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Ecological Resilience and its Relevance within a Theory of Sustainable Development

Fridolin Brand

UFZ Centre for Environmental Research Leipzig-Halle
Department of Ecological Modelling

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Preface

Why – the world – ecosystem resilience?

Writing a 200-pages thesis about a theoretical topic can be fun. It is interesting to find the structure, the leitmotifs, and the most important topics during the work. Sometimes – every two months or so - there occurs something like a breakthrough that encourages you to keep on going. As a matter of course most of the time it is just hard work.

The purpose of this preface is mainly to thank all the people that have contributed to this thesis that I handed in at the Ernst-Moritz-Arndt-Universität Greifswald in February 2005.

First, I want to appreciate all the work of numerous authors producing this vast amount of literature which is listed in my bibliography. It represents both the basis and the content of my thesis.

Thanks a million to my supervisors Prof. Konrad Ott[1] and PD Dr. Volker Grimm[2]. Konrad Ott helped me meticulously to find a topic that suits my interest and supported me in doing a thesis about a topic I have chosen on my own. He gave me precious advices for chapter 2 about sustainability discourse, the sections about theory of science and the overall structure of my thesis. Especially I want to thank him for his openness and kindness. He has always been taken me seriously. In general, he aroused my interest in Sustainable Development and related questions through his fabulous lectures and tutorials.

I want to thank Volker Grimm for supervising my thesis which has been a piece of luck for me. As genuine biologist and physicist he could help me a great deal to understand much of the theory behind stability concepts. Many, many thanks for all the e-mails, discussions and comments on my provisional creations. Particularly I want to thank him for his kindness and benevolence.

Thanks a million to my parents. You have given me so much affection which is invaluable. I know that I can rely on you completely.

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[1] Prof. Konrad Ott, Botanisches Institut, Professur für Umweltethik, Ernst-Moritz-Arndt Universität Greifswald.

[2] PD Dr. Volker Grimm, Umweltforschungszentrum Leipzig-Halle, Abteilung Ökologische Systemanalyse.

Preface (cont.)

Thanks a lot to all my friends. Without you, all would be nothing. Especially I want to thank Andi for bearing the life of suffering with me and sitting in the Staatsbibliothek München each day during the last months.
And eventually I want to thank myself for standing all that intellectual work - which can be hard sometimes - and for leading the life of my choice.

Fridolin Brand
frido_brand@hotmail.com

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List of Abbreviations

BEFP	Biodiversity-Ecosystem Function Paradigm
CC	Carrying capacity
Cf.	Confer
CNC	Critical natural capital
EE	Ecological Economics
E.g.	Exempli gratia = for example
Et al.	Et alii = and others
DNA	Deoxyribonucleic acid
EKC	Environmental-Kuznets-Curve
GBA	Global Biodiversity Assessment
GNP	Gross National Product
G&W	Grimm & Wissel (1997)
H&H	Hansson & Helgesson (2003)
I.e.	Id est = that is
ISEE	International society for Ecological Economics
ISEW	Index of Sustainable Economic Welfare
L	Latitude
OECD	Organization for Economic Cooperation and Development
P	panarchy
Pr	Precariousness
R	Resistance
SCOPE	Scientific Commitee on Problems of the Environment
SD	Sustainable Development
SES	Social-ecological system
StS	Strong sustainability
UFZ	Umweltforschungszentrum Leipzig-Halle
UNCTAD	United Nations Commission on Trade and Development
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UNWSSD	United Nations World Summit on Sustainable Development

List of Abbreviations (cont.)

WBGU	Wissenschaftlicher Beirat für Globale Umweltveränderungen
WCED	World Commission on Environment and Development
WS	Weak sustainability
WSSD	World Summit on Sustainable Development
zB	Zum Beispiel

1 Introduction

On the occasion of the recent Tsunami disaster in South Asia, many people involved called for a hazard early warning system for the Indian Ocean in order to be able to anticipate similar extreme events in the future. There are some voices, however, that argue for additional measures that make regions less vulnerable to natural disasters and capacitate them to respond in a flexible way.

Speaking at the World Conference on Disaster Reduction in Kobe, Japan, Klaus Töpfer, head of the United Nations Environment Programme (UNEP), states that technological solutions will not be sufficient for vulnerable regions to prepare and respond to extreme events and suggests that

> “we need a robust nature that can tolerate impacts of disasters and help fight the consequences (…). Such systems must incorporate more than technology (…). They should represent a new way of thinking that ensures environmental stability factors, based on local wisdom and knowledge, are built into disaster plans (…). In the same way that we have building standards for construction in earthquake zones, we need to put a disaster prevention value on our natural ecosystems. We need to make our own ‘construction criteria’, a criteria that places an ecosystem value on our homes and infrastructure (…). Such an investment, whether in the coastal communities of the Indian Ocean or elsewhere, will lessen the impact of disasters when they happen, and provide for greater stability and reduced vulnerability around the world” (UNEP 2005).

From a similar perspective, considering the Tsunami disaster and its consequences, a German magazine recently presented the earth from the moon and headed their title page mysteriously “the fragile planet” (SPIEGEL 2005).

Both statements reflect the notion that there is something as “stability” out there in nature, something that is prone to disturbance and disasters and can be overwhelmed but something like “stability” that is also alterable by human action and can be enhanced.

Along with waxing global recognition of the devastating effect of natural disasters, awareness for environmental problems increased heavily. Persistent problems such as climate change, loss of biodiversity, habitats and fertile soils or shortage of drinking water can not be just dismissed as fashion (Ott 1994). Technical optimists like Björn Lomborg, who deny the severity of the problems and argue for *first-get-rich! strategies* in order to combat environmental degradation that is due to economic growth within Southern countries, are criticized strongly (Ott et al. 2003).

Following the latest survey of the UNEP, the ‘state of the world’ appears not only as a *Garden of Eden* (UNEP 2002).

World population increased from about 3.85 billion people in 1972 to 6.1 billion in mid-2000 and is currently growing by 77 million people a year. Population in developing regions is projected to increase from 4.9 billion in 2000 to 8.1 billion by 2050.

It has been estimated that 23 per cent of all usable land has been affected by degradation to a degree sufficient to reduce its productivity. 15 per cent of the earth’s land area have been degraded through human activities. Some 25-30 million ha of the world’s 255 million ha of irrigated land were severely degraded due to the accumulation of salts. Degradation in arid, semi-arid and dry sub-humid areas (*desertification*) amounts to 70 per cent of the total area. More than 250 million people are directly affected by desertification. The main driving force leading to pressure on land resources has been increasing food production. In 2002, food has been needed for some 2220 million more people than in 1972.

The net loss of forest area during the 1990s was an estimated 9.4 million ha per year (2.4 per cent of total forest). The world’s natural forests have continued to be converted to other land uses (e.g. forest plantations, agricultural land) at a very high rate. During the 1990s, the total loss of natural forests was 16.1 million ha per year.

There has been a sharp global trend towards increasingly intense exploitation and depletion of wild fish stocks. In 1994, an estimated 37 per cent of the global human population lived within 60 km of the coast – more people than inhabited the planet in 1950. As a result, three-quarters of the fish stocks are maximally exploited and many have collapsed. Additionally, various marine ecosystems have been deteriorated and collapsed including kelp forests, coral reefs, tropical and subtropical sea-grass beds, estuaries and offshore benthic communities (Jackson et al. 2001).

These persistent environmental problems constitute, among others such as climate change and loss of biodiversity, the *environmental crisis* (Sachs 1999).

On the other hand, humanity is facing severe problems of human development and unjust conditions of economic distribution in countries of the North and the South (*fairness level 1*) and between the middle class and the marginalised majority or minority within countries (*fairness level 2*) which is coined the *crisis of justice* (Sachs 1999).

At least, the past 30 years saw dramatic improvements in the developing world as life expectancy increased by eight years, illiteracy was cut nearly in half to 25 per cent and in East Asia the number of people surviving on less than 1$ a day was almost halved just in the 1990s. However, still human development is proceeding too slowly as some 54 countries are poorer now than in 1990, in 21 a larger proportion of people is going hungry, in 14, more children are dying before age five, in 12 primary school enrolments are shrinking, in 34 life expectancy has fallen (UNDP 2002). Still, 1.2 billion people live in extreme poverty on less than 1$ a day, and 2.8 billion people on less than $2 a day. Poverty is not limited to developing countries as more than 130 million people in developed countries of the Organization for Economic Cooperation and Development (OECD) are considered income-poor (UNEP 2002).

As a respond, the *Millenium Declaration* has been launched in 2000, which includes the *Millenium Development Goals* that bind countries, among other goals, to eradicate extreme poverty and hunger, to achieve universal primary education and to promote gender equality and empower women (UNDP 2002).

The environmental crisis and the crisis of justice represent the main sources for the emergence of the new paradigm *Sustainable Development*. Konrad Ott (2001) suggests that Sustainable Development is mainly about reflections on distributive justice in view of the natural capital stock. It represents the regulative idea “that present and future persons have the same right to find, on the average, equal opportunities for realising their concepts of a good human life” (Ott 2003, 60). This implies poverty eradication in Southern countries (Sachs 1999, Kopfmüller et al. 2001, Ott 2001, WSSD 2002).

Within ecological science, the *biodiversity-stability debate* outclasses every other discussion since 30 years. The main questions are whether biodiversity begets ecosystem stability and what mechanisms are responsible for the maintenance of ecosystem structure and function in the face of disturbances (Loreau et al. 2001, 2002). Hereby, *ecosystem 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” (Holling & Gunderson 2002, 4) which reflects the capacity (i.e. the underlying mechanisms) of ecosystems to maintain services in the face of a fluctuating environment and human perturbations (Carpenter et al. 2001, Folke et al. 2002). The theoretical aspects of the concept of ecosystem resilience are mainly treated within ecological science.

Within the new emerging research paradigm *sustainability science* (Clark & Dickson 2003) and *sustainability debate* (Kopfmüller et al. 2001, Ott 2001, Neumayer 2003), proponents of strong sustainability argue for the maintenance and preservation of natural capital due to moral reasons since future generations are to be at least as well off as present generations. The natural capital stock should remain intact which includes the notion that this stock is to be “stable” in the long run. Hereby, scholars often refer to the concept of ecosystem resilience. Kates et al. (2001) identify the *core questions of sustainability science*, which are, among others, represented by

> “(…) What determines the vulnerability or resilience of the nature-society systems in particular kinds of places and for particular types of ecosystems and human livelihoods? Can scientifically meaningful “limits” or “boundaries” be identified that would provide effective warning of conditions beyond which the nature-society systems incur a significantly increased risk of serious degradation? (…)” (Kates et al. 2001, 642),

and explicitly refer to the concept of (ecosystem) resilience. The relevance of the concept of ecosystem resilience has to be understood against this background of sustainability debate.

Within environmental management, many resource systems have collapsed leaving the regions with decreased essential ecosystem functions and services (Gunderson, Holling & Light 1995, Holling & Meffe 1996). These collapses may be due to an unsustainable management approach that does not ensure the ecosystem resilience of ecological systems in the long run. Hereby, research groups, that work on the concept of ecosystem resilience, try to find recommendations for a management approach that creates resource systems that are robust enough to face environmental fluctuations. Attempts to manage for the provision of ecosystem services in the long run represent another source for the relevance of the concept of ecosystem resilience.

For this thesis, I have reviewed relevant papers and books most of which are the scientific product of the research group *Resilience Alliance* since this scientific network presents the leading authority in the field of ecosystem resilience. The purpose of the thesis is two-fold: (1) to present the theoretical fundamentals of the concept of ecosystem resilience which includes the identification of concepts that are related to ecosystem resilience and together constitute its background theory, as well as (2) to examine the abilities for estimating ecosystem resilience empirically as well as theoretically and to unfold its relevance for a Theory of Sustainable Development.

Chapter 2 initially reflects the occurrence of the concept of ecosystem resilience within the argumentative space of sustainability discourse and its relation to various notions and concepts occurring frequently in this debate. Hereby, section 2.1 points to the relation to the concept of outer limits. This section more or less examines a path on which ecosystem resilience emerged within sustainability debate. Section 2.2 considers the connection to a Theory of Sustainable Development and explores some of the reasons why we should care about ecosystem resilience, i.e. its moral and political relevance. Section 2.3 turns our attention to Ecological Economics, a currently emerging, scientific discipline that is highly relevant for Sustainable Development, and its relation to the concept of ecosystem resilience. In this respect, the concept of critical natural capital appears to be highly relevant.

Subsequently, chapter 3 explores the theoretical foundations of the concept of ecosystem resilience and the concepts that are related to it, which together constitute the *background theory of ecosystem resilience* (section 3.2). It turns out that a notion of complex adaptive systems is needed in order to understand the ecosystem resilience concept more profoundly (section 3.2.2). Additionally, ecosystem resilience presupposes the concept of alternative stable regimes currently hotly contested within ecological science (section 3.2.3). Section 3.3 examines how ecosystem resilience is embedded in biodiversity-ecosystem functioning debate and biodiversity-stability debate, respectively. Hereby, section 3.3.3 considers the mechanisms that are seen as responsible for the emergence of ecosystem resilience on the ecosystem level which comprise ecological redundancy, response diversity and ecological memory. A concluding discussion of theoretical aspects and models of ecosystem resilience in section 3.4 results in the distinction of seven levels of meaning.

