences (e.g. *Lancet Infectious Diseases, Transactions of the Royal Society of Tropical Medicine* *and Hygiene*), social sciences (e.g. *Rural Sociology, Annual Review of Anthropology, World Development*), water sciences (*Water Science and Technology*) and journals with a broader sustainability-related scope of readership (e.g. *Ambio, Ecology & Society*).

This comes as no surprise, as "ecological resilience" can be interpreted as a boundary object, i.e. a term that facilitates communication across disciplinary borders (Star & Griesemer 1989), similar to "biodiversity" or "sustainability" (Brand & Jax 2007). Boundary objects can be highly useful as a communication tool in order to bridge scientific disciplines and the gap between science and policy (Eser 2002) (Cash *et al.* 2003). Yet boundary objects may also be a hindrance to scientific progress, as they tend to be used ambiguously or arbitrary and thus to loose specific meaning. The concept of ecological resilience displays both the positive and the negative aspects of boundary objects, as considered in more detail in Brand and Jax (Brand & Jax 2007).[217]

Concerning the specific sub-concepts of resilience, the 33 reviewed papers refer to either social resilience (20), economic resilience (3), ecological resilience (5) or social-ecological resilience (5). Only 15 of the 32 papers explicitly define the term "resilience", whereas 17 articles do not. In addition, only 6 articles propose a resilience proxy for indicating or measuring resilience in real-world systems, whereas 26 articles do not. Also, 15 articles suggest specific management strategies for reducing poverty in developing countries, whereas 17 do not. The message of this is that in many of the analyzed publications the concept of resilience remains vague and is used as a catchword rather than a clearly-defined scientific concept.

From the publications analyzed in the mini-review one can extract the following lessons. First, apparently there is a focus on social resilience and economic resilience, which are sometimes used interchangeable in the analyzed literature. Second, some authors suggest resilience surrogates for indicating or measuring resilience in real-world ecosystems. For instance, Nelson *et al.* (2007) argue for income variability as a surrogate for economic resilience, while Zhang *et al.* (2006) propose the degree of livelihood diversification as a measure for social resilience. Third, several management strategies are being proposed with respect to social resilience and economic resilience. For instance, Berzborn (2007) suggests (*a*) economic diversification, i.e. people diversify and integrate various sources of

[217] Cf. also section 4.1.4.

household income in order to achieve a household portfolio with low covariate risk between its components, (*b*) investment in social networks, mainly based on kinship, but also on reciprocity and neighbourhood, and (*c*) pooling and redistribution of resources as appropriate management strategies to reduce the vulnerability of households to hazards. In a similar vein, de Waal and Whiteside (de Waal & Whiteside 2003) propose kinship networks for assistance as the preferred coping strategy, while Kesavan and Swaminathan (2006) hold that it is important to promote concerted efforts for preserving natural ecosystems and diversifying the households' economies. Fourth, there is a high degree of conceptual vagueness regarding the sub-concepts of resilience. Only occassionally specific definitions, operational steps or management measures are being put forward. In this vein, Andrew *et al.* (2007) state although the concept of resilience is appealing there is a danger it will remain academic or rhetorical and unusable for improving lives of people in developing countries. This conceptual vagueness and linguistic uncertainty (Regan *et al.* 2002) is a great weakness in the literature on the resilience and poverty-linkage, which is again related to the use of resilience as a boundary object (Brand & Jax 2007). In the following section, this article suggests some steps forward in order to achieve a more workable and useful approach to the concept of resilience for applying it to the issue of poverty reduction in developing countries.

5.3.4 Recommendations for Future Work

As the mini-review in the previous section demonstrates, there is a focus on social/ economic resilience in the literature on the resilience-poverty nexus. This is not only true for the early literature: as many as 6 of the 8 articles published in 2007 and 7 of the 11 articles published in 2006 refer to social or economic resilience. There are good reasons for stressing the social and economic dimensions, because it is these dimensions that are of particular importance for the alleviation of poverty in developing countries (cf. section 5.3.2). I argue here, however, there are less good reasons to just use the concept of resilience in order to examine the social or economic dimensions in poverty reduction, because there are well-established alternative scientific concepts that primarily refer to the social and the economic facets, such as vulnerability (Schröter *et al.* 2005; Adger 2006) and the sustainable livelihoods approach (Ellis 2000; Allison & Ellis 2001). Certainly, there is a place for the concepts of social/ economic resilience and for cross-fertilization with other relevant concepts, such as vulnerability, natural hazards research or political ecology (Adger 2006). Yet the point I want to make here is that scholars working on the resilience-poverty nexus should shift their attention to the real innovative approaches that arose out of resilience research, namely "ecological resilience" (cf. chapter 3 and 4.1.3) and "adaptive co-management" (cf. section 3.5.2).

In the following sections, I will illustrate the innovative character of these concepts by applying them to two widespread management situations in developing countries, namely the management of coral reefs and the management of small-scale fisheries.

5.3.4.1 The Management for Ecological Resilience in Coral Reefs

Due to the delivery of essential ecosystem services, such as sea food products, raw materials, coastal protection or recreational possibilities, coral reefs are the cornerstone for economic and societal development throughout tropical coastal areas and thus for millions of people in developing countries (Moberg & Folke 1999; Newton *et al.* 2007). Yet over the last centuries, the impact of human activities has accelerated resulting in serious degradation of at least 30% of world's coral reefs, owing primarily to over-harvesting, pollution, disease and climate change (Jackson *et* *al.* 2001; Hughes *et al.* 2003; Bellwood *et al.* 2004). For instance, as much as 55% of coral reef fisheries in 49 island countries are considered to be unsustainable (Newton *et al.* 2007).

Dependent on combinations of pollution, fishing, climate change and human management, coral reef ecosystems may shift from a state characterized by corals and fish to a variety of alternative states dominated by macro algae, sea urchin or rock (McClanahan *et al.* 2002; Bellwood *et al.* 2004; Hughes *et al.* 2007c). In most management situations, the coral-dominated state is evaluated as the most valuable, due to the maintenance of desired ecosystems services (e.g. fish, raw materials, recreational possibilities). Ecological resilience in this case thus describes "the ability of coral reefs to absorb natural and human-induced disturbance events and still be retained within the same ecosystem state" (Nyström 2006: 30).

Management strategies to maintain the desirable coral reef states range from the establishment of no-take areas and the increase of gear diversity to reforms in property rights, governance systems and markets for reef resources (Bellwood *et al.* 2004; Berkes *et al.* 2006; McClanahan *et al.* 2008). No doubt, in many management situations the wider societal, political and economic context has to be taken into account (Andrew *et al.* 2007), yet this section focusses on the management for the ecological resilience of coral reefs by means of the maintenance of resilience mechanisms.