Subsequently, chapter 4 considers the abilities to operationalize ecosystem resilience as well as the possibilities to implement it through measures of environmental management. Thereby, section 4.2 suggests a measurable concept of ecosystem resilience which is part of an detailed *ecosystem resilience analysis* which, in turn, comprises the identification of temporal and spatial scale, the choice of desired ecosystem services and the specification of slow variables that control the desired regime of the ecosystem. Section 4.3.2 points to some possibilities to implement ecosystem resilience into appropriate, environmental management measures. Hereby, ecological knowledge, biodiversity and small-scale disturbances are regarded to be important.

Providing a summary and synthesis, chapter 5 concludes with findings and results of the thesis and recommendations for further research. The Appendix supplies a German translation of the summary and synthesis chapter as well as hints to relevant literature arranged by the topic of study.

This thesis is entirely theoretical although comprising many case studies and empirical results of investigations related to ecosystem resilience. Theoretical reflections can be seen as an essential partner to empiricism (Levin 1992). As a result from the *Biodiversity and Ecosystem Functioning Synthesis Conference* held in Paris in 2000, Naeem et al. state that

> “ecological truth lies at the confluence of observation, theory and experiment. It is through discourse among empiricists and theorists that findings and theory are sorted and matched and where there is a lack of correspondence, new challenges identified” (Naeem et al. 2002, 11).

In this sense, I consider this thesis to be a serious contribution to the debate about ecosystem resilience in order to achieve a sound approach to nature conservation.

2 Relevance of Ecosystem Resilience within Sustainability Discourse

Ecosystem resilience represents a term that is used increasingly within sustainability discourse.

This chapter will provide some insights about the relevance of the concept of ecosystem resilience with respect to the idea of *limits to human growth* (section 2.1), a *Theory of Sustainable Development* (section 2.2) and *Ecological Economics* (section 2.3).

2.1 Ecosystem Resilience & Limits to Growth

A milestone in establishing the notion of *sustainability* in international consciousness presented the often-quoted book *The limits to growth* edited by the Club of Rome (Meadows et al. 1972). The 50 self-appointed wise women and men developed, for the first time, the *concept of outer limits* – the idea that development could be limited by the finite size of the Earth's resources, fundamentally affecting the understanding of human life.

Also in the 1970s, the United Nations Conference on the Human Environment in Stockholm was held in 1972 being the first international conference of this size on environmental issues. Results of the conference included the formulation of an Action Plan of 109 recommendations and an Declaration of 26 principles (UNEP 2002b). These principles comprise the claim to maintain earth's capacity to produce renewable resources (*nature as source*, cf. principle 3) as well as the principle that pollution must not exceed the environment's capacity to clean itself (*nature as sink*, cf. principle 6).

One year later, the Cocoyoc Conference in Mexico, organized by the United Nations Environmental Program (UNEP) and the United Nations Commission on Trade and Development (UNCTAD), identified the economic and social factors which lead to environmental deterioration. The *Cocoyoc Declaration* ends with the statement that "the road forward does not lie through the despair of doom-watching or through the easy optimism of successive technological fixes. It lies through a careful and dispassionate assessment of the 'outer limits'" (quoted from UNEP 2002b).

In the so called "lost decade" of the 1980s [no growth in income in most developing countries[1], number of refugees doubled (UNEP 2002b)] the *Global 2000* report published in 1980 recognized for the first time that species extinction was threatening biodiversity as an essential component of the Earth's ecosystems. Similarly, the World Council for Nature stated that "ecosystems and organisms, as well as the land, marine and atmospheric resources that are utilized by man (*sic*),

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[1] I am aware of the problematic terms *developed* and *developing* since it implies a development telos which is not reflected of its justification. I will use the terms developed and developing in the comprehensive sense of the UNDP reports on Human Development (e.g. UNDP 2002).

shall be managed to achieve and maintain optimum sustainable productivity, but not in such a way as to endanger the integrity of those other ecosystems or species with which they coexist"[2] (quoted from UNEP 2002).

The term *ecosystem* gets relevant here. For the *Global Biodiversity Assessment*, Mooney et al. (1996) consider an ecosystem to refer to all individuals, species and populations in a spatially defined area, the interactions among them, and those between the organisms and the abiotic environment. Similarly, in the *Dictionary of Ecology, Evolution and Systematics* an ecosystem is defined as "a community of organisms and their physical environment interacting as an ecological unit; the entire biological and physical content of a biotop (…)" (Lincoln, Boxshall & Clark 1998, 95). Note, however, that definitions are only useful for given objectives (Jax 2002, cf. section 3.1.1). In the following I will use the UNEP definition pragmatically as provisional notion of the term ecosystem.

In 1987, the World Commission on Environment and Development (WCED), also known as the *Brundtland Commission*, published the book *Our Common Future* which highlights environmental problems such as global warming and ozone layer depletion and concludes that existing decision-making structures and institutional arrangements, both national and international, could not cope with the demands of Sustainable Development (UNEP 2002b). The well-known and often-cited definition of the WCED defines Sustainable Development (SD) as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987).

The United Nations Conference on Environment and Development held in Rio de Janeiro in 1992, launched the *Agenda 21*. This action plan states that

> "[e]arth's capacity to sustain and nourish life depends primarily on the qualities and composition of its atmosphere. Human activities have now reached the extent to which they are altering the atmosphere's balancing systems that make life on Earth possible.

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[2] *Biological integrity* represents an imprecise term but can be defined as "the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and meta-population processes), expected in the natural habitat of a region" (Karr 1996, 101). It is related to the concept of ecosystem resilience and *ecosystem health*. In my view, the normative connotations increase from ecosystem resilience via biological integrity to ecosystem health. Ecosystem health actually is an explicitly normative concept since ecosystems are only considered to be healthy if they maintain the human-valued ecosystem services. A degraded and unproductive ecosystem is not regarded as healthy (Karr 1996). On the contrary, degraded ecosystems can be highly ecosystem resilient to disturbances. Ecosystem resilience is, thus, *not something that is desirable as such* (Carpenter et al. 2001, Carpenter & Cottingham 2002, Gunderson & Holling 2002, Walker et al. 2002, cf. section 4.2).

> (…) With rising production and the continuance of wasteful and destructive consumption patterns, economic development at the local, national and global level could well be overwhelmed by the waste and pollution it produces" (UNCED 1992).

Ten years later, the *World Summit on Sustainable Development* was held in Johannesburg. Similarly, the authors of the plan of implementation insist on promoting "social and economic development within the carrying capacity of ecosystems" (UNWSSD 2002).

The important point I want to make here is that most of these steps for establishing the notion of SD refer to the concept of outer limits, on the patch or landscape scale as well as on the regional and world-scale[3]. It was in 1978 when C.S. Holling stated that "it is commonplace now to perceive limits – limits to growth, to resources, to climatic and environmental stability" (Holling 1978, 5). In my view, it is justified to propose that the notion of outer limits encourages the use of the ecosystem resilience concept since ecosystem resilience puts the notion of outer limits in concrete terms. The historical steps to establish the *idea of limits to growth*, thus, represents a path that *levelled the way for the recognition of the relevance of the ecosystem resilience concept*. How is the notion of outer limits related to the ecosystem resilience concept, however?

In the first place, the notion of outer limits appears to be closely related to the concept of *carrying capacity* that is put forward by ecologists. Seidl & Tisdell (1999) suggest that the discussion in the late 1960s and early 1970s about looming limits of the Earth's carrying capacity due to population and economic growth initiated the widespread development of environmental awareness. Presently, not many people doubt the rapid decline and deterioration of environmental resources, the overuse of ecological sinks, and the fact, that such overuse deteriorates and destroys ecosystems and ultimately living conditions of humans and other species.

The concept of carrying capacity has played a significant part in promoting public and political awareness and understanding of existing limits to economic activity. Carrying capacity (CC) usually refers to the "maximum number of organisms that can be supported in a given area or habitat" (Lincoln, Boxshall & Clark 1998, 51) and since the late 1960s sustainability scholars use the concept likewise for earth's

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[3] *Scale* can be defined as "a range of spatial and temporal frequencies" (Peterson et al. 1998, 11).

capacity to sustain a certain size of human population[4] (e.g. Daily & Ehrlich 1992, Ehrlich 1994, Seidl & Tisdell 1999).

Ehrlich (1994) distinguishes between two kinds of CC. The first is *biophysical carrying capacity*, defined as the maximum number of people that can be supported at given levels of technology. The second is *social carrying capacity*, or the maximum number of people that can be supported at a given level of technology within a given social organization, including patterns of consumption and trade. It is possible that the social K is higher, *ceteris paribus*, for a global society of vegetarians than for a global society with the consumption preferences of rich Americans.

Estimates of the carrying capacity refer to two distinct concepts (Daily & Ehrlich 1992). The first is the concept of *maximum sustainable abuse* or the ability of ecosystems to withstand human exploitation in the long-run. The second is the concept of *maximum sustainable use* which defines the amount of resources that world's ecosystems are able to provide in order to satisfy human needs and wants. Hence, nature's ability to function as source and sink is expressed when estimating earth's carrying capacity. Both aspects of carrying capacity, in turn, are limited which is reflected by the idea of outer limits. In this respect, Folke et al. (1994) propose that for a SD the physical human scale must be more or less limited within the carrying capacity.

Note, however, that the notion of a fixed biophysical or social carrying capacity of human activity assumes a certain view of nature [*Nature Balanced*, cf. section 3.2.1] that is not incorrect but incomplete. There are indeed forces of balance in ecological systems but these forces can become overwhelmed (Holling et al. 2002).

Moreover, the concept of CC is influenced by value-judgements and institutional settings. CC has an important normative component (Seidl & Tisdell 1999), i.e. comprises statements about what *should* be the case (with respect to carrying capacity e.g. what average level of material well-being should we choose and how should well-being be distributed? Which natural conditions we want to live in?). As Seidl & Tisdell point out, CC seems to be "foremost socially determined rather than biologically fixed due to the important influence of human consumption

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[4] CC is particularly difficult to define for human populations (Daily & Ehrlich 1992). CC is a function of both the population's resource base (in the broadest sense) and the characteristics of the organism. Humans, however, are able to modify both their resource base and their behaviour (Ehrlich 1994).

patterns, technologies, infrastructure, and impacts on the environment or food availability" (Seidl & Tisdell 1999, 403).

In this respect, Arrow et al. state that:

> "[c]arrying capacities in nature are not fixed, static, or simple relations. They are contingent on technology, preferences, and the structures of production and consumption. They are also contingent on the ever-changing state of interactions between the physical and the biotic environments. A single number for human carrying capacity would be meaningless because the consequences of both human innovation and biological evolution are inherently unknowable" (Arrow et al. 1995, 93).

One of the most recent developments of the carrying capacity concept has been to relate it to ecosystem resilience (Seidl & Tisdell 1999). We have defined *ecosystem resilience* provisionally as the "magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior" (Gunderson & Holling 2002, 4).

The ecosystem resilience concept, in turn, gets more concrete and measurable if one specifies ecosystem resilience *of* what *to* what (Carpenter et al. 2001). Walker et al. (2002) distinguish a *to-what part* from a *of-what part* of the so-called *ecosystem resilience analysis*. The to-what part refers to the disturbance regime. The of-what part, on the other hand, includes the specification of the spatial and the temporal scale as well as the reference state (Walker et al. 2002, cf. section 4.2).

For instance, the mire's capacity to absorb CO_2 emissions or the forest's ability to provide timber can be regarded as desired *ecosystem services* of an ecosystem's reference dynamic. Thus, both fundamental capacities of nature - the ability to provide resources and to absorb wastes of human impact - can become included in the concept of ecosystem resilience, which means so understood the *capacity of an ecosystem to maintain the ecosystem services of concern in the face of a fluctuating environment and human perturbations* (the *ecological-systemic meaning* of ecosystem resilience cf. in detail section 3.2.4 and 3.4.3).

According to Perrings et al. (1995b), the notion of carrying capacity is an indirect measure of the level of stress that is consistent with a tolerable level of ecosystem resilience. Carrying capacity has been exceeded when ecosystem resilience is lost and a system shifts from one alternative stable *basin of attraction* [5] to another because the ecosystem is so altered that its ecosystem resilience in relation to its original basin is overcome (Seidl & Tisdell 1999, cf. Holling & Gunderson 2002, Gunderson & Pritchard Jr. 2002). Since, for a given technology, human population growth implies an increasing level of stress on the ecosystems exploited, there is necessarily some point at which the associated loss of ecosystem resilience will become critical[6] (Perrings et al. 1995b, Ekins, Folke & deGroot 2003).

Therefore, it appears that the carrying capacity that is linked to the idea of outer limits is, in turn, closely linked to the concept of ecosystem resilience. Perrings, for instance, proposes that "[l]essons about carrying capacity turn out to be very closely related to lessons about the resilience and stability of ecological economic systems" (Perrings 2002). Folke et al. state that "[e]arth's carrying capacity is dependent on the resilience of ecosystems" (Folke et al. 1994, 5) and, similarly, Arrow et al. suggest that "economic activities are sustainable only if the life-support ecosystems upon which they depend are resilient"[7] (Arrow et al. 1995, 93). These statements are interesting in suggesting that the limits to growth literature and the idea of outer limits encourage the use of the concept of ecosystem resilience and *vice versa*. Outer limits refer to the carrying capacity of the environment to support human activities at various scales and this capacity is, in turn, dependent on the ecosystem resilience of ecosystem functioning (similar Folke et al. 1994).