Step 1 of the resilience mechanisms approach - as considered in section 3.4.2.3.3 - assesses the RCPs that are present in the specific study area. The empirical evidence shows that reef resilience is generally related to the following properties: (a) functional important types of species and their specific life-history traits, such as dominant species (e.g. coral and algae species) or key predators (e.g. sea urchins and herbivorous fish) (McClanahan *et al.* 2002; Bulleri & Benedetti-Cecchi 2006); (b) the amount of ecological redundancy and response diversity within functional effect groups, e.g. in zooxanthellae, coral or herbivore species (Bellwood *et al.* 2003; Elmqvist *et al.* 2003; Micheli & Halpern 2005; Kiflawi *et al.* 2006; Nyström 2006), (c) critical functional groups, such as bioeroding fish species (Bellwood *et al.* 2004; Graham *et al.* 2006; Hughes *et al.* 2007c); (d) source habitats in the vicinity of the disturbed patch that provide external ecological memory (Nyström & Folke 2001; Bengtsson *et al.* 2003); (e) appropriate levels of connectivity within the meta-community including mobile link species (Nyström & Folke 2001; McClanahan *et al.*

2002); and (f) the occurrence of intermediate, pulse disturbances at a small spatial and temporal scale (Nyström *et al.* 2000; Bengtsson *et al.* 2003). *Step 2* requires the quantification of the RCPs occurring in the specific case study area. Finally, *step 3* estimates the amount of ecological resilience by weighing and assessing the resilience-conducive properties and mechanisms with reference to the specific disturbance regime the study region is prone to.

From this approach to reef resilience, several management strategies can be drawn. First, ecological processes that underpin desirable reef states and promote ecological redundancy and response diversity should be identified. A particular focus should be placed on functional important types of species and critical functional groups, e.g. bioeroding fish species, rather than on increasing species richness per se (Hughes *et al.* 2005; Nyström 2006). Second, a sound approach to reef management requires the establishment of a matrix of reserves and no-take areas. These regions provide the source habitats for the renewal and reorganization of disturbed sites nearby (Bengtsson *et al.* 2003; Bellwood *et al.* 2004; Hughes *et al.* 2007a). Third, it is essential to find the appropriate degree of connectivity within the coral reef meta-community. On the one side, too low a dispersal rate means that both stochastic extinctions and negative interactions (e.g. competitive exclusion) cause local populations to become extinct without rescue. On the other side, at too high rates dominant competitors are introduced into all local communities and, thus, spatial variation in fitness is homogenized by immigration resulting in reduced local and regional diversity (biotic impoverishment) (Leibold *et al.* 2004; Cadotte 2006; Mouquet *et al.* 2006). Furthermore, in highly connected meta-communities diseases, pests, fires and some invasive species can be spread more easily (van Nes & Scheffer 2004). The aim of these management strategies is to increase the ecological resilience within the coral reef meta-community. Management for reef resilience can help to secure the living base of millions of people in developing countries.

5.3.4.2 Adaptive Co-Management of Small-Scale Fisheries

Resilience theory channels in a specific and contentious approach to environmental management termed “adaptive co-management” (Berkes *et al.* 2003; Olsson *et al.* 2004; Armitage *et al.* 2008) (cf. section 3.5.2). Only few studies have considered the relationship between adaptive co-management and poverty reduction (e.g. Granek & Brown 2005; Wilson *et al.* 2006; Andrew *et al.* 2007; Fennell *et al.* 2008). This section focusses on the adaptive co-management of fisheries in developing countries.

Fisheries make important contributions to national and regional economies and to the food security of people in developing countries. For instance, as many as 200 million people may depend on small-scale fisheries and related activities (Andrew *et al.* 2007). Yet 75% of harvested fish populations that are monitored are already overexploited or will become so without stringent management interventions (Balmford & Bond 2005). Global threats of fish resources thus warrant consideration of an innovative management approach.

According to Nielsen *et al.* (2004) and Wilson *et al.* (2006), adaptive co-management of fisheries in developing countries must be able to address the problems facing the fisheries communities in the present situation including (*a*) the risk of exclusion from resources and markets due to globalisation, competing uses of the freshwater and coastal environment and other activities which may lead to reduced resource productivity; (*b*) provision of an institutional framework to control access and resolve questions of distribution of access between fishers; (*c*) reverting overexploitation to sustainable use of the living aquatic fonds on which the fishing communities rely; and (*d*) reconciling the immediate needs of fishing communities with international agreements focussing on the aquatic ecosystems. Threats to small-scale fisheries may be largely internal, such as overfishing and pollution, or primarily external, such as competition with industrial fisheries, trends in world markets, fuel costs or climate change. Adaptive co-management must therefore take into account both the ecological characteristics of the fishery domain and the external political, economic and institutional drivers. Also, there is a need for a diagnosis tool for the re-evaluation of small-scale fisheries in terms of recognizing opportunities and threats and from them, suitable entry points for management (Andrew *et al.* 2007).

Efforts to foster the collaborative character of governance include the establishment of cross-scale linkages that bring together groups and polities with broad local foci with NGOs and government agencies with specific trans-local mandates, and the maintenance of a vibrant civil society. In addition, the integration of fisher's knowledge and practices already in early stages of the management process (e.g. in the phase of setting objectives) is a pivotal objective. Yet there is also an important

role for the state in the provision of legal and policy support to settle disputes between competing resource users (Nielsen *et al.* 2004; Wilson *et al.* 2006).
According to the theoretical framework of *active adaptive management* (cf. section 3.5.2.1), policy choices are treated as large-scale experiments stimulating an iterate learning process. The aim is to collaborative plan, monitor and evaluate outcomes, and to change management in response to those evaluations (Andrew *et al.* 2007), which includes the use of suitable and easy-to use models to scientifically aid decisions (Lachica-Alino *et al.* 2006).
In addition, any tool proposed within the theoretical framework of “adaptive co-management to promote social-ecological resilience within social-ecological systems” may be applied (cf. section 3.5.2.4). This includes management measures in several dimensions. In the “ecological” dimension, environmental managers strive for ecological knowledge, biodiversity, small-scale disturbances and back-loop management (Berkes & Folke 1998; Berkes & Holling 2002). In the social and institutional dimension, managers foster visionary leadership, trust, legislation that creates social space for ecosystem management, funds for responding to environmental change, information flow and social network building, arenas of collaborative learning, polycentric institutional arrangements and bridging organizations, to name but a few of the suggestions proposed by Olsson *et al.* (2004) and Folke *et al.* (2005).[218] It has become clear that adaptive co-management of small-scale fisheries in developing countries is an interdisciplinary undertaking.
According to Walters (2007), implementing sound measures of active adaptive management requires expanded monitoring measures, the political will by decision makers to admit and embrace uncertainty in making policy choices, and leadership to do all the hard work needed to plan and implement new and complex management programs. To sum up: using the adaptive co-management approach in small-scale fisheries in developing countries may provide an innovative tool to maintain food security and reduce poverty in developing countries.

[218] Note that there is a high degree of linguistic vagueness in each concept of “social-ecological resilience”, “social-ecological system” and “adaptive co-management”. Therefore, the suggestions made for the social dimension appear as a mere unsystematic listing (cf. section 3.5.3).

5.3.5 Conclusions

The study of poverty reduction in developing countries is in its essence an interdisciplinary undertaking. Profitable strategies to fight poverty range from the containment of population growth, the stimulation of economic growth or the improvement of governance and institutions to the maintenance and effective management of environmental assets (Kates & Dasgupta 2007). Yet as most of the people in poor countries depend directly on the natural capital (UNEP 2005b; Dasgupta 2007), innovative approaches to environmental management warrant particular consideration.

This section suggests the resilience approach as an innovative tool for the management of the natural capital and for reducing poverty in developing countries. The mini-review of the articles published in the last decade on the resilience-poverty nexus shows that scholars have largely focussed on social or economic resilience. In addition, the literature has used the term “resilience” in a rather vague and indiscriminate way. This section does not neglect the relevance of the concepts of social or economic resilience, but suggests to shift the focus on the real innovative concepts within resilience research: “ecological resilience” and “adaptive co-management” (Folke 2006). Ecological resilience represents a well-defined concept that can be (indirectly) estimated and can stimulate sound conservation efforts. The adaptive co-management approach tries to combine learning processes with some form of sharing of power and responsibility between stakeholders (Hughes *et al.* 2007b; Armitage *et al.* 2008). Each of the concepts can be applied to environmental management situations in developing countries for reducing poverty by means of the maintenance of ecosystem services.

Resilience research does not represent the one-fits-all solution, but rather complements alternative lines of research on vulnerability, adaptation, the sustainable livelihoods approach or political ecology (Adger 2006). A critical issue then pertains to the question about which scientific concept is appropriate in a particular management situation. When threats are largely internal referring to the ecosystem as such, the concepts of ecological resilience or vulnerability would be appropriate solutions. By contrary, when threats refer to the wider political, economic and institutional context of the ecosystem, the adaptive co-management approach, political ecology or the sustainable livelihoods approach may be more adequate.

Andrew *et al.* (2007) rightly suggest that we need a diagnosis framework to decide about suitable entry points for management. Undertaking a diagnosis signals a fundamental re-evaluation of the environmental management situation for assessing the relative importance of different opportunities, strengths and threats (Andrew *et al.* 2007). This may help to orient place-based studies on particular poverty issues in developing countries.

The reduction of poverty in developing countries is a multi-facet task (Sachs 2005; Kates & Dasgupta 2007). Resilience research represents a small but important step in our quest for a just and sustainable world.

6 Summary: Conclusions and Prospects

This thesis entitled "Resilience and Sustainable Development: an Ecological Inquiry" examines the concept of ecological resilience and its relation to a specific conception of sustainable development. Primary objectives of this thesis are: (i) to describe and analyze the fundamental theoretical framework of resilience theory; (ii) to understand, critizice and question the theoretical framework of resilience theory; and (iii) to investigate the relevance of resilience theory with respect to achieving sustainable development (cf. section 2.1).

In order to meet these objectives I carried out conceptual work *sensu* Laudan (1972), which includes literature research, solving and clarifying linguistic uncertainties *sensu* Regan *et al.* (2002), using specific criteria for the usefulness of terms *sensu* Jax (2002: 14), using several fundamental insights within ecological science, applying the concept of a boundary object used within Science & Technology Studies *sensu* Star and Griesemer (1989) and finally using a cultural position within philosophy of science, following Kirchhoff (2007) (cf. section 2.2).

This thesis does not refer to ecological science only but can rather be subsumed under "social-ecological research" or "sustainability science", understood as the science of the relationships of humans to their specific natural and societal environment (Becker 2003). This thesis corresponds to use-inspired research within sustainability science *sensu* Clark (2007) that is oriented at the concept of interdisciplinarity *sensu* Lélé and Norgaard (2005) and conducts conceptual research *sensu* Laudan (1977) with the aim to generate societal knowledge.

There are some limitations to this thesis. The first limitation is related to the pure extent of the literature on the topic of resilience. Sometimes the discussion of particular topics, e.g. the biodiversity-ecosystem functioning debate, refers to such a large amount of literature that it is impossible to include every paper and every book on the subject. In these cases, I was dependent on review papers, databases or my own choice of relevant literature.

The second shortcoming is related to resilience research *per se*. Resilience research includes the concept of ecological resilience (applied to ecological systems) but focuses on the concept of social-ecological resilience (applied to social-ecological systems). In this thesis however, I regard the concept of social-ecological resilience as conceptually and theoretically weak (confer section 4.1.5). I therefore focused on the concept of ecological resilience, and only included examinations of social-ecological resilience where appropriate. This may be a certain shortcoming of this thesis.

Third, the topic of resilience and sustainable development is essentially interdisciplinary, as several disciplines are involved (e.g. ecology, economics, sociology, political science) (confer section 2.2). Yet my competence stems largely from biology and ecology. This thesis is thus limited to my competence in the natural sciences, yet I tried to include insights from other disciplines when appropriate. The important point here is that such a topic should ideally be examined by a research team, comprised of scholars from different areas of scholarship.

Fourth, resilience research advocates an ontological, holistic view of ecosystems (cf. section 4.2.3.1). Even though I assume that it is *methodologically* useful to examine the ecological resilience of ecosystems, I do not champion a particular position with respect to (*a*) the discussion whether or not ecological units, such as ecosystems, exist ontologically, and (*b*) the debate between holistic and individualistic systems notions (confer section 2.4). This separation of method and ontology may also be a certain shortcoming of this thesis.

In the following, this chapter first offers a summary and conclusion of the previous chapters (section 6.1 to section 6.7). Subsequently, I will present some prospects for the orientation of further research (section 6.8).

6.1 A 'Short History' of Resilience

The term "resilience" has been used in two fundamentally different ways in the ecological literature. The two meanings refer to two opposing research traditions, which constitute what is termed the alternative-stable-state-controversy (Gunderson 2000). On the one hand, the equilibrium paradigm states that for any system there is only one equilibrium or steady state and that any system returns to this equilibrium after disturbance. Resilience is hereby defined as the time required for a system to return to an equilibrium point following a disturbance event (Pimm 1984). This meaning has been coined "engineering resilience" by Holling (1996) and is largely identical to the stability property "elasticity" (Grimm & Wissel 1997).

On the other hand, a particular type of the non-equilibrium paradigm holds that ecosystems can exhibit alternative stable states or alternative basins of attraction (Holling 1973; Gunderson 2000; Walker *et al.* 2004). There exists only local stability and when disturbed a system may shift to another basin of attraction, which is characterized by a different structure and different processes. Resilience is then dubbed *ecological resilience* or *ecosystem resilience* and defined as *the capacity of an ecosystem to resist disturbance and still maintain a specified state* and this is the meaning of resilience this thesis is mainly concerned with. In this interpretation, the concept comes close to the stability concepts "resistance" (Grimm & Wissel 1997) or "robustness" (Hansson & Helgesson 2003). This definition may thus underestimate other important characteristics of ecological resilience, such as the capacities for renewal, reorganization and development, which are also often considered a major characteristic of this kind of resilience (Folke 2006). Yet this definition is in my view useful and workable to be used for the measurement of ecological resilience in real-world ecosystems (cf. section 3.1 and section 4.1.3).

The term "ecological resilience" has been introduced by Holling (1973) as a stability concept for ecological systems. During the 1990s the concept gained high momentum and was viewed as one of the most important research topics of sustainability science. Since then it has been adopted by numerous scientific disciplines, e.g. sociology, economics, political science, environmental planning and ethics (Folke 2006; Brand & Jax 2007). Today, the concept is one of the rising stars within ecology, sustainability science and environmental management (Janssen 2007).