Eventually, it appears to me that the concept of outer limits is often related to the earth scale whereas the concept of carrying capacity and, in particular, the concept of ecosystem resilience are applied more to smaller scales, especially to the landscape scale.

Thus, the concept of ecosystem resilience presumes the idea of outer limits and *vice versa*.

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[5] The concept of ecosystem resilience assumes that ecosystems can exist in alternative basins of attraction which, in turn, corresponds to a region in state space (i.e. the variables considered and their relation) in which a system tends to remain (Walker et al. 2004, cf. section 3.2.3).

[6] What does it mean, however, to say the loss of ecosystem resilience will become critical (Seidl & Tisdell 1999)? This leads to the examination of *ecological thresholds* which will be explored in detail in section 3.2.3.

[7] Note that Perrings (2002), Folke et al (1994) and Arrow et al. (1995) use the term *resilience* for the meaning of *ecosystem resilience sensu* Gunderson & Holling (2002).

2.2 Ecosystem Resilience & Strong Sustainability

This section will provide some insights about the connection of the concept of ecosystem resilience to a Theory of Sustainable Development. After examining the structure and content of a sustainability theory, this section will explore some of the reasons why we should care about ecosystem resilience.

The first use of the term *sustainability* is thought to be found in the *Sylvicultura Oeconomica* from von Carlowitz written in 1713 (cf. Kopfmüller et al. 2001). Von Carlowitz used the term sustainability for a forestry which uses only as much wood as is renewed by new plantations.

Since then various definitions of Sustainable Development (SD) and sustainability, respectively, emerged within the literature relevant to the subject. Both scientists and political agents have used the umbrella term Sustainable Development in an increasingly indiscriminate and arbitrary way (Ott 2003). SD comprises the potential to function as a flagship for various social, environmental, as well as economical policy goals within an international context for both developed and developing countries. This comprehensiveness lead to the attractiveness and, hence, to the broad acceptance of the SD concept in the first place. The term Sustainable Development is successful because everybody can agree (Acker-Widmaier 1999).

As a result, the widespread *three-pillar concept*. i.e. the distinction of an environmental, a social, and an economic sphere, for instance, has been reduced to a listing of any societal objectives that agents happen to think as important (Ott 2003). The meaning of sustainability, thus, gets vague.

For greater clarity, Ott (2001, 2003) distinguishes between eight layers of the overall debate on sustainable development which is outlined in *Table 1*.

If each layer is filled with content it will be justified to speak of a *Theory of Sustainable Development*. A sustainability theory has to be conceived as a form of applied ethics that rests on a *general ethical theory* [8] and a position with respect to *environmental ethics* [9]. Ott's (2001, 2003) theory of strong sustainability rests on the

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[8] Within the history of ethics, various general ethical theories have attempted to justify moral obligations. Ott (2001b), for instance, distinguishes between *contractarianism, consequentialism, Kantian ethics*, the approach from Alan Gewirth, and *discourse ethics*.

[9] Environmental ethics represents a type of applied ethics which rests on a general ethical theory. Frankena (1997) distinguishes several types with respect to the extent of the *moral community*. The moral community

foundation of discourse ethics (Habermas 1981, cf. Ott 2001b, Ott 2003b) and sentientism (e.g. Ott 2003d).

Table 1: Eight layers of sustainability debate
(according to Ott 2003)

(1) Idea
(2) Concepts
(3) Guidelines (resilience, sufficiency, efficiency, etc.)
(4) Dimensions (environment and nature, social systems, economy, education, culture, etc.)
(5) Management rules in single dimensions
(6) Objectives (targets, time frames, set of instruments)
(7) Indicators
(8) Implementation

Note that (ecosystem) resilience is perceived as *guideline* for a theory of sustainable development (cf. *Table 1*). In general, the higher layers do not strictly determine the lower layers, rather they provide some orientation (Ott 2001). The concept of (ecosystem) resilience, thus, influences the lower levels of *operationalization* and *implementation* steps within SD that will be the focus of chapter 4.

2.2.1 Ethical Idea

The starting point for theory formation is the ethical idea of sustainability which includes an appropriate definition and some ethical principles.

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corresponds to natural entities which are taken into consideration morally. The community is separated, in turn, into entities that are able to think and judge about moral questions (*moral agents*) and entities that are taken into consideration morally only (*moral patients*). In this respect, Frankena (1997) distinguishes between *egoism*, *humanism* , i.e. only humans but *all* humans are moral patients, *sentientism*, i.e. all beings that are able to suffer are treated as moral patients, *biocentrism*, i.e. all living entities are taken into consideration morally, *ecocentrism*, i.e. the moral community includes all living entities and above-individual natural units such as species and ecosystems and *holism*, i.e. all entities that exist are considered to be moral patients [cf. in detail Krebs (1997), Birnbacher (1997) and Ott & Gorke (2000)].

Kopfmüller et al (2001), for instance, base their conception of SD on the *Brundtland-definition* which states that "sustainable development is development that meets the needs of the present without compromising the needs of future generations to meet their own needs" (WCED 1987). They consider it as to be the only definition that is internationally accepted as well as acknowledged within numerous UN documents (Kopfmüller et al. 2001).

Building on that, Ott suggests that sustainability means "that present and future persons have the same right to find, on the average, equal opportunities for realising their concepts of a good human life" (Ott 2003, 60).

Both the *Brundtland-definition* and the *Ott-definition* include demands for intra-generational justice ("meets the needs of the present") and inter-generational justice ("without compromising the needs of future generations")[10]. The definition of the term Sustainable Development is, thus, directly connected to the demand for justice[11] – the obligation to provide all (present and future) people a life in human dignity[12] (Ott 2001). The ethical foundation of justice in general is possible on several tracks[13].

Eventually, according to Acker-Widmaier (1999), inter-generational and intra-generational justice are logically linked and not separable since inter-generational justice presupposes a general theory about distributive justice. To speak of intergenerational justice means, due to logical reasons, to speak of justice between humans at a certain time (cf. also Hampicke 1999, Attfield 1999, Kopfmüller et al. 2001, Ott 2001). The Brundtland definition and the Ott-definition, therefore, imply poverty eradication in Southern countries which is also the position of the *Agenda 21* launched in Rio de Janeiro in 1992, and of the *Plan of Implementation* of the *World Summit on Sustainable Development* held in Johannesburg in 2002 (Sachs 1999, World's Scientific Academies 2000, Ott 2001, WSSD 2002).

Another widely known definition, which emphasizes the ecological dimensions of sustainability, is that offered by Costanza et al.:

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[10] There arguably remain two questions: whose needs?, and what needs? (cited in Sachs 2002, 89). Eventually sustainable development is not only a moral and ethical issue but a prudential one also. Saving the environment actually means saving ourselves (Folke et al. 1994, 3).

[11] There are several forms of justice. Within sustainability debate, however, justice is always conceived as *distributive justice* (Acker-Widmaier 1999).

[12] Needs in the Brundtland definition can be understood as basic needs (food, shelter, education, etc.) or as notions of a good human life (Acker-Widmaier 1999, Ott 2003)

[13] First, through rational construction (e.g. *contractualism*), second, through generalization of subjective attitudes and social traditions (e.g. *comunitarianism*), third, through the claim of non-debatable premises (*natural law* conceptions), and fourth, through basing the conception on existing international public law (*human rights* approach) (Kopfmüller et al. 2001).

> “Sustainability is a relationship between human economic systems and larger dynamic, but normally slower-changing ecological systems, in which (1) human life can continue indefinitely, (2) human individuals can flourish, and (3) human cultures can develop; but in which effects of human activities remain within bounds, so as not to destroy the diversity, complexity, and the function of ecological life-support systems” (cited in Faucheux & O´Connor 1998, 19).

This definition is linked to the concept of limits to growth. As I mentioned before, I suppose outer bounds to be implicitly connected to the concept of ecosystem resilience. The important point here is whether the ecosystem resilience concept is taken into consideration by some definitions of Sustainable Development. In my opinion, regarding sustainability the ecosystem resilience concept functions as a *core idea sensu* Jacobs (1999).

2.2.2 Weak versus Strong Sustainability

Like other political terms such as democracy, liberty or social justice sustainable development itself is a *contestable concept*. Jacobs (1999) considers the term Sustainable Development as to have two levels of meaning.

On the first level it can be expressed in a short definition and by a number of *core ideas*. Jacobs (1999) identifies environment-economy integration, futurity, environmental protection, equity, quality of life, and participation, respectively, as candidates for SD.

The interesting feature of contestable concepts, however, emerges at the second level of meaning. This is where a scientific discourse and political debate about alternative conceptions of the concept occurs. “For common political concepts, the battle is neither over the first level of meaning nor indeed whether one accepts the normative goal. (…) The debate is over alternative conceptions of what they mean, at the second level” (Jacobs 1999, 25). Similarly, Yorque et al. (2002) state that the question is not whether to seek sustainable futures, but rather *how* to seek sustainable futures.

The basic question at this layer of examination is “what to sustain?” which is rather seen as controversial within public debate as well as among sustainability scholars. “Environmentalists want environmental systems sustained. Consumers want consumption sustained. Workers want jobs sustained” (Norgaard 1988, quoted from Faucheux & O’Connor 1998, 20). According to Ott (2001), the task what to sustain is mainly about a reflection on our obligations to future generations which are met by a *fair bequest package* of societies’ productive potential. Within sustainability debate scholars differ about how the bequest package should be composed[14].

What is rather clear is that it can be interpreted as a compound of different types of capital. *Capital*, as used within sustainability debate, is interpreted as “stock that yields a flow of valuable goods or services into the future” (Constanza & Daly 1992, 38). Sustainability scholars distinguish between four types of capital: (1) *manufactured capital* comprises material goods, (2) *human capital* takes into account all individuals’ capacities for work, (3) *social/ organisational capital* includes the networks and organisations through which the contributions of individuals are mobilised and coordinated, and eventually (4) *natural capital* (also named *ecological capital*) (Ekins et al. 2003)[15].

Daly (1996) defines natural capital as the capacity of the ecosystem to yield a flow of natural resources and a flux of natural services. This type of capital comprises, first, the provision of resources for production, which is coined *source function*, second, the absorption of wastes from production, which is termed the *sink function*, which both, in turn, rest on, third, the basic *life-support functions*, creating, fourth, the *amenity services*, such as the beauty of wilderness (Ekins et al. 2003). The concept of natural capital is often employed in describing the dependence of human economic activity on the environment (Faucheux & O’ Connor 1998) and is, thus, much more than the stock of produced assets traded in the market (Perrings et al. 1995).

It is questionable what exactly counts as natural capital[16]. Ott (2001), therefore, states that the term natural capital has not yet been clarified satisfactorily.

In one sense the value of natural capital is infinite since the Earth would stop to function without the services of ecological life-support systems. Costanza et al. (1998) made the effort to value natural capital in monetary terms knowing that

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[14] For Costanza & Patten the achievment of a consensus on what we want to sustain is crucial for sustainability debate (1995, 193)

[15] A clear distinction between natural and artificial capital is impossible in reality since untouched nature does not exist anymore. Some (mainly European) authors offer *cultivated natural capital* as mixed but distinct type.

[16] Related to this, Ekins, Folke & deGroot (2003) interpret natural capital as a metaphor to indicate the importance of elements of nature.

only a fraction of ecosystem services affects private goods traded in existing markets. Costanza et al. (1998) do not think that monetary valuation can be a substitute for moral reflection on what facets of natural capital we value enough in order to preserve them. However, any decision that society make imply valuations and society can choose to make them explicit or not. The authors estimate the marginal use value (not the total use value which is infinite) of world's ecosystems at a range of annually US$ 16 – 54 trillion. This amounts to 1.8 times the global Gross National Product (GNP)[17]. The value of earth's ecosystems is one way to show the dependence of the economy on natural systems.

If the bequest package of natural resources that we will pass to future generations is poorer than the one we received, by intuition, many of us would consider this to be unfair toward our descendants. A *future ethics* tries to name and justify obligations toward future people that should constrain the actions of the living[18] (Ott 2003c). It has to take into consideration, first, several *no-obligation arguments* that suggest that present generations have no obligations towards future people at all, and second, how we should deal with risk and uncertainty. For the latter, Ott (2003c) argues for a discourse-oriented risk evaluation and suggests several *criteria of risk evaluation*. The widely accepted *precautionary principle*, for instance, states that "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental damage" (quoted from Ott 2003c, 49f). The *minimax criterion* argues for a policy that minimizes the maximum expected environmental damage due to reflections what a rational agent should do if she is the one who suffers most from what she decides. Eventually, the *avoid-false-positives criterion* states that in order to be safe from disaster we should choose a pessimistic course of action, even though it precludes the possibility of a very good outcome, i.e. we should err on the side of caution.

What we are mainly concerned with here, however, is the structure of the fair bequest package that we leave future generations and that consists of society's productive potential. Two alternatives are possible: either (1) the sum of human-made capital and natural capital or (2) each individual component ought to be

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[17] Herman E. Daly suggests to think of this amount as an index of how far we have moved away from the 'empty world' when marginal utility of natural capital was zero (Daly 1998, 22). For a critical voice cf. Serafy (1998).

[18] A future ethics is rooted in a general ethical theory and includes a concept of (strong) sustainability (Ott 2003c).

sustained (Daly 1996). Option (1) is justified if one assumes that human-made capital and natural capital are substitutes, an assumption of *weak sustainability*. Alternative (2) is advisable if both forms of capital are complements[19], a criteria of *strong sustainability* (Daly 1996, Kopfmüller et al. 2001). Hence, the *substitutability-controversy* is crucial for the distinction of weak sustainability (WS) and strong sustainability (StS) (Neumayer 2003).

Proponents of WS (e.g. Robert Solow, John Hartwick, David Pearce) suppose super-abundance of natural capital or infinite ability to substitute natural capital through human-made capital (Neumayer 2003). It, thus, does not matter which type of capital is sustained. Most important is that the sum of capital will not decline. Asked what to sustain scholars of weak sustainability refer to the utility concept. Each type of capital effects utility and each utility produced by one sort of capital can be set off against utilities effected by other sorts of capital. The so-called *unstructured bequest package* could in theory consist of only human-made capital as long as the utility function is non-declining [*genuine savings* or *genuine investments* according to Neumayer (2003)]. If investments in human-made capital is big enough, an explicit policy of Sustainable Development is not even necessary for then sustainability is guaranteed quasi-automatically.

Proponents of StS (e.g. Carl Folke, Robert Costanza, Herman E. Daly, Charles Perrings, Konrad Ott) argue for a well-structured, highly complex bequest package being composed of all types of capital. Especially natural capital and human-made capital are conceived to be rather complements and their substitution to be limited (Jansson et al. 1994, Daly 1996, 1999, Costanza et al. 2001, Ott 2001, 2003).

One argument for the complementary character is that human-made capital is itself a physical transformation of natural resources that effects from natural capital. In order to produce a bigger amount of the substitute (human-made capital), more of the thing is needed that ought to be substituted (natural capital) – the definition of complements (Daly 1996).

The most telling argument against substitution is related to the *multi-functionality* of ecosystems. If natural capital and human-made capital were perfect substitutes it would be possible to find substitutes for each ecological function and

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[19] To be complements means that the productivity of the human made capital is dependent on the availability of the natural capital (Kopfmüller et al. 2001).

ecosystem service of an ecosystem (e.g. a rain forest, a savannah, a lake). This seems to be rather a futility (Ott 2001, 2003, Kopfmüller et al. 2001).

Additionally, if the process of industrialisation is viewed as the application of human, social and manufactured capital to natural capital to transform it into more human and manufactured capital, then it is possible to view current environmental problems as evidence that such substitutability is not complete (Ekins et al. 2003).

Weak sustainability optimistically and riskily assumes that ecological functions can be substituted for. According to Ott,

> “the precautionary principle, the minimax criterion, and the safe minimum standard shift the burden of proof in cases where stakes are high and uncertainty is prevailing. These concepts justify a prescription to maintain a constant amount of natural capital indefinitely because we cannot be certain whether or which natural entities will be substitutable at any time in the future” (Ott 2003, 62).

Over and above that, if one extends the ethical basis of our behaviour towards natural entities according to *sentientism* [20], obviously strong sustainability is the right choice since this environmental ethic theory includes moral obligations towards “higher” animals within their natural habitats. This part of natural capital can not be substituted due to its moral status as moral patient. This, genuinely moral-ethical reason could be the decisive factor to choose between weak and strong sustainability (Ott 2001).

Eventually, according to Ott (2003), strong sustainability is compatible with market-based economics, a liberal culture, and a democratic state.

Differences between weak and strong sustainability consist in various other controversies. We will not concern ourselves here with the whole discussion on this issue. Neumayer (2003) asserts that, in principle, neither weak nor strong sustainability can get falsified. There is no *experimentum crucis* (Ott 2001). Even though there is nothing like a proof for weak or strong sustainability one can justify the choice for strong sustainability due to good reasons however, i.e. conduct a *discourse-rational choice* (Ott 2001, 2003). Neumayer (2003) concludes that science seems to support WS more with regard to the source side of natural capital and support StS more with the regard to the sink side of natural capital.

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[20] Sentientism represents a position within environmental ethics that argues that only those natural entities have to be taken into consideration which are able to suffer (Frankena 1997, Ott 2003).

Thus, in the following, I will suppose strong sustainability to be the better concept for the sink side of natural capital. There are limits to the substitutability between natural capital and human-made capital. Hence, each type of capital is to maintained intact composing a *highly-structured fair bequest package* [21]. This leads to the claim to keep total natural capital constant, which is coined the *constancy of total natural capital rule*. It is regarded as key idea for sustainability (Costanza & Daly 1992).

How is all this related to the concept of ecosystem resilience, however?

Whether facets of natural capital are substitutable or not is, in my view, fundamental for the relevance of ecosystem resilience in the first place. From the ecological-systemic perspective, ecosystem resilience can be regarded as the *ability of ecosystems to maintain ecosystem services in the face of fluctuating environments and human perturbations*. If human-made capital was a perfect substitute for natural capital, then conservation of natural capital, and thus, the ecosystem resilience of natural systems to maintain ecosystem services would not be of high priority for policy actions. If, however, human-made capital was not a good substitute for natural capital and several facets of natural capital have to be preserved, it would be likely that the ecosystem resilience concept becomes fundamental for the maintenance of natural capital and the provision of ecosystem services.

Thus, strong sustainability can be seen as a *pre-requisite for the political and even moral relevance of the concept of ecosystem resilience*. An important criterion for strong sustainability is the maintenance of natural capital stocks at or above some threshold level (Faucheux & O'Connor 1998). Following this position, ecosystem resilience gets highly relevant. Norton (1995), for instance, considers resilient ecological systems as important feature of the human environment and proposes that any fair bequest package that meets the strong sustainability criterion must include resilient ecological systems. In this respect, Ott (2001, 2003) identifies (ecosystem) resilience as guideline for a Theory of Sustainable Development.

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[21] A society that cares only for the natural capital component, however, also can not be considered as sustainable.

2.3 Ecosystem Resilience & Ecological Economics

Whether one should interpret *Ecological Economics* (EE) as a new start and antithesis against the Neoclassic or rather as continuous development might be a controversial question (Hampicke 1992). Building on several forerunners, for instance Karl Marx and John Stewart Mill, and early pioneers, such as Georgescu-Roegen, Ciriacy-Wantrup and Kapp, the establishment of the international periodical *Ecological Economics* in 1989 provided the institutionalization of the new science. Not only nowadays, EE shows up in being a flowering science and presents a sharp shift of emphasis with regard to epistemological interest, objective of study, and relation to other sciences (Hampicke 1992).

In 1992, a workshop was held in Stockholm following the second international conference of the *International Society for Ecological Economics* (ISEE). A point of consensus found among ecological economists during this workshop is the *pre-analytic vision of the macro-economy* which was originally put forward by Herman E. Daly (Daly 1996, cf. Folke et al. 1994). It pictures the macro-economy as an open subsystem of the finite natural ecosystem (Perrings et al. 1995b, Daly 1996, Faucheux & O’Connor 1998) which “is totally dependent on it, both as a source for inputs of low entropy matter/energy and as a sink for outputs of high-entropy matter/energy” (Daly 1996, 48). Costanza et al. (2001) consider this insight to be one of the three fundamental elements of Ecological Economics. Implied are binding ecological limits to growth both to the use of nature as a source and as a sink (Daly 1996, Costanza et al. 2001), simply because earth is finite, entropy must increase, and ecological systems are dependent on each other (Daly 1999).

This signifies at the same time the third point of consensus found during the workshop, mentioned above, i.e. there are limits to biophysical throughput passing the economic system (Folke et al. 1994, Arrow et al. 1995)[22]. O’Connor (1998) conceives the notion of biophysical constraints as leit motif of the limits to growth literature.

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[22] It is, of course, possible that improvements in the management of resource systems, accompanied by resource-conserving structural changes in the economy, would enable economic and population growth to take place despite the finiteness of the environmental resource base but principally this resource base is finite (Arrow et al. 1995).

If it is right that there are biophysical limits to human growth, a *sustainable scale* of human impact (measured by population times per capita resource use) seems admirable (Ehrlich 1994, Daly 1996) out of ethical and prudential[23] reasons. Costanza et al. (2001) conceive the specification of this scale of human impact as to be one of the three fundamental goals of Ecological Economics, along with a fair distribution and an efficient allocation. Hereby, the size of human impact on Earth's ecosystems has to be identified before searching for a fair distribution and an optimal allocation[24]. Costanza et al. (2001) suggest that the work on a sustainable scale of human impact represents the main difference to *Neoclassical Economics* [25]. This scale is again to be met both for the amount of resources which earth's population exploits as well as for earth's ability to cope with human wastes and emissions.

Herman E. Daly compares the sustainable scale of human activity with the Plimsoll line – the maximum absolute weight on a boat (Daly 1996). Natural and social scientists differ in their assessment, however, whether the actual scale of human activity is approaching the "Plimsoll line" or not. Some of them state that human activity is already overshooting the sustainable scale which is termed the *full world hypothesis*.

Costanza et al. (2001), for instance, interpret several environmental problems, such as climate change, the depletion of the ozone layer, loss of fertile soils or declining biodiversity, as hints that humanity is already approaching the Plimsoll line. Also Folke et al. (1994) conclude that the world economy and human population have reached magnitudes at which the effects of its various activities can no longer be absorbed by ecosystems without significant changes and better adjustments of its material flows to the biogeochemical cycle of the biosphere.

Macro-economically, the scale of human activity is expressed as growth of Gross National Product (GNP). Referring to ecological economists, macroeconomic growth can become uneconomic. Uneconomic growth increases environmental and social costs by more than it increases production benefits. Macroeconomic scales of human activity exist corresponding to the microeconomic scale beyond which the

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[23] As Gretchen C. Daily put it: "Unless humanity is suicidal, it should want to preserve, at the minimum, the natural life-support systems and processes required to sustain its own existence" (Daily 1997, 365).

[24] The sustainable scale would be smaller for sentientism as for personalism, hence, depends in part on ethical judgements (Daly 1996).

[25] The opinions differ on this question. For Acker Widmaier (1999), the main difference between Ecological Economics and Neoclassical Economics is the focus of the former on distributive justice and the primacy of the latter on Pareto-efficiency.

benefits of further growth equals the marginal costs to human beings of sacrificed natural capital[26] (Daly 1996). This point is made by Max-Neef who suggests that

> “for every society there seems to be a period in which economic growth (as conventionally measured) brings about an improvement in the quality of life, but only up to a point – the threshold point – beyond which, if there is more economic growth, quality of life may begin to deteriorate” (cited in Max-Neef 1995, 117, originally formulated in Max-Neef 1991)

which is dubbed the *threshold hypothesis*.

At the same time, the *Index of Sustainable Economic Welfare* (ISEW) was published for several Northern countries[27] showing that in all countries studied so far, economic welfare per capita rose in the early parts of the study period (mostly 1950 -1970) and then began to decline despite continued growth of per capita GNP (Max-Neef 1995)[28]. This “may reveal the existence of a point in a country’s economic evolution where quantitative growth must be metamorphosed into qualitative development” (Max-Neef 1995, 117)[29]. This economic argument illustrates that it is not only environmentally but also economically useful to propose an optimal scale. Following Max-Neef (1995), this scale is already reached in several high income countries which, in turn, supports the full world hypothesis.

Critics of the full world hypothesis and supporters of Weak Sustainability, respectively, suggest that general development occurs in accordance with an inverted U-shaped relation between per capita incomes and some types of pollutants, a hypothesis dubbed *Environmental-Kuznets-Curve* (EKC). It proposes that in the earlier stages of economic development, increased environmental problems are regarded as an acceptable side effect of economic growth. However, when a country has attained a sufficiently high standard of living, people give greater

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[26] For microeconomics this scale is the point where increasing marginal costs equals declining marginal benefit, and beyond which further growth in the activity would be uneconomic because it would increase costs more than the benefits. The point when growth ought to end microeconomically is labelled *when-to-stop-rule*.

[27] Daly & Cobb published the ISEW for the USA in 1990, Jackson & Marks for the United Kingdom, Diefenbacher for Germany, Obermayr for Austria, and Jesperson for Denmark, each in 1994, and Rosenberg & Oegema for the Netherlands in 1995.

[28] How quality of life or economic welfare per capita GNP should be estimated is surely a controversial question. The attempts of raising the ISEW, however, presents so far the most comprehensive effort of its kind to challenge the GNP as a significant measure (Max-Neef 1995).

[29] Note hereby that Daly (1996) distinguishes *development*, i.e. more utility per unit of throughput from *growth*, meaning more throughput.

attention to environmental amenities. This leads to environmental legislation, new institutions for the protection of the environment, and so forth.