In this thesis we[219] have carried out an analysis of the various meanings the term "ecological resilience" represents in relevant literature. We identified at least eleven[220] levels of meaning ordered according to their specific degree of normativity (cf. section 4.1.2). In our further analysis we focus on two subconceptions of resilience which we figure to be the central concepts of resilience research. The first concept is ecological resilience, which is applied solely to ecological systems, such as forests, savannahs or lakes. The second concept is termed "social-ecological resilience" and defined as "the capacity of social-ecological systems to absorb recurrent disturbances (…) so as to retain essential structures, processes and feedbacks" (Adger *et al.* 2005: 1036). This concept is a tool for assessing "social-ecological systems", i.e. systems in which human and natural components interact reciprocally across diverse organizational levels (Liu *et al.* 2007).

[219] This analysis is the result of the collaboration with Kurt Jax, München and Leipzig (cf. Brand & Jax 2007).
[220] In Brand and Jax (2007) we identified ten levels of meaning, yet recent literature added an eleventh.

6.2 Evaluating the 'Background Theory' of Resilience

The concept of ecological resilience is based on at least three related concepts, which together build its background theory (cf. section 3.2).

First, resilience research presupposes a notion of ecosystem structure and dynamics, which is dubbed "panarchy" (cf. section 3.2.1) This "meta-model" (Cumming & Collier 2005) of ecosystem dynamics consists of four-phase adaptive cycles (r-, K-, Ω- and α-phases) that occur on each level of a system's hierarchy (Holling 2001; Gunderson & Holling 2002). Resilience research assumes that this notion holds true across all ecosystem types (and even across social and economic systems).

However, the adaptive cycle is merely one meta-model of ecosystem dynamics among many. Although the adaptive cycle offers a persuasive approach to characterizing and understanding system dynamics, it is only one of a set of possible meta-models that might explain or clarify different aspects of system behaviour. Alternative meta-models are represented by replacement, e.g. lotic ecosystems or volcanic eruptions on oceanic islands, succession (Weidemann & Koehler 1997) or dynamic limitation (Cumming & Collier 2005). The important point is that the adaptive cycle does not apply to all situations and is not an useful metaphor for all system dynamics. For instance, there may be no release phase, as for instance the transition from a bog to a forest, or no conservation phase if external disturbances are intensive and occur early in the development stage. For these systems it makes no sense to speak of an adaptive cycle (Walker *et al.* 2006).

Also, the notion of ecosystem dynamics proposed by resilience research is based on a holistic and organic systems notion. There has been an ongoing controversy between proponents of rather holistic positions and proponents of rather individualistic positions for several decades (Trepl 1987; Kirchhoff 2007; Voigt 2008). Therefore a holistic systems notion cannot regarded as mainstream position in ecology. This makes the assumption of an organic systems notion rather weak.

The second assumption of resilience research refers to the concept of key variables (cf. section 3.2.2). According to the "rule of hand", important changes in ecosystem dynamics can be understood by analyzing a few, typically no more than five, key variables. Only a few variables are ever dominant in observed system dynamics. To understand change in ecosystems, it is important to identify this small set (Yorque *et al.* 2002; Walker *et al.* 2006). These key variables may be mainly driven by keystone process species, while the remaining species exist in the niches formed by these keystone process species (Folke *et al.* 1996). The distribution of lumps and gaps in species body masses is viewed as empirical evidence for the existence of key variables operating at distinct speeds and scales (Allen & Holling 2002).

Yet this notion of key variables proposed by resilience research has shortcomings. First, it uses a top-down approach to ecological modelling, which can be critiziced as being partial and ought to be complemented by bottom-up approaches, such as individual-based models (Grimm 1999). Second, the notion of key variables is only one among a variety of approaches to understanding the "functioning" of ecosystems (in the sense of how the whole is sustained) (Jax 2005). Several other approaches can be found in the relevant literature, for example referring to basic ecosystem processes (Breymeyer 1981) or ecosystem services (de Groot *et al.* 2002). Another sound approach to ecosystem functioning is provided by the methodology of self-identity (Jax *et al.* 1998; Jax 2006). Hence, the concept of key variables is at least contentious.

The third assumption of resilience "theory" is the concept of alternative stable states (cf. section 3.2.3). The concept holds that ecosystems can exhibit alternative stable regimes (or alternative basins of attraction). For example, a shallow depth lake appears to exhibit two regimes with respect to nutrient load. In the oligotrophic or clear-water regime, the water has a low biomass of phytoplankton and low recycling rates of nutrients from sediment to water. In the eutrophic or turbid-water regime, phytoplankton biomass is high, often forming noxious blooms, and recycling of nutrients from sediment to water is rapid (Dent *et al.* 2002). Numerous observational studies show that alternative basins of attraction are common for many ecosystem types, such as temperate lakes, tropical lakes, wetlands, estuaries and coastal seas, coral reefs, kelp forests, pelagic marine fisheries, savannas, woodlands, deserts, forests, arctic tundra and oceans (Carpenter 2001; Scheffer *et al.* 2001; Folke *et al.* 2004), yet experimental evidence is not as clear. Only 13 out of 21 conclusive experiments (62%) found evidence for the existence of alternative stable regimes (Schröder *et al.* 2005). In addition, some observational studies show the lack of alternative stable regimes. For example, Sim *et al.* (2006) argue that a concept of alternative stable regimes is not appropriate for understanding the dynamics of seasonal drying wetlands in South-Western Australia. In a recent review of the empirical evidence, Didham *et al.* (2005) held that the overwhelming majority of cases in which alternative stable regimes have been detected come from systems that were historically subject to moderate abiotic regimes, for example wetlands, streams, deserts, arid grasslands, rangeland, woodland, savannas, salt marshes or intertidal mud flats. All this indicates that the existence of alternative stable regimes is no ecological law nor rule; many but not all ecosystem types may exhibit alternative stable states. This weakens the assumption that alternative stable states exist, as proposed in resilience research.

In addition, resilience research suggests that "sudden" regime shifts from one regime to the other may occur. There is evidence for abrupt regime shifts in response to gradual environmental change (i.e. for non-linear regime shifts) with respect to models (van Nes & Scheffer 2005), observations (Carpenter 2001; Scheffer *et al.* 2001; Folke *et al.* 2004) and experiments (Schröder *et al.* 2005). At a critical breakpoint a regime shift alters feedback mechanisms (Walker & Meyers 2004; Briske *et al.* 2006) causing the ecosystem to reorganize around another set of controlling variables and processes. The critical breakpoint in state space between two regimes of a system, i.e. the critical values of the variables around which the system shifts from one stable regime to the other, is termed ecological threshold (Muradian 2001; Walker & Meyers 2004).