Supporters of strong sustainability reply, first, that economic growth may lead to improvements in some environmental indicators, but that this implies neither that economic growth is sufficient to induce environmental improvement in general, nor that the environmental effects of growth will not be ignored, nor, indeed, that the Earth's resource base is capable of supporting indefinite economic growth. Additionally, the inverted-U-relations have been uncovered for emissions of pollutants, not resource stocks (Arrow et al. 1995) and misses larger issues of impacts on environmental capital and global pollutants[30] (Farber 1995). Conceivable are nightmare scenarios according to which societies get stuck at the peak of environmental degradation rates or create irreversible damage to natural stocks (Ott 2001)[31]. The EKC presupposes that environmental objectives are met, the wounds of nature will be healed and the loss of species and habitats is not decisive - assumptions which are rather doubtful (Ott 2001).

Considering the environmental as well as the economical argument discussed above the full world hypothesis seems to be justified. Therefore, it would appear that natural capital in a full world is scarce compared to human-made capital.

There is an important point here.

If the substitution of natural capital and human-made capital is limited and natural capital is scarce, indeed, it will be justified - in accordance with economic theory - to invest in the scarce type of capital that is needed for production. Hence, it is *economically justified to invest in natural capital* (Daly 1996).

Thus, the full world hypothesis which nourishes strong sustainability supports the claim to invest in several facets of natural capital, i.e. its provision of resources, its wastes-reception services, its maintenance of life-support functions and amenities.

However, the need of investing in natural capital begs the question in which of the facets of the natural capital stock society should invest. Reflecting on this question, ecological economists developed the concept of *critical natural capital* (CNC). Critical natural capital is often defined provisionally as "that part of the natural

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[30] We see wealthy countries such as the US being unwilling to make substantive sacrifices for global environmental quality.

[31] High-income countries could substitute environmental degradation in poorer countries for their own degradation in the process of economic growth. This is illustrated most clearly by Japan's rapid economic growth and its devastation of forested areas in Southeast Asia (Farber 1995).

environment that performs important and irreplaceable functions" (Ekins et al. 2003, 161). CNC is, thus, non-substitutable in respect of the function in question (Ekins 2003)[32].

The CRITINC research project established by the European Commission tried to clarify in what way and to what extent natural capital is critical in societal development and how society can capture criticality to make better decisions (Ekins et al. 2003). The contributors from various European countries consider CNC to be one way to put the strong sustainability principle into concrete terms.

Natural ecosystems are defined by a number of *ecosystem characteristics* (e.g. climatological processes, water quality, geological processes, vegetation characteristics) that, in turn, determine the ecosystems' capacity to provide *ecosystem goods* (resources that are usually provided by the ecosystem components) and *ecosystem services* (usually provided by the ecosystem processes).

Environmental functions are defined as "the capacity of natural processes and components to provide goods and services that satisfy human needs (directly and/or indirectly)" (Ekins et al. 2003, 169). Environmental functions, in turn, are divided into four categories, namely *regulation functions*, *production functions*, *habitat functions*, and *information functions*.

Underlying the categorisations of environmental functions an important distinction may be drawn between *functions-of natural capital* and *functions-for humans* (Ekins 2003). Functions-for on the one hand are those environmental functions which provide direct benefits for humans, i.e. ecosystem goods and services.

Daily's term *ecosystem services* originally comprised goods and services (Daily 1997). Daily defines ecosystem services as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" (Daily 1997, 3) which include purification of air and water, mitigation of floods and drought, detoxification and decomposition of wastes, generation and renewal of soil and soil fertility, pollination of crops and natural vegetation, to name but a few.

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[32] Noel & O'Connor define critical natural capital as „that set of environmental resources which, at a prescribed geographical scale, performs important environmental functions and for which no substitute in terms of manufactured, human or other natural capital currently exist" (Noel & O'Connor 1998, 78).

Functions-of natural capital (regulation functions or life-support functions) are those which maintain the basic integrity of ecosystems (Ekins 2003). Functions-of are not easily perceived, and scientific knowledge about them is still uncertain and incomplete. They are, however, a prerequisite for the continued performance of many functions-for humans. As Ekins et al. point out: “Without the ‘functions of’ natural capital, no other category of functions would be able to exist on a sustained and systematic basis” (Ekins et al. 2003, 170). The functions-of are constituted by the basic processes and cycles in the internal functioning of natural systems, which are responsible for sustaining and maintaining the stability and resilience of ecosystems (Ekins et al. 2003, similar Holling et al. 1995), hence, for the basic life support for humans.

I would like to draw attention to the fact that the of-what part of an ecosystem resilience analysis considers the ecosystem services of concern before examining underlying variables that are a prerequisite for the maintenance of ecosystem services (Carpenter et al. 2001, Walker et al. 2002, cf. section 4.2). Thus, the distinction of functions-for and functions-of is highly relevant for the ecosystem resilience concept.

Concerning CNC a *safe minimum standard* is recommended in order to avoid the *critical zone*, i.e. “those physical conditions brought about by human action, which would make it uneconomical to halt and reverse depletion” (Ekins et al. 2003, 174). The critical zone can be thought of as “the threshold , the passing of which may flip an ecosystem into another stability domain” (Ekins et al. 2003, 174). As we will explore in section 3.2.3, the ecosystem resilience concept assumes alternative *basins of attraction* to be common in nature. Ecosystems shift to another stable regime when they pass a certain threshold. The CRITINC project, thus, uses the ecosystem resilience concept in order to operationalize CNC. In this sense, Ekins et al. (2003) suggest to avoid the critical zone by building buffer capacity and resilience.

Additionally, following Deutsch, Folke & Skanberg (2003), CNC can be addressed from two angles. (1) CNC in relation to environmental functions for human well-being, i.e. ecosystem services *sensu* Daily (1997) and (2) CNC in relation to the performance of life-support functions of ecosystems, i.e. the *capacity of an ecosystem to maintain services in the face of a fluctuating environment and human perturbations* (the *ecological-systemic sense* of ecosystem resilience, cf. section 3.4.3). In this respect, Ekins & Simon suggest that

> “what matters about the environment is not particular stocks of natural capital per se, but the ability of the capita stock as a whole to be able to continue to perform the environmental functions which make an important contribution to human welfare” (Ekins & Simon 2003, 255f).

According to Deutsch, Folke & Skanberg (2003), the ecosystem resilience concept provides a framework for analyzing the performance and maintenance of ecosystem functioning.

The point I want to make here is to stress the close connection of the concept of ecosystem resilience to the CNC concept. The point of contact is represented by the regulation functions (the *functions-of* natural capital) that are maintained by the capacity of ecosystem resilience. Additionally, the ecosystem resilience concept provides an operational tool in order to unfold the mechanism that are responsible for the regulation functions which links to the focus on *slow variables* of an *ecosystem resilience analysis* (cf. section 3.2.4 and 4.2.3.3).

The need to maintain or invest in natural capital gets obvious at this point. Faucheux & O’Connor (1998) consider the maintenance of specified critical natural capitals above thresholds needed to maintain vital services and functions as the *strong criterion for sustainability* (cf. also Perrings et al. 1995).

3 Ecological Aspects of Ecosystem Resilience Theory

In the previous chapter I have suggested some connections of the ecosystem resilience concept to notions and conceptions such as limits to growth and strong sustainability and to sciences that are relevant within sustainability discourse, such as Ecological Economics.

Along with these findings that point to the relevance of the concept of ecosystem resilience within the *argumentative space of sustainability discourse*, various authors regard ecosystem resilience as a hopeful and important concept. Perrings et al. (1995) identify the ecology and economy of resilience as one of the key research issues of the future. A group of highly acknowledged ecologists and ecological economists regard ecosystem resilience as an useful index of environmental sustainability and conclude that “economic activities are sustainable only if the life-support ecosystems upon which they depend are resilient” (Arrow et al. 1995, 93). Levin points out that further “research needs to address whether or not our vital systems have sufficient capacity for resilience” (Levin et al. 1998). Both Ott (2001) and Kopfmüller et al. (2001) do not doubt the theoretical relevance of the ecosystem resilience concept and are interested in the possibilities to operationalize it. Folke, Holling & Perrings (1996) state that in order to provide essential ecosystem services for human development in the face of a fluctuating environment the maintenance of a level of biodiversity is to be ensured that will guarantee the ecosystem resilience of ecosystems that sustain human societies. Perrings (2002) regards ecosystem resilience as a clear and operational concept of sustainability.

The research questions put forward by the ecosystem resilience concept lead to an immense dialogue dubbed the *Resilience Project*, a five-year collaboration among an international group of ecological economists, ecologists, social scientists, and mathematicians labelled *Resilience Alliance* [33]. “The project was initiated to

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[33] Facing environmental problems on a global scale, the failure of resource management, and the limited capability of ecological science to provide solutions, ecologists, economists and social scientists joined and established a research organization dubbed the *Resilience Alliance*. The *Resilience Alliance* represents a multidisciplinary research group that explores the dynamics of complex *social-ecological systems* (SESs) in order to discover foundations for sustainability. Established in 1999, the *Resilience Alliance* is supported by an international network of member institutions that includes universities, government, and non-government agencies. Influential members are, for instance: Brian Walker, working at CSIRO, Australia, Lance Gunderson, chair of the Department of Environmental Studies at Emory University in Atlanta, USA, Stephen Carpenter, Halverson Professor of Limnology and Professor of Zoology at the University of Wisonsin-Madison, USA, Carl

search for an integrative theory and integrative examples of practice. Its goal was to develop and test the elements of an integrative theory that had the degree of simplicity necessary for understanding but also the complexity required to develop policy for sustainability" (Holling 2001, 391). The *Resilience Alliance* represents the main authority in the field of ecosystem resilience and encourages new forms of analyses for stability properties and adaptive environmental management.

After some conceptual clarifications and preliminaries which include the definition of resilience (section 3.1), the subsequent sections will explore the background theory of the concept of ecosystem resilience (section 3.2), before considering the mechanisms that are responsible for ecosystem resilience (section 3.3) and, finally, discussing some theoretical features of the ecosystem resilience concept entirely (section 3.4).

Folke, scholar at Stockholm University, Sweden, Garry D. Peterson, lecturer at the Centre for Limnology at the University of Wisconsin, USA, and many others but in particular the outstanding scholar Crawford S. (Buzz) Holling, Emeritus Eminent Scholar and Professor in Ecological Sciences at the University of Florida, USA, who stimulated ecological research on (ecosystem) resilience and complex systems in many ways. The declared scientific aim of the *Resilience Alliance* is to improve understanding of the dynamics of social-ecological systems with a view to learning how to effectively influence their ecosystem resilience, *adaptability*, and *transformability*. Their hope is that this knowledge will provide the basis for improved governance and management of natural systems such that their capacity to sustain human and natural capital is enhanced. In order to disseminate a growing body of knowledge surrounding ecosystem resilience in complex SESs the *Resilience Alliance* established an Internet Portal (http://www.resalliance.org) and publishes a peer-reviewed, electronic journal, named *Ecology and Society* (former: *Conservation Ecology*) which is available at http://www.ecologyandsociety.org . In my view, the *Resilience Alliance* represents the main authority in the latest research on ecosystem resilience and on related concepts. The main body of literature that I used for this thesis is derived from the publications of the *Resilience Alliance*.

3.1 Conceptual Clarifications and Preliminaries

3.1.1 Definitions

Definitions are formulated in order to specify the meaning of a given term (Jax 2002). A classical question within theory of science is whether the definition of a term reflects reality *per se*, hence, something that is "found" in nature, or whether definitions represent arbitrary human specifications. '*Realdefinitionen*' are statements about the essence of natural phenomena, about reality *per se*, whereas '*Nominaldefinitionen*' represent genuinely linguistic arrangements, i.e. conventions. Within theory of science, *Realdefinitionen* are not conceived as relevant anymore (Jax 2002). Eventually, a *Nominaldefinition* is not conceived to be entirely arbitrary. People always have certain preconceptions with respect to a given term. Hempel (1952) introduced the analysis of meaning, which represents the attempt to grasp verbally in the most precise form possible, the accepted meaning, i.e. the widespread use, of a given term. Therefore, there are no universally agreed on definitions for a given term, but rather definitions that are appropriate for a certain objective (Jax 2002). The definitions proposed in the following must be understood against the background of these reflections.

This position likewise results in an *epistemological approach* towards the perception of ecological units. Ecological *units* (e.g. communities, ecosystems) are not something that is given in nature and can be "found" and identified. Rather, the observer constructs ecological units due to a given objective (Jax 2002). However, still notions of ecological units can be more or less accurate and, therefore, a relation to nature is needed. This discussion refers to theory of science issues and it is well beyond this thesis to treat them in detail.

3.1.2 Relevance of Concepts in Ecology

Natural scientists are confronted with the anything but simple complexity of *nature*[34]. Their descriptions, investigations and experiments provide a vast amount of data that always can be interpreted in several ways (Edwards 1967, cf. Mutschler 2002). Even the simplest ecological systems have at least a few different possible descriptions (Grimm & Wissel 1997). Moreover, biological sciences at the level of organisms and above are plagued by multiple causation. It can be difficult if not impossible to find the fundamental cause of an interaction within a complex system. Several theories might be necessary in order to explain a certain phenomena or process (Paine 2002). Moreover, every process that occurs in living organisms is the result of two distinct sorts of causes, (1) the *functional cause* (how? – *physiological explanations*) and (2) the *evolutionary cause* (why? – *historical explanations*) (Mayr 2000).