It is possible to predict the position of such a threshold by means of several methods. Wissel (1984) suggests that the characteristic return time of a system will increase when a slow controlling variable approaches an ecological threshold. This "universal law" may be used to predict the position of an ecological threshold by extrapolation of empirical data, which is recorded at safe distance from this threshold. Carpenter and Brock (2006) use Monte-Carlo simulation and show that the standard deviations (or variance) of fast variables should increase in the vicinity of an ecological threshold. This feature may indicate a general characteristic of ecosystems, in that they "stutter" before a regime shift occurs. Finally, Fath *et al.* (2003) and Mayer *et al.* (2006) refer to Fisher Information, which represents a statistical measure of indeterminacy and can be interpreted as a measure of the variability in the time the system state spends in the various sections of its steady state trajectory (Mayer *et al.* 2006). Hereby, shifts between alternative regimes constitute periods of high variability and Fisher Theory is able to identify such transitions between regimes in datasets, which are characterized by a great deal of noise.

Altogether it has become clear that resilience research is based on three concepts: (i) a notion of ecosystem dynamics, termed adaptive cycle and panarchy; (ii) the concept of key variables; and (iii) the concepts of alternative stable states and ecological thresholds. Each assumption is contentious, as there are many empirical situations in which it does not hold. Using the classification of theories proposed by Pickett *et al*. (1994: 85ff), resilience theory does not exist in a confirmed stage of theory development but rather in a *consolidating* and *empirical-interactive* stage.

6.3 A List of Resilience Mechanisms

The term “resilience mechanisms” points to distinct properties and mechanisms that have causal or nomic (Cooper 1998) force in creating ecological resilience at the ecosystem level (cf. section 3.3). Whenever there is talk about ecological “stability”, two major debates in ecological science have to be taken into account. These are: (1) the classical *diversity-stability debate* and (2) the modern *biodiversity-ecosystem functioning debate* (BDEF). In this thesis I try to unify these fundamental debates with the literature on resilience mechanisms.

Lessons on the diversity-stability- and the biodiversity-ecosystem functioning-debates include the following topics. First, most of classical equilibrium approaches based on deterministic systems may be inadequate to understand stability properties, such as ecological resilience at the ecosystem level (Loreau *et al.* 2001). They are replaced by a non-equilibrium ecology that takes the relevance of alternative basins of attraction, disturbance, historical contingency, adaptation at the ecosystem-level, openness as well as heterogeneity into account (Wallington *et al.* 2005).

Second, there is no universal relationship of “biodiversity” and “stability” (Trepl 1999; Schwartz *et al.* 2000; Hooper *et al.* 2005; Thompson & Starzomski 2007). Both terms (“biodiversity” as well as “stability”) are slippery, as there is a variety of concepts for both biodiversity (e.g. species richness, functional group richness, regional diversity) and stability (e.g. persistence, resistance, resilience). In addition, the relationship is only valid for a particular ecological situation (for example, the level of description, the variables of interest, temporal and spatial scale). Moreover, results from one ecosystem do not necessarily inform us about other types of systems (Bengtsson *et al.* 2002; Symstad *et al.* 2003). This creates a large matrix of potential combinations of various facets of “biodiversity”, distinct stability concepts and a potentially infinite number of ecological situations. Most studies in BDEF research have liberally extrapolated evidence from individual studies to the role of species richness in general (Johnson *et al.* 1996). Yet different types of ecosystems may require different kinds of concepts, measures and experimental design (Loreau *et al.* 2002). As Trepl (1999) points out there is not a lack of data but a lack of knowledge about the power of explanation or applicability of this data. The potential domain of resiliency (Cooper 1998) of the relationship between “biodiversity” and stability concepts such as ecological resilience, that means the cases for which the relationship is valid, its degree of generalization, is in my view, one of the most interesting and pressing questions within BDEF research.

Third, species richness is only one of the facets of biodiversity that affect ecosystem processes and “stability”. In fact, other components can be more important, e.g. functionally important types of species (e.g. keystone species, ecological engineers, ecologically dominant species), species composition, functional diversity and (the number of) functional groups. A modern approach takes into account the relationship between the adequate composition of functional attributes and ecosystem processes and “stability” (Grime 1997b; Bengtsson 1998; Loreau 1998; McCann 2000; Hooper *et al.* 2005; Srivastava & Vellend 2005).

Fourth, most studies have been performed under closed and unrealistic conditions at a small spatial and temporal scale (Schwartz *et al.* 2000; Loreau *et al.* 2001; Diaz *et al.* 2003). Many experiments used artificial species assemblages closed to immigration and propagule supply and containing a single trophic level. It is not at all clear if such studies can be used to inform the public and policymakers about the large-scale consequences of a loss in biodiversity and global change (Bengtsson *et al.* 2002). In contrast, a modern approach evaluates the effects of biodiversity in a more natural, open and realistic setting on a large temporal and spatial scale (Bengtsson 2002; Bengtsson *et al.* 2002; Diaz *et al.* 2003; Naeem & Wright 2003; Symstad *et al.* 2003; Cardinale *et al.* 2004; Srivastava & Vellend 2005; France & Duffy 2006; Matthiessen & Hillebrand 2006).

A modern approach uses (*a*) a more natural and open setting and includes multiple trophic levels and multitrophic interactions (Schwartz *et al.* 2000; Ives *et al.* 2005; Thebault & Loreau 2006) as well as dispersal, invasion and colonization events (Diaz *et al.* 2003; Naeem & Wright 2003; Symstad *et al.* 2003; Cardinale *et al.* 2004; Srivastava & Vellend 2005; France & Duffy 2006; Matthiessen & Hillebrand 2006).

A modern approach incorporates (*b*) realistic scenarios of (human-caused) disturbance and species loss or addition. This means, real world scenarios of human-caused environmental changes through habitat destruction, overexploitation of biological resources, pollution and climate change (Sala *et al.* 2000), and realistic scenarios of species loss (Solan *et al.* 2004; Zavaleta & Hulvey 2004; Naeem 2006) or species additions (Sax & Gaines 2003; Srivastava & Vellend 2005) need to be taken into account. This implies research on assembly or disassembly rules (Weiher & Keddy 1999; Ostfeld & LoGiudice 2003; Solan *et al.* 2004). For example, the effects of species loss or addition strongly depend on the trophic position of species gained or lost because consumers for example have particular effects on population control and "stability". Changes in species richness at the consumer trophic level alone have very different effects than simultaneous changes at both plant and herbivore trophic levels do (Thebault & Loreau 2006). It is also crucial to acknowledge that at local and regional scales species addition is in fact more likely to occur than species loss (Sax & Gaines 2003). Such "enlightened scenarios" contribute to ecological realism within BDEF research (Srivastava & Vellend 2005). Finally, a modern approach strives for (*c*) evidence on large temporal and spatial scales. Most research on stability properties has been performed on small spatial and short temporal scales. It is not at all clear if such studies can be used to inform the public and policymakers about the large-scale consequences of biodiversity loss and global change. When the scale being investigated changes, community and ecosystem properties do not necessarily change in any coherent fashion (Bengtsson 2002; Bengtsson *et al.* 2002; Cardinale *et al.* 2004). Therefore, evidence on a landscape and long-term scale is critical for the application of ecological resilience to conservation and environmental management issues (Naeem & Wright 2003; Symstad *et al.* 2003; France & Duffy 2006).

Taking these insights into account, this thesis suggest a list of resilience mechanisms. These are: (1) population diversity, which includes (a) population richness, i.e. the number of populations of a species in a given area, (b) population size, i.e. the number of individuals per population, (c) spatial distribution of the population under investigation, i.e. the extent of the populations relative to their maximum in a defined area as well as population dispersion and (d) genetic diversity within and among populations (Luck *et al.* 2003; Reusch *et al.* 2005); (2) the maintenance of functional important types of species and their specific life-history traits, such as real/ potential drivers, keystone process species, keystone species, ecological engineers, umbrella species or dominant species (Grimm *et al.* 1999; Fahrig 2001; Ernest & Brown 2001; Hooper *et al.* 2002; Folke *et al.* 2004; Hooper *et al.* 2005; Bulleri & Benedetti-Cecchi 2006); (3) the amount of ecological redundancy, i.e. species that perform the same ecosystem process, and response diversity, i.e. ecologically redundant species showing different responses to disturbances, within functional effect groups (Woodward 1994; Tilman 1996; Peterson *et al.* 1998; Trepl 1999; Ives *et al.* 2000; McCann 2000; Hughes *et al.* 2002; Hooper *et al.* 2002;

Elmqvist *et al.* 2003; Loreau *et al.* 2003; Symstad *et al.* 2003; Hooper *et al.* 2005; Srivastava & Vellend 2005; Nyström 2006); (4) the maintenance of critical functional groups, i.e. those functional effect groups that have low ecological redundancy and whose lost results in a dynamic regime shift (Mooney *et al.* 1996; Bellwood *et al.* 2004; Micheli & Halpern 2005); (5) local functional compensation, i.e. the tendency of coexisting, competing species within a patch to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition (Ernest & Brown 2001; Vinebrooke *et al.* 2004); (6) the concentrations of biological legacies within the disturbed area, i.e. organisms and organic structures that survive a disturbance event and serve as foci for regeneration and allow species to colonize (Turner *et al.* 1998; Franklin & MacMahon 2000; Elmqvist *et al.* 2003); (7) the existence of source habitats in the vicinity of the disturbed patch (Loreau & Mouquet 1999; Moberg & Folke 1999; Nyström *et al.* 2000; Nyström & Folke 2001; Mouquet & Loreau 2002; McClanahan *et al.* 2002; Bengtsson *et al.* 2003; Elmqvist *et al.* 2004; Hughes *et al.* 2005; Cadotte 2006); (8) appropriate levels of connectivity within the meta-community, i.e. (a) small to moderate distances and (b) intermediate (but not high) dispersal rates between habitat patches (Fahrig 2002, 2003; Loreau *et al.* 2003; Leibold *et al.* 2005; Cadotte 2006); (9) an intermediate degree of regional heterogeneity (Mouquet *et al.* 2006); (10) regional functional compensation, i.e. the tendency of coexisting, competing species at a landscape scale to increase in abundance should other species decrease or get lost, so that ecosystem-level processes are more stable than would be expected from random shifts in species composition (Ernest & Brown 2001; Vinebrooke *et al.* 2004); (11) the occurrence of intermediate, pulse disturbances at a small spatial and temporal scale (Gunderson & Holling 2002; Bengtsson *et al.* 2003; Shea *et al.* 2004); and (12) the value of controlling abiotic variables (Carpenter *et al.* 2002; Jansson & Jansson 2002; Gunderson & Walters 2002; Walker & Abel 2002).

What I want to show with this list is that no single mechanism is *the* cause for ecological resilience, rather it is the interplay between the variety of mechanisms present at different levels of description that is responsible for the emergence of ecological resilience at the ecosystem level. These mechanisms may be regarded as “(biophysical) components” of ecological resilience (cf. Walker *et al.* 2002).

6.4 Operationalization – No More Than Indication

The question whether it is possible to operationalize, i.e. measure, estimate or indicate ecological resilience has been posed by many authors within the discourse on sustainability (e.g. Ott 2001; Kopfmüller *et al.* 2001). My answer is short but vague: ecological resilience is hard to measure (confer section 3.4). This may be due to the specific systems notion of resilience research. As considered in section 4.2.3.1, the resilience approach champions a holistic and organic systems notion. An ecosystem is viewed as a real entity (i.e. ontological realism) and to be similar to an organism. That means, ecosystems are viewed as "delimited wholes" that self-organize by means of the interactions between their components (e.g. species). Yet it is highly contested whether ecosystems can be viewed in analogy to an organism, as there is a long-standing controversy regarding holistic and individualistic systems notions in ecology (confer section 2.4) (Trepl 1987; Jax 2002, 2006; Kirchoff 2007; Voigt 2008).

It is my impression that the discussion on the operationalization of ecological resilience in a sense unconsciously reflects this controversy. Scholars within resilience research realize a tension between a concept of general resilience and a concept of specified resilience. *General resilience* refers to a rather holistic systems notion and is defined as "the general capacity of a system that allow it to absorb unforeseen disturbances". In contrast, *specified* (or *targeted*) *resilience* refers to a rather individualistic systems notion and is understood as "the resilience of a specified system to a specified disturbance regime" (Walker & Salt 2006: 120f). Yet to my knowledge the resilience approach does not explicitly take into account the enduring controversy between holistic and individualistic systems notions in ecology.

In resilience research, the way out of this tension between holistic and individualistic systems notions is to focus on the empirical indication of ecological resilience by means of resilience surrogates that focus on specific components of the ecosystem. Resilience scholars realize that ecological resilience – understood as an "emergent property" of "ecosystems" – is elusive. Consequently, recent work proposes that it is hard to measure ecological resilience directly and suggests the use of proxies derived from theory as a tool for assessing resilience. These proxies have been termed *resilience surrogates* by Carpenter *et al.* (2005) and Bennett *et al.* (2005a). The important point here is that these resilience proxies do not correspond to a real holistic measure of ecological resilience. Rather than “measurement”, it is therefore more appropriate to speak of the *indication* of ecological resilience.

Despite these problems, it is possible to indicate ecological resilience by means of a comprehensive resilience analysis, which specifies what systems ought to be resilient to exactly what type of disturbances (cf. section 3.4.2). As to the to-what part, it turns out that the focus of the concept of ecological resilience are compounded and/ or synergistic press and/ or large-infrequent disturbances (cf. section 3.4.2.2). Resilience surrogates are then estimated by means of slow variables and ecological thresholds (Peterson *et al.* 2003; Bennett *et al.* 2005), the concept of maintained system identity (Cumming *et al.* 2005), landscape models and cross-scale edge (Peterson 2002), discontinuities and functional groups (Allen *et al.* 2005; Stow *et al.* 2007), costs of restoration (Martin 2004), vegetation indices and digital remote sensing (Washington-Allen *et al.* 2008) and resilience mechanisms (Nyström 2006). As an example, I have applied the surrogate of resilience mechanisms to coral reef ecosystems in section 5.3.4.1.