Facing these difficulties, scientists construct concepts “to narrow down the never-ending tally of possible questions that we empirically or theoretically ask of nature” (Grimm & Wissel 1997). The role of concepts in the progress of biological sciences, however, is seen as controversial among biologists (cf. Paine 2000 and Naeem 2002). The highly acknowledged biologist Ernst Mayr states that most of the progress in more complex biological sciences (ecology, behavioural biology, evolutionary biology) is due to the development of new concepts (Mayr 2000). Paine (2002) likewise considers conceptual evolution to be the driving force in the progress of ecological science.

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[34] The meaning of the term *nature* is not straightforward (Mutschler 2002). The majority of natural and social scientists draw a line between culture (the human system) and nature (all natural entities except humans). Other scholars, however, do not accept this distinction and regard it as artificial. I will use the term nature along the former distinction. Agreeing with this distinction the philosopher Hans-Dieter Mutschler states that not only the human realm is not entirely explainable by natural sciences (most scholars agree with this position) but neither is the natural realm (most scholars disagree with this position). A plurality of methods is needed to grasp the full amount of phenomena (Mutschler 2002). This distinction gets relevant when we will consider the human system as opposed to the natural system. The *Resilience Alliance* suggests to consider the human system to be a part of the whole ecosystem. The distinction of culture and nature still counts from a philosophical perspective even though human land use is conceived as part of the dynamics of social-ecological systems when considering environmental management (cf. section 4.1.2).

3.1.3 Stability Properties

In ecology, a vast amount of terminology and concepts emerged in the face of the idea of *stability of something* within the natural system[35]. Stability is seen as a meta-concept that covers a range of different properties or components (Loreau et al. 2002) which are made explicit more or less by several *stability concepts* within ecological literature. Thus, in order to understand the meaning of the term *ecosystem resilience* it is, first of all, important to delimit ecosystem resilience from the meaning of these distinct stability concepts.

Grimm, Schmidt & Wissel (1992) count 163 definitions from 70 different stability concepts and more than 40 measures. The key question is: Do these various stability concepts reflect real different properties of natural systems or entities? In their sound inventory of stability concepts *Babel, or the ecological stability discussion*, Grimm & Wissel (1997) (G&W) 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*.

Note that properties (1) and (2) refer to a certain reference state or dynamic whereas property (3) corresponds to whole ecosystems. Persistence, therefore, represents a *holistic* and *qualitative* concept.

> “[H]olistic, because it refers to entire systems by definition, whereas constancy refers to dynamics of certain variables of interest; and qualitative, because the focus is no longer on dynamics with its quantitative details, but on the qualitative question of whether the whole set of variables of interest used to characterize the system remains within certain boundaries” (Grimm et al. 1999, 230).

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

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[35] The term *stability* is used not only in ecology but likewise in many other natural (physics, chemistry) and social sciences (Hansson & Helgesson 2003).

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 G&W, Hansson and Helgesson (2003) (H&H) identify three basic stability properties in the relevant literature of both natural and social sciences while using a different terminology compared to that of G&W. 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* [36].

There is an important point here and that is the distinction between "a measure what actually happens in a system" (H&H's constancy) and "the tendency of a system to remain unchanged" (H&H's robustness). This distinction between *descriptive stability concepts* that stress what actually happens in a system and *analytical stability concepts* that stress the *ability* (or capacity, or power) to remain unchanged (or withstand, resist) when exposed to disturbances is fundamental for a proper understanding of the ecosystem resilience concept. To be useful for a Theory of Sustainable Development, to my view, the ecosystem resilience concept must refer to the underlying *key* or *slow variables* of whole ecosystems on a landscape scale. Therefore, Grimm & Wissel's (1997) *persistence* will be important in the following.

Hansson & Helgesson (2003) go one step further in proposing that robustness (*property 2*) is a limiting case of resilience (*property 3*). Intuitively robustness and resilience are separate properties of a given system. In a formal representation, however, robustness can be treated as a special case of resilience, namely when the return time of the system to its reference state is zero. Therefore, H&H state that there are only two fundamentally different stability properties that

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[36] Note that both G&W and H&H identify the term *resilience* for the same property, namely, to return to a reference state after a certain perturbation, while H&H stress more the ability of a system to perform this property. Note also that, in turn, the *Resilience Alliance* does not agree with this terminology.

cannot be defined in terms of the other – constancy and resilience. However, the distinction between robustness and resilience and the emphasis on robustness or persistence, respectively, remains fundamental for several reasons (cf. section 3.4.2).

Table 2: Terminology within stability talk
(according to Grimm & Wissel 1997)

Terminology	Content
Stability term	Description of property
Stability definition	Definition of a property
Stability concept	Term + definition
Stability measure	Measure of the quantitative assessment of a property
Stability statement	Assessments of stability properties with the aid of stability measures
Stability mechanism	Mechanism that is responsible for stability properties

The term *stability*, thus, functions only as a common qualifier for the three basic properties or, seen more broadly, for the six properties identified by Grimm & Wissel (1997). Stability itself is not a property but a terminological link that emphasizes the close connection between the three fundamental properties[37]. According to Grimm & Wissel (1997) it would be better for the progress in ecological science to delete the descriptions *stable*, *stabilising* and *stability* as individual expressions completely from the ecological vocabulary. The authors regard this claim as naïve, however, and propose to use “stability” (in quotation marks) as a short form or substitute for stability properties. In order to provide a common terminology, G&W suggest the following terms for being able to communicate clearly which are summarized in *Table 2*.

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[37] This analytical separation of stability properties is artificial. In reality, all three properties influence each other. The nature of this relation depends on the concrete ecological situation (Grimm & Wissel 1997). In some models mentioned below it seems that the fundamental stability properties cannot be separated properly (Ludwig, Walker & Holling 1997, 2002) which is reflected in the *extended-ecological definition* of ecosystem resilience by Walker et al. (2004) (cf. section 3.4.3). However, the distinction of the stability concepts is still useful for methodological and analytical purposes in order to make explicit what is going to be examined within scientific investigations.

> “A “stability definition” is the definition of a stability property and a “stability term” is the description of this property. Stability terms and definitions together form the “stability concept”. A “stability measure” is a measure for the quantitative assessment of a stability property. (…) The results of the use of stability concepts in ecology are “stability statements”, i.e. assessments of stability properties with the aid of stability measures. Finally, by “stability mechanisms” we mean the mechanisms that are responsible for certain stability properties” (Grimm & Wissel 1997, 327f).

The important point here is that scientists have to refer explicitly to one of the fundamental stability properties (constancy, resilience, robustness or persistence) when proposing their descriptions and experiments. According to Grimm & Wissel (1997), the ignorance of this precision represents one of the three sources of confusion within stability research.

It is important to understand clearly all the sources of confusion within stability debate in order to be able to overcome them. Beside the terminological confusion there are two other sources (Grimm & Wissel 1997). The second source lies in the fact that even the simplest ecological systems are so complex that stability concepts always can be applied to many different possible variables and reference states (such as biomass, diversity, nutrient pools, community structure, certain populations) on different spatial and temporal scales as well as to different disturbances (cf. also Loreau et al. 2002). Ashby points out:

> “every situation in nature can be described in an infinite number of ways. An observer can choose any arbitrary set of variables and parameters to define an abstract system, which then can be the object of study by empirical and theoretical methods” (cited from Grimm & Wissel 1997, 328).

Referring to this problem Jax, Jones & Pickett (1998) and Jax (2002), respectively, and in accordance with them, Grimm (1998) make an important point. For stability properties to be useful one must identify what exactly is being “stable”. An *ecological unit* (e.g. community, ecosystem) has no meaning unless it is defined and specified (Grimm 1998, Jax 2002). Jax, Jones & Pickett (1998) suggest the term *self-identity* of ecological systems which comprises four dimensions that constitute the minimum of what has to be identified and specified if identity or change are to be assessed. The first dimension is whether the unit is defined topographically or functionally; the second dimension is the degree of expected internal relationships.

Hereby, the first two dimensions deal with the general definition of an unit while the third and fourth dimensions specify the level of abstraction at which the elements of an ecological unit are perceived. The third dimension is the set of selected phenomena or elements of an ecological unit. Eventually, the fourth dimension is the resolution (or degree of aggregation) under which the selected elements are perceived (Grimm 1998). The last three dimensions form a three-dimensional system of coordinates within which the definition of any ecological unit can be located as a point or small volume (Jax, Jones & Pickett 1998, Jax 2002).

Additionally, stability statements can only relate to particular *ecological situations* (Grimm & Wissel 1997). As soon as the situation changes (e.g. consideration of another state variable, different disturbances, different reference states), the stability statement is, *ipso facto*, no longer valid. In order to address this issue, Grimm & Wissel (1997) introduce an *ecological checklist* [38] that consists of the features that are most important for characterising an ecological situation: (1) *level of description* (individual, population, community, ecosystem, landscape), (2) *variable of interest* (e.g. biomass, population size, spatial structures), (3) reference state, or *reference dynamic*, respectively (e.g. equilibrium, cycles, configurations), (4) *disturbance* (disturbance of the state variable or of a system parameter, lasting disturbance or short term disturbance, frequency of disturbance), (5) *spatial scale*, and (6) *temporal scale* (e.g. time horizon of the statement)[39]. If only one of the features is changed this results in a new ecological situation and the old stability statements will, in general, no longer be valid. If one accepts the ecological checklist general assessments of the “stability” or ecosystem resilience of an entire ecosystem are considered to be impossible (Grimm et al. 1999, cf. also Walker & Abel 2002).

The concept of self-identity *sensu* Jax, Jones & Pickett (1998) and the ecological checklist *sensu* Grimm & Wissel (1997) represent tools (cf. *Table 3*) that will be used in the following for examining critically the concept of ecosystem resilience as well as related concepts. Stability statements to be useful *must* refer to an identified ecological unit and a particular ecological situation.

The distinction of the three fundamental stability concepts, the features of the ecological checklist, and the types of ecosystems create a large matrix of

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[38] The main reason that necessitates a checklist is that stability concepts were not invented for the investigation of ecological systems but for the characterisation of simple dynamic systems (Grimm & Wissel 1997).

[39] Specification of temporal and spatial scale is conceived as fundamental in ecology. In many cases, the human life span is implicitly or explicitly regarded as a definitive norm. Hence, the choice of scale often reflects certain human interests or objectives (Gorke 1996).

potential combinations. First, the relation of the three fundamental stability concepts to distinct reference states and variables may be different (Loreau et al. 2002). Second, stability statements are only valid for a certain temporal and spatial scale. Third, various disturbances might affect the different stability concepts in a different way (Grimm & Wissel 1997). Fourth, scientific results from one ecosystem do not necessarily inform us about other systems (Bengtsson et al. 2002). In my view, awareness of this matrix and of the temptation to generalize scientific results, is fundamental to avoid future confusion as far as stability statements are concerned.

Moreover, stability properties can only be perceived on a certain level of description and have no direct meaning at lower levels of description (Grimm et al. 1999). They can be considered as *emergent properties*, i.e. properties that emerge on higher levels of the ecological hierarchy and can not be explained solely by the entities on lower levels. Some authors doubt the existence of emergent properties. A distinction must be drawn between *collective properties* and emergent properties. Collective properties are the sum of properties of a system (e.g. the total biomass). Emergent properties are more than the sum of its parts (cf. Dittmann & Grimm 1999).

Table 3: Critical tools for stability statements
(according to Jax, Jones & Pickett 1998 and Grimm & Wissel 1997)

Tool	Content	Reference
Self-identity of ecological units	(1) defined topographically or functionally (2) degree of expected internal relationships (3) set of selected phenomena or elements (4) resolution	Jax, Jones & Pickett (1998); Jax (2002)
Ecological checklist	(1) level of description (2) variable of interest (3) reference dynamic (4) disturbance (5) spatial scale (6) temporal scale	Grimm & Wissel (1997)

3.1.4 Definition of Resilience

It has been almost three decades since the term *resilience* as a concept distinct to other stability concepts was first established and defined by C.S. Holling in his influential paper *Resilience and stability of ecological systems* (Holling 1973)[40].

Etymologically, the term resilience was formed in English on the model of Latin *resilientem* (nominative *resiliēns*), present participle of *resilīre* which means to rebound or to recoil (Barnhart 1995).

Resilience in its *original-ecological sense* [41] has been defined in two different ways in the ecological literature (cf. Holling 1986, Holling et al. 1995, Gunderson 2001, Gunderson & Holling 2002, Gunderson & Pritchard 2002). There is no right or wrong use of the term. Rather, the different usage emphasizes two distinct stability properties.

The first definition (1) concentrates on stability near an *equilibrium steady state* [42], where the rate and speed of return to pre-existing conditions after a disturbance event are used to measure the property (deAngelis 1980, Pimm 1984, Tilman & Downing 1994, WBGU 2000, Lugo et al. 2002). Resilience is then defined as the time required for a system to return to a steady state following a disturbance[43]. This definition matches the etymological meaning of the term resilience.

The second definition (2) emphasizes conditions far from any equilibrium steady state, where instabilities can shift a system to another *basin of attraction* [44] which is controlled by a different set of variables and characterized by a different structure (Holling 1973, 2001, Gunderson 2001, Gunderson & Pritchard Jr. 2002, Holling & Gunderson 2002, Walker et al. 2002, 2004). Resilience, understood in this way, is the “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior” (Gunderson & Holling 2002, 4).

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[40] Some authors name Walker (1969) as first source. I cannot use this article as I could not find the reference source.

[41] Along with the original-ecological meaning, the term *ecosystem resilience* exhibits multiple meanings within the relevant literature (cf. section 3.4.3).

[42] “A mechanical system is at equilibrium if the forces acting on it are in balance (…). The “balance of nature” (Pimm 1991) is an extension of this idea to the natural world. The concept usually refers to steady flows of energy and materials, rather than to a system whose components do not change” (Ludwig, Walker & Holling 1997, 2).

[43] This stability concept also has been frequently used in Neoclassical Economics.

[44] A stability basin is part of a *stability landscape* and represents the region around an equilibrium where the regime of an ecosystem is likely to persist despite small disturbance events (cf. section 3.2.3).

Note that both definitions of resilience use several concepts that are hotly contested among ecologists. For instance, the first definition implies an assumption of *global stability*, i.e. an ecosystem has only one equilibrium or steady state and the focus of study is on slow dynamics near this stable equilibrium (Ludwig, Walker & Holling 1997). The second definition presupposes the existence of *alternative stable regimes*, i.e. ecological systems can exhibit a shift from one regime to another that is controlled by a different set of variables (Holling 1973, 1986, Carpenter 2001, Muradian 2001, Scheffer et al. 2001, Scheffer & Carpenter 2003, Walker & Meyers 2004). Focus of interest are slow dynamics in a region that separates two alternative stable basins. In order to understand the ecosystem resilience concept entirely we have to go into these concepts in a more profound way (cf. section 3.2).

The terminology of Hansson & Helgeson (H&H) (2003) provides a good tool in order to distinct the two stability properties which are associated with the term *resilience*. Definition (1) corresponds to the property resilience which is defined as “tendency of a system to recover or return to (or close to) its original state after a perturbation” (ibid, 222). Definition (2) matches the property “robustness” or the “tendency of a system to remain unchanged, or nearly unchanged, when exposed to perturbations” (ibid, 221). Hence, the two different definitions of the resilience concept reflect *two different properties* of ecological systems: resilience and robustness or persistence, respectively. Both definitions are contrastive aspects of the common qualifier “stability”. Whether they are related to each other in a close or loose manner depends on the concrete situation (Grimm & Wissel 1997).

In the *Dictionary of Ecology, Evolution and Systematics* (Lincoln, Boxshall & Clark 1998) the suggested definitions for *stability* and *resilience* both reflect the different usage of resilience and the confusion of two or even more distinct stability properties. The authors define stability as “resistance to change; tendency to remain in, or return to, an equilibrium state; the ability of populations to withstand perturbations without marked changes in composition” (ibid, 283) and resilience as “the ability of a community to return to a former state after exogenous disturbance; the capacity to continue functioning after perturbation” (ibid, 261). In this connection, aspects of robustness and resilience [both *sensu* H&H] are intermixed. It is important to separate stability properties meticulously in order to be able to communicate clearly.

The distinction of the two definitions for resilience is also highlighted by the *Resilience Alliance*. The research group uses a different terminology and provides the term *engineering resilience* for the property resilience and the term *ecosystem resilience* or *ecological resilience* for the stability property robustness [the latter both according to H&H] (in the following: ecosystem resilience). The term *vulnerability* is used, in turn, as an antonym for ecosystem resilience, i.e. the “propensity of ecological systems to suffer harm from exposure to external stress and shocks” (Folke et al. 2002, 5).

These terms (cf. *Table 4*) get crucial within this thesis since the *Resilience Alliance* represents the main authority within resilience debate, which, in my view, necessitates not to ignore their terminology entirely. In the following I will use the terms *engineering resilience* and *ecosystem resilience* while referring also to the terminology of Grimm & Wissel (1997) and Hansson & Helgesson (2003). Note that the term *vulnerability* gets important when we will consider the concept of ecosystem resilience within the framework of *sustainability science* (e.g. Kates et al. 2001, Turner et al. 2003, cf. section 4.3.3).

Table 4: Terminology used by the Resilience Alliance
(according to Holling & Gunderson 2002)

Stability term	**Definition**	**Reference**
Ecosystem resilience	Magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour	Holling & Gunderson (2002)
Engineering resilience	Rate and speed of return to pre-existing conditions after disturbance	Holling & Gunderson (2002)

Both resilience definitions refer to the term *disturbance*. Disturbance represents an important and widespread phenomena in nature which is considered as an ecologically significant object for study in itself (Pickett & White 1985). Ecosystems cannot be seen as static entities, rather, they represent always-changing, fluctuating, dynamic systems (Reichholf 1998). There is no balance of nature, rather endless change and the ongoing creation of novelty are the rule (Carpenter & Turner 2001). In reference to Jax, Jones & Pickett (1998), Grimm

(1998) speaks of *intermediate self-identity* since certain parts of ecosystems do always change whereas other parts do not.

Following White & Pickett (1985), Paine et al. (1998), Turner & Dale (1998) Dittmann & Grimm (1999) and Colding, Elmqvist & Olsson (2003) define *disturbance*, rather general, as “any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment” (quoted from White & Pickett 1985, 7).

The definition of White & Pickett includes environmental fluctuations and destructive events, whether or not these are perceived as “normal” for a particular system. However, a distinction between *natural small-scale disturbances* and *human large-scale disturbances* tries to delimit disturbances that are considered to be part of the system from others that are superimposed on the system. This distinction gets important when relatively small disturbances are conceived as being an integral part of the ecosystem dynamics. Small-scale disturbances as integral parts of ecosystems are fundamental for the generation of ecosystem resilience (Holling 1986, cf. section 3.3.3.5).

Bengtsson et al. (2003) suggest that ecosystems are subject to pulse disturbances at various spatial and temporal scales. The point of interaction between the disturbance force and the ecosystem is termed *biotic interface* (Lugo et al. 2002). The organisms that occupy these interfaces are most exposed to the disturbance and absorb most of the impact.

The term *perturbation* is used, if (1) the parameters or behaviours that define a system have been explicitly defined, (2) a given disturbance is known to be new to the system at hand (e.g. some kind of human disturbance), *or* (3) the disturbance is under direct experimenter control (White & Pickett 1985).

From a system perspective several descriptors of disturbances can be considered that together constitute the *disturbance regime* (cf. White & Pickett 1985): (1) spatial distribution of the disturbance relative to environmental or community gradients, (2) frequency as the mean number of events per time period, (3) return interval, cycle or turnover time, (4) rotation period as the mean time to disturb an area equivalent to the study area, (5) predictability, (6) area disturbed, (7) magnitude of intensity and severity, and (8) synergism as a measure of the effects on the occurrence of other disturbances. In each case disturbance statements have to be specified for both the spatial and temporal scale. In my view the ecological checklist proposed by Grimm & Wissel (1997) for stability statements could function analogously as an appropriate tool for disturbance statements. Key processes common to all disturbances are alterations of resource availability and system structure (Pickett & White 1985).

Another distinction is given by Peterson (2002) between *contagious disturbances* such as fire, insect outbreaks and grazing herbivores, and *non-contagious disturbances*. Contagious disturbances appear to be more relevant for the structure of landscapes.

Most disturbances produce heterogeneous and patchy effects, a phenomena for which White & Pickett (1985) suggest the term *patch dynamics*. The authors propose that, in general, biological systems, on some level expose a heterogeneous community structure and behaviour (White & Pickett 1985). Levin (1992) suggests that disturbance is relevant for the maintenance of the “character” of ecosystems as a structuring agent. Additionally, natural disturbances occur in a wide variety of biomes (coniferous, deciduous, evergreen and tropical forests, grasslands, shrub-lands, tundra and deserts) and impacts are observable at all levels of ecological organization (Pickett & White 1985b).

As a matter of course, it is beyond the scope of this thesis to point out the whole discussion on disturbances. This notion of disturbance provides a provisional concept only. In the proceeding examinations further insights will expand the notion of disturbance putting it into relation to other concepts, such as ecosystem resilience or the *adaptive cycle* (cf. section 3.2.2.1). If ecosystem resilience – which represents the main topic of this thesis - is conceived from an operational perspective the disturbance regime provides the *to-what part* of the *ecosystem resilience analysis* (cf. section 4.2).

There is another important point with respect to the two distinct resilience definitions.

Although being only a different *aspect* of “stability” the emphasis on one of the two distinct stability properties – ecosystem resilience or engineering resilience – can be decisive. It can result in different views of nature (cf. section 3.2.1), different basic assumptions (e.g. the debate about alternate stable regimes, cf. section 3.2.3), different views on stability itself (cf. section 3.4.1) and as an outcome of the whole to a different environmental management which is characterised by entirely different methods (cf. section 4.3) (cf. Holling & Meffe 1996, Gunderson & Holling 2002, Gunderson & Pritchard Jr. 2002). The *Resilience Alliance* considers ecosystem resilience to be the more valuable concept both theoretically as well as operationally (cf. section 3.4.2). In the following, I will concentrate on ecosystem resilience.

The distinction between engineering resilience or resilience on the one hand and ecosystem resilience or robustness on the other, refers only to the original-ecological meaning. However, the term ecosystem resilience is used for several meanings in a broader sense by the *Resilience Alliance* as well as by other scientists. Due to its history and usage ecosystem resilience is a concept with multiple meanings (Gunderson 2000) that have to be separated meticulously (cf. section 3.4.3).

3.2 Background Theory of Ecosystem Resilience

The definition of ecosystem resilience presupposes several ecological concepts and notions that function as theoretical *background theory*. The aim of the following chapters is to discuss these related concepts which ought to lead to a more profound understanding of the ecosystem resilience concept in its entire meaning.

3.2.1 Different Views of Nature

Within ecological science different views of nature that underlie explanations of how nature works can be identified. Each of these caricatures, or myths, are not solely hypothetical and non-affecting theories. Rather they lead to different assumptions about stability properties, different perceptions of the processes that affect these properties, and even different policies that are deemed appropriate. All these views are hold by several authorities and affect environmental thinking and policies. It is, thus, crucial to be aware of one's view of nature.

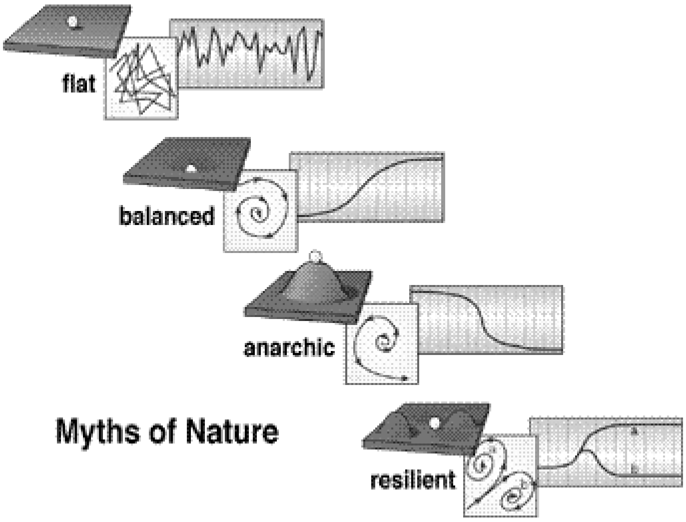

Figure 1: Myths of nature occurring in the relevant literature
(from [[online] URL: http://www.resalliance.org, resource library)

As illustrated in *Figure 1*, Holling & Gunderson (2002) distinguish between different *myths of nature* (cf. also Holling 1986, 1995).

The first view – *Nature Flat* – stresses the characteristics of ecological systems in which few or no forces affect stability. There are no feedbacks or consequences from nature of human actions and processes that affect the configuration of nature are random or stochastic. It 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. Such a "flat worlder" view is not wrong, just incomplete. There are indeed strong stochastic elements affecting ecological systems.

The second view – *Nature Balanced* – perceives ecological systems to exist at or near an equilibrium condition. If nature is disturbed, it will return to an equilibrium through negative feedback. It is the myth of *maximum sustainable yield* [45] and of achieving fixed carrying capacity for animals and humanity. This is the view of several organizations with a mandate for reforming global resource and environmental policy, e.g. the *Brundtland Commission*, the *World Resource Institute*, the *International Institute of Applied Systems Analysis*, and the *International Institute for Sustainable Development*. The view *Nature Balanced* is not wrong but also incomplete. There are indeed forces of balance in ecological systems.

The third view – *Nature Anarchic* – assumes that ecological systems are dominated by hyperbolic processes of growth and collapse, where increase is inevitably followed by decrease. It is a view of fundamental instability, humans are seen as incapable of learning. The "anarchist worlder" is again not wrong but incomplete. There are indeed destabilizing forces in ecological systems.

The fourth view is dubbed *Nature Resilient*. It is a view of alternative stable regimes[46] and fundamentally discontinuous events and nonlinear processes[47]. There are periods of exponential change, periods of growing stasis and brittleness, periods of readjustment or collapse, and periods of reorganization for renewal[48]. These are

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[45] Almost every textbook in ecology considers maximum sustainable yield which is defined as "the maximum yield or crop which may be harvested year after year without damage to the system" (Lincoln, Boxshall & Clark 1998, 180).