6.5 A Promising Approach to Environmental Management

Resilience research proposes a specific approach to environmental management termed "adaptive co-management" (confer section 3.5.2) (Berkes *et al.* 2003; Olsson *et al.* 2004; Armitage *et al.* 2008). Adaptive co-management tries to combine learning processes with some form of sharing of power and responsibility between stakeholders. There are certain weaknesses to this approach, as it is based on the rather vague concept of social-ecological resilience and is based on general systems theory (confer section 3.5.2.2 and section 3.5.2.3). Also, it has been widely misunderstood in recent literature (Gregory *et al.* 2006). Nevertheless I consider adaptive co-management as an innovative approach for dealing with problems characterized by high levels of uncertainty. Adaptive co-management combines insights from a variety of different disciplines, such as ecology, sociology and political science. The result is a flexible system of resource management, tailored to specific places and situations, supported by, and working in conjunction with various organizations on different levels (Armitage *et al.* 2008). I have applied the adaptive co-management approach to the topic of fisheries in developing countries in section 5.3.4.2.

6.6 Challenging Theoretical and Conceptual Problems

The resilience approach is in jeopardy due to several conceptual and theoretical problems.

Conceptual Vagueness

The first problem is the immense vagueness of the term "resilience" (cf. section 4.1.4). As I have considered above, resilience has been introduced in the 1970s as a relatively clearly defined ecological stability concept, but has been adopted since then by numerous disciplines, such as economics (Farber 1995; Batabyal 1998; Perrings & Stern 2000; Brock *et al.* 2002; Perrings 2006), political science (Olsson *et al.* 2006), sociology (Adger 2000) or planning (Pickett *et al.* 2004). In particular, resilience is increasingly interpreted in a broader sense across disciplines as a way of thinking, as a perspective or even a paradigm for analyzing social-ecological systems (Folke *et al.* 2002; Folke 2003; Anderies *et al.* 2006; Folke 2006; Walker *et al.* 2006). As a result, the original, ecological concept of resilience first defined by Holling (1973) has been transformed considerably. This becomes apparent in several points.

First, the specific meaning of resilience gets diluted and increasingly unclear. This is due to the use of the concept (a) with many different intensions and (b) with a very wide extension, as in Hughes *et al.* (2005) for instance. Second, a broad concept of resilience often includes normative dimensions. Third, the term "resilience" is used ambiguously as divergent conceptions of resilience are proposed. Fourth, the original ecological dimension of resilience is vanishing. Recent literature increaslingly stress the social, political and institutional dimensions of resilience (e.g. Folke 2002; Olsson *et al.* 2004, 2006) or address whole social-ecological systems (e.g. Adger *et al.* 2005; Hughes *et al.* 2005; Folke 2006; Walker *et al.* 2006), while genuinely ecological studies of resilience are becoming rare (but cf. Bellwood *et al.* 2004; Nyström 2006, 2008). Fifth and finally, resilience is increasingly conceived as a perspective, rather than a clear and well-defined concept. According to Anderies *et al.* (2006) "resilience" is better described as a collection of ideas about how to interpret complex systems. As a result, the meaning of resilience becomes increasingly vague and unspecified.

This thesis suggests that the increased vagueness, inherent normativity and ambiguousness of resilience is due to its use as a boundary object. Within the field of

Science & Technology Studies, it signifies a term that facilitates communication across disciplinary borders by creating shared vocabulary although the understanding of the parties would differ regarding the precise meaning of the term in question (Star & Griesemer 1989). Boundary objects (e.g. “biodiversity” or “sustainability”) my have very positive effects. Boundary objects can be highly useful as communication tools for bridging the gap between science and policy as well as for combining scientific disciplines, provided they are open to interpretation and are valuable for various scientific disciplines or social groups (Eser 2002; Cash *et al.* 2003). Indeed, it is this vagueness and malleability, i.e. the potential variety of interpretations or applications of the term, that makes boundary objects politically successful (Eser 2002). Yet on the other hand boundary objects can in fact be a hindrance to scientific progress, as they can result in the ambiguouness, vagueness and unreflected normativity of the specific term, as in the case of resilience considered above. Boundary objects thus appear to be Janus-faced and difficult to handle (Brand & Jax 2007).

This thesis on the one hand suggests that both conceptual clarity and practical relevance of “resilience” are critically at stake. A scientific concept of resilience must have a clear and specified meaning that is consistantly used in the same way. It must be possible (*a*) to specify the specific objects the concept refers to, (*b*) to decide whether particular states in nature are resilient or non-resilient and it should be possible (*c*) to assess the degree of resilience in a certain state. In fact, the quality of the term “resilience” is strongly dependent on the ability to exclude phenomena that do not meet this term, as both operationalization and application with respect to environmental management are strongly dependent on a clear and delimited meaning of the term (Pickett *et al.* 2004: 57ff).

On the other hand however, this thesis proposes that the increased vagueness and malleability of “resilience” is highly valuable because it is for this reason that the concept is able to foster communication across disciplines and between science and practice (cf. Eser 2002). Therefore, it is not the suggestion to eradicate this vagueness and ambiguousness entirely but to grasp the ambivalent character of boundary objects and, hence, of a wide and vague use of “resilience”.

To counterbalance the positive and negative aspects of the conceptual development of resilience this thesis thus argues for division of labor in a scientific sense. Resilience conceived as a descriptive concept should be a clear, well-defined and specified concept that provides the basis for operationalization and application within ecological science. For the sake of clarity this meaning may be dubbed *ecological resilience/ ecosystem resilience* (for ecological systems) or just *resilience* (if applied to systems other than ecological, e.g. climatic systems). In contrast, resilience conceived as a boundary object should be designed in a manner so as to foster interdisciplinary work. In this sense, resilience constitutes a vague and malleable concept that is used as an transdisciplinary approach to analyze social-ecological systems. For the sake of greater clarity this meaning may be termed *social-ecological resilience* (as in Folke 2006).

Application of Ecological Theory to Social Systems

The second theoretical problem refers to the assumption of resilience research that the theory gained from ecological systems can be applied to social, economic or insitutitional systems as well (cf. section 3.5.2.3). Resilience research does indeed take literally all of its concepts or theory drawn from ecological systems (e.g. adaptive cycle, panarchy, alternative stable states, ecological thresholds, ecological redundancy, ecological memory) for achieving understanding in other types of systems (e.g. Gunderson & Holling 2002; Allison & Hobbs 2004; Kinzig *et al.* 2006). As Walker *et al.* point out "[t]he ecological and social domains of social-ecological systems can be addressed in a common conceptual, theoretical, and modeling framework" (Walker *et al.* 2006: 6). Thus, the theoretical aspirations and explanatory claims of resilience theory go far beyond ecological systems. Resilience research claims applicability to purely social domains as well (Reusswig 2007).

This is a non-trivial position of points. It is not at all self-evident that the theory gained for ecological systems holds true for other types of systems as well. As Reusswig (2007: 122) states with respect to the adaptive cycle: "[t]here is social dynamics not covered by the four phases, and there is history not following the cyclical mode of phase interconnectivity". This thesis only puts forwards a tentative critique of this assumption of resilience research, and suggests that the resilience approach unconsciously assumes the validity of a form of general systems theory. Therefore, the validity of resilience theory with respect to social systems should be discussed with respect to the strengths and weaknesses of general systems theory. In any case, the validity of ecological theory for social systems cannot be taken for granted.

Investigating the status of resilience theory within the social sciences and the humanities warrants further treatment.

Cultural partiality of resilience research

The third theoretical problem refers to the cultural presumptions of the resilience approach (cf. section 4.2). In this thesis, we[221] champion a cultural position within philosophy of science. We assume that scientific theories (such as resilience "theory") have philosophical and cultural underpinnings and we wonder how the way of empirical observation conducted in resilience research emerged. In particular, we tried to uncover structural similarities between philosophical/ cultural theories and resilience theory. Our results show that the systems notion of the resilience approach corresponds to a holistic and organic system notion, which is structurally similar to the philosophical ontology (i.e. monadology) of G. W. Leibniz. Also, the man-nature relationship proposed within adaptive co-management is structurally similar to the counter-enlightenment cultural theory and philosophy of history of J. G. Herder. Likewise, our results show that there are well-founded alternatives to resilience theory with respect to the explanation of ecosystem dynamics and man-nature couplings, in the form of an individualistic systems notion and conventional resource management. We conclude that resilience theory is based on specific cultural theories. Consequently, resilience theory focusses on particular holistic systems notions and counter-enlightenment theories of the man-nature relationship, and blanks out other well-founded, useful and reasonable notions and theories, such as the individualistic systems notion. We therefore consider resilience theory to be partial and as not universally valid with respect to the explanaition of ecosystem dynamics and the man-nature couplings.

Altogether it has become clear that the resilience approach (i) presupposes particular cultural and philosophical ideas; (ii) is based in its broader form (i.e. social-ecological resilience) on the contentious general systems theory; and (iii) is characterized by increasing vagueness, unreflected normativity and ambiguousness. These topics have to be taken into account when applying the resilience approach to a specific study region.

[221] The following insights result from the collaboration with Thomas Kirchhoff (München) and Deborah Hoheisel (München).

6.7 Stimulating Sustainability Science

Sustainable development is recognized the world over as a key challenge facing 21st century society (UNEP 2002; WBGU 2005; Komiyama & Takeuchi 2006; Clark 2007). The concept of ecological resilience is increasingly used as an innovative tool within sustainability science (Janssen 2007). Thus, the relation of resilience and sustainable development warrants special consideration and is one of the fundamental objectives of this thesis (cf. section 5).

At first, the term "sustainable development" must be clarified (cf. section 5.1). By reviewing the current international and national debate, this thesis identifies a lack of a well-founded normative basis for the justification of sustainable development within sustainability science. In order to fill this gap, we[222] aim to call attention to two of the salient conceptions in the German discourse, namely the 'Theory of Strong Sustainability' developed at the university of Greifswald and the 'Integrative Sustainability Concept' proposed by the Helmholtz Association of German Research Centres (HGF), the biggest research institution in Germany. Both conceptions highly value the justification of a strong theoretical and normative core of sustainable development. This thesis suggests that a well-founded 'Theory of Sustainable Development' provides the distinctiveness that allows for the assessment whether or not there is progress toward sustainability. A clear scientific comprehension of sustainability may inform politics in sustainability affairs and function as a rational corrective for the otherwise diffuse discussion in the general public (cf. Schultz et al. 2008). This thesis thus assumes an enlightened and multi-layer conception of sustainable development.

Based on such a clear conception of sustainable development, this thesis examines two fundamental topics within sustainability science: "critical natural capital" and "poverty reduction in developing countries". These topics are treated because "critical natural capital" is a fundamental concept within ecological economics (e.g. Ekins *et al.* 2003a), while poverty reduction represents one of the grand challenges of sustainability science (Kates & Dasgupta 2007).

Critical natural capital represents a multidimensional concept, as it mirrors the different frameworks of various scientific disciplines and social groups in valuing nature (confer section 5.2). Within the assumed conception of sustainable

[222] This section is the result of the collaboration with Julia Schultz (Berlin, Greifswald), Konrad Ott (Greifswald) and Jürgen Kopfmüller (Karlsruhe).

development, it is considered to be that part of the natural environment that ought to be maintained under any circumstances for the benefit of present and future generations. This thesis suggests that the criticality of the natural capital is related to two criteria: importance and degree of threat. I suggest that the degree of threat ecosystem are prone to can be indicated by the degree of ecological resilience. The concept of ecological resilience may complement other measures, such as integrity or vulnerability, in estimating the degree of threat that specific ecosystems are exposed to. The empirical estimates of ecological resilience add a further criterion in order to build a comprehensive and clear conception of critical natural capital (cf. Brand 2009).

Poverty reduction in developing countries is one of the fundamental objectives of sustainable development (cf. section 5.3). This thesis reviews the literature on the resilience-poverty nexus published in the last ten years. The results show a sharp increase in publication efforts in recent years. Resilience has been largely used as a catchword, rather than as a clearly defined concept. In addition, research has largely focussed on the concepts of economic resilience and social resilience. I suggest to switch the focus on the real innovative concepts of the resilience approach, namely ecological resilience and adaptive co-management. Subsequently, this thesis offers clear definitions, operational steps and management measures to sharpen these concepts and apply them to the topic of poverty reduction in developing countries. The concept of ecological resilience is applied to coral reef resilience, while the concept of adaptive co-management is applied to fisheries in developing countries.

This thesis concludes that the resilience approach has numerous shortcomings and limitations. Yet despite its weaknesses, resilience research can be of high relevance for several fields within sustainability science, such as identifying critical or managing natural capital.

6.8 Some Ideas for Further Research

I will now suggest some ideas for achieving progress in resilience research. An important research topic is the prediction of the position of ecological thresholds (confer section 3.2.3.3.4.2). Many ecosystems show regime shifts from one stable state to another, and often one state is undesirable from a societal perspective. Thus, forecasting the timing of regime shifts and the position of ecological thresholds would be of high relevance for environmental management. In addition, resilience mechanisms at larger scales (e.g. regional scales) are treated indiscriminately within resilience research. Hereby, the resilience approach should stick to one of the theoretical frameworks suggested by Leibold *et al.* (2004) (cf. section 3.3.2.2). Also, the operationalization of resilience (confer section 3.4) requires further inquiry. It should especially be examined in how far an ontological, holistic concept such as ecological resilience can in principal be measured or estimated and if it cannot how useful empirical measures would look like. Finally, the adaptive co-management approach should increasingly be applied to land-use decisions in developing countries (confer section 5.3), as the former represents one of the real innovations within resilience research and the latter one of the grand challenges of sustainability science.

Further ideas are related to external conceptual problems (confer section 2.2) of the resilience approach. The status of resilience theory within the social sciences and the humanities should be clarified. Resilience theory is extended from ecological systems to economic, social, institutional and coupled social-ecological systems, which is a non-trivial position of points (cf. section 3.5.2.3). It is very important to reflect on this extension of theory from the perspective of the social sciences and the humanities. Hereby, the relation of resilience theory to general systems theory along with its merits and threats warrants particular consideration.

Finally, resilience represents a boundary object (confer section 4.1.4). It would be interesting to investigate the general features of boundary objects, which includes the relation of resilience to other boundary objects, such as biodiversity, ecosystem services or sustainability (K. Jax., *personal communication*). If applied correctly the concept of resilience has the potential to stimulate environmental innovation within sustainability science.

7 Bibliography

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