[46] Ecosystems can exist in alternative regimes that are each stable and characterized by a different structure and function. Structuring variables in ecosystems can cause the shift between the regimes (cf. section 3.2.3).

[47] Non-linear processes are described with respect to alternative stable regimes (cf. section 3.2.3).

[48] These periods constitute the adaptive cycle as model for ecosystem dynamics which is important likewise for resilience theory (cf. section 3.2.2.1).

the dynamics argued for ecosystems (Holling 1986) which, however, presumes a stationary *stability landscape* [49]. Ecological resilience, for instance, is perceived as a fixed quantity. *Nature Resilient* is not wrong, i.e. there are indeed cycles of change that can move variables among stability domains, but rather incomplete, since those very movements contribute to the apparent fixed nature of the contours of the stability landscape.

The fifth view – *Nature Evolving* – is evolutionary and adaptive. The stability landscape is actively shifting and exhibits self-organization, i.e. the stability landscape affects behaviour of the variables, and the variables, plus exogenous events, affect the stability landscape. Ecosystem resilience and other properties of the ecosystem represent dynamic characteristics that change during ecosystem dynamics and succession, respectively. It is a view of abrupt and transforming change which exposes the need to integrate an understanding of unpredictable dynamics in ecosystems.

According to the *Resilience Alliance*, a transition from the structurally static view of *Nature Resilient* to a structurally dynamic view of *Nature Evolving* is needed. The challenge is to be able to respond in a flexible way to uncertainty and surprises, i.e. a capacity to maintain options in order to buffer disturbance and to create novelty (Holling & Gunderson 2002). The whole theoretical work of the *Resilience Alliance* can be seen as attempt to develop an appropriate picture of the view *Nature Evolving.* Corresponding to this, the concepts spelled out in the following chapters and put forward by the *Resilience Alliance* are ingredients needed in order to understand *Nature Evolving*. Thus, ecosystem resilience as well as other stability properties have to be understood against the background of *Nature Evolving*.

3.2.2 A Model of Complex Adaptive Systems

Facing the anything but simple complexity of nature it is crucial to develop an understanding of the organizational and dynamical patterns of complex systems. In his *MacArthur Award Lecture* Levin states: “now, more than ever, we need to

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[49] A stability landscape illustrates in a pictorial way alternative stable regimes and instable regions that an ecosystem can exhibit. It represents a crucial part of ecosystem resilience background theory (cf. section 3.2.4)

develop mechanistic evolutionary theories of how ecosystem patterns arise and are maintained" (Levin 1992).

In this respect, Levin (1998) suggests three essential elements of *complex adaptive systems*: (1) sustained diversity and individuality of components, (2) localized interactions among those components, and (3) an autonomous process that selects from among those components, based on the results of local interactions, a subset for replication or enhancement.

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 rather from a small number of controlling processes or variables. These variables establish a persistent [*sensu* Grimm & Wissel (1997)] 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 approach of the *Resilience Alliance* to complexity and is outlined in several key publications (Holling 1986, 1992, 2000, 2001, Holling & Gunderson 2002).

It is beyond the scope of this thesis to outline the whole discussion on the different views on complexity. Rather, in the following, I will explore the position of the *Resilience Alliance* on complex systems as a basis from which different aspects of the ecosystem resilience concept can be explored more profoundly. The goal of the *Resilience Alliance'* view is to rationalize the interplay between change and persistence, between the predictable and the unpredictable processes occurring in complex systems. The *Alliance* 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. It might be possible to identify the points at which a system is capable of accepting positive change and the points where it is vulnerable (Holling 2001).

3.2.2.1 Adaptive Cycle

The framework for explaining complex systems rests on three requirements (Holling & Gunderson 2002). First, the system must be productive, i.e. must acquire resources and accumulate them, for the potential they offer for the future. Second, 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). Third, somehow 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 & Wissel (1997).

Thus, a *framework for adaptive change*, that has generality, contains three ecosystem properties that shape the future responses (Holling & Gunderson 2002): (1) the *potential for change*, since that determines the range of options possible, (2) the *degree of connectedness* between internal controlling variables and processes , a measure that reflects the degree of flexibility or rigidity of such controls, and (3) the *ecosystem resilience* of the system. 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 the ecosystem succession proceeds. They show a regular behaviour on a patch scale[50] passing through four distinct phases which together constitute the so-called *adaptive cycle* (Holling 1986, 2001, Gunderson & Holling 2002) proposed as a general model of systemic change which is illustrated in *Figure 2*.

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[50] The adaptive cycle that is being suggested in the following refers to the patch scale in the first place. Section 3.2.2.2 will explore the *Panarchy* that includes adaptive cycles at multiple scales.

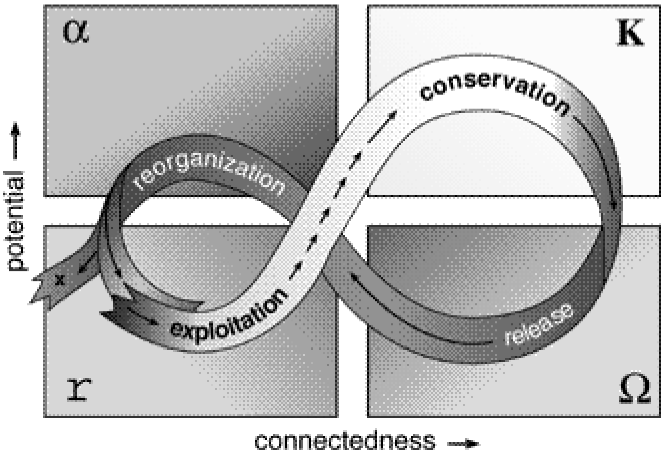

Figure 2: Adaptive cycle: the heuristic of ecosystem dynamics for two ecosystem characteristics: potential and connectedness
(from [online] URL: http://www.resalliance.org, resource library)

The traditional view of ecosystem succession has been seen 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 (Krebs, C.J. 2001). In ecology, the species in the exploitive phase have been characterized as r-strategists and in the conservation phase as K-strategists[51]. The *Resilience Alliance*, therefore, dubs the exploitation phase *r phase* and the conservation phase *K phase* (e.g. Holling 2001).

But, according to the *Resilience Alliance*, recent empirical examinations and subsequent ecological understanding indicates that two additional functions are needed (Holling & Gunderson 2002). 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 nutrient loss and reorganize nutrients in such a manner that they become available for the next phase of exploitation. Part of

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[51] 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, C.J. 2003).

this reorganization involves the transient appearance or expansion of organisms that begin to capture opportunity – the pioneer species.

If the omega phase represents the end, then it is immediately followed by the alpha phase, the beginning, to complete the (adaptive) 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 (e.g. Holling et al. 1995, Holling & Gunderson 2002).

As illustrated in *Figure 2*, the flow of a system through the adaptive cycle is plotted as changes in two properties - connectedness and potential - which expand and contract as proceeding the cycle. During the slow sequence from exploitation to conservation (K phase), connectedness and constancy [*sensu* Hansson & Helgesson (2003)] 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 regimes 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, i.e. the system’s connectedness increases, eventually to become over-connected and increasingly rigid in its control.

The actual change 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 addition, new entrants, those that survived to the α phase, and the *biotic legacies* [52] of past cycles (Franklin & MacMahon 2000) 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.

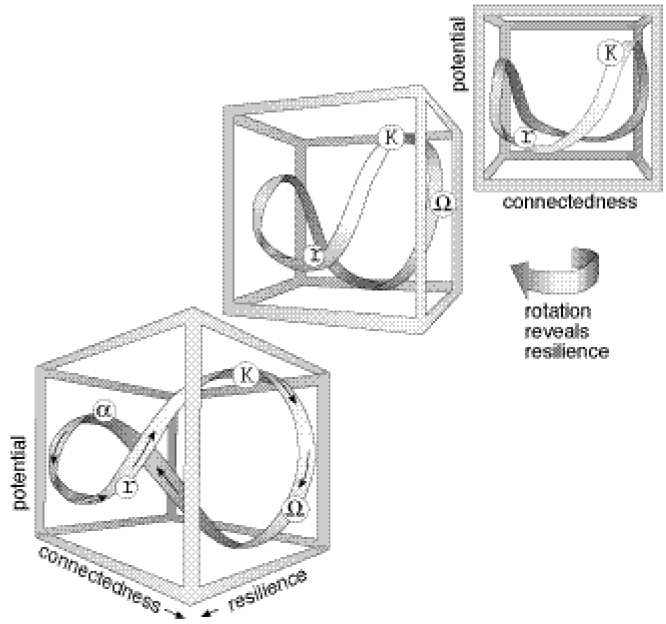

Figure 3: Three-dimensional adaptive cycle for three ecosystem characteristics: potential, connectedness and ecosystem resilience
(from [online] URL: http://www.resalliance.org , resource library)

Figure 3 adds the third dimension of an adaptive cycle, the ecosystem resilience. The myth of *Nature Resilient* (cf. section 3.2.1) sees ecosystem resilience as a fixed quantity for the whole system whereas the notion of *Nature Evolving* considers ecosystem resilience as a dynamic quantity expanding and contracting within a cycle as *slow variables* [53] change (Holling & Gunderson 2002).

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[52] *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 part of the *ecological memory* discussed in section ???.

[53] State variables of ecosystems can be divided in fast and slow variables. Slow variables appear to be responsible for the underlying structure of ecosystems (cf. section 3.2.4).

The latter view encourages to recognize that conditions are needed that occasionally foster novelty and experiment. Those become possible during periods when connectedness is low and ecosystem resilience is high. The low connectedness permits novel re-assortments of elements that previously were tightly connected to one another. The high ecosystem resilience allows tests of those novel combinations because system-wide costs of failure are low (Holling & Gunderson 2002).

During the α phase ecosystem 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 represents, thus, one of the key elements in the notion of *Nature Evolving* - the condition where novel re-assortments of species in ecosystems generate new possibilities that are later tested (Holling & Gunderson 2002).

Ecosystem resilience remains high in the r phase since thriving biota is adapted to high variability of microclimate and extremes of soil conditions – the pioneer species.

During K phase the abundance of K-strategists and the connectedness among them increases resulting in conditions that are more predictable. Engineering resilience is high since small changes in regime configuration are removed quickly whereas ecosystem resilience, however, 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 ecosystem resilience is low. The system becomes an “accident waiting to happen” (e.g. 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. Ecosystem 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).

One of the important points for the ecosystem resilience concept is that from the adaptive cycle perspective it presents a dynamic quantity that changes - on the patch scale - when the cycle proceeds. There are times when the patch exhibits high ecosystem resilience (α and r phases), and there are times when the patch has low ecosystem resilience (K and Ω phases), i.e. when the system is either more or less vulnerable to internal and external fluctuations (Peterson 2000).

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, the outcomes of the second stage in the back loop can be *highly unpredictable* (Holling 2001). In this perspective, *uncertainty and* surprise[54] *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).

This metaphor suggests 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[55] (Holling & Gunderson 2002, 47). 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 ecosystem resilience (Walker et al. 2002).

To sum up, four key features characterize an adaptive cycle and can be distinguished (Holling 2001): (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 ecosystem resilience) to such threats as fire or insect

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[54] "Surprise denotes the condition when perceived reality departs *qualitatively* from expectation" (Berkes and Folke 1998, 6).

[55] This insight is highlighted by the distinction of the *pathology of resource management* (where management tries to focus on efficiency of production) and *adaptive management* explored in section 4.3.

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 functions as a metaphor (e.g. 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). The *Resilience Alliance* considers it rather to be a tool for thought, a heuristic, which should stimulate thoughts and hypothesis that can be tested empirically (Resilience Alliance 2002, Gunderson & Holling 2002).

3.2.2.2 Panarchy

The ontogeny of an ecosystem represents a particular form of evolution, which Lewontin termed *transformational evolution* (Levin 1998). In that transformational process, clusters form, flows become modified, and the system assumes shape through a process of self-organization. *Self-organized processes* in this respect, represent processes that produce patterns and are in turn reinforced by those patterns (Kauffman 1993). The development of patterns of aggregation and hierarchical organization both are seen to be a natural consequence of the self-organization of any complex system (Levin 1998). This self-organization produces hierarchical structures – with smaller and faster distinct from larger and slower levels - since these structures increase the ecosystem resilience and, thus, the maintenance of the whole system whereas non-hierarchical do not (Peterson 2000).

In 1947, Simon was one of the first to describe the adaptive significance of hierarchical structures. He termed them *hierarchies* but not in the sense of a top-down sequence of authoritative control. Rather, semi-autonomous levels are formed from the interactions among a set of variables that share similar speeds and spatial attributes (Holling 2001). As long as the transfer from one level to the other is maintained, the interactions within the levels themselves can be transformed, or the variables changed, without the whole system losing its integrity (Peterson 2000, Holling 2001).

Building on Simon, Allen and Starr (1982) stimulated a shift of attention from the small-scale view to a multi-scale and landscape view that recognized that biotic and abiotic processes could develop mutually re-enforcing relationships over distinct ranges of scale (Holling 2001). 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 1986, 2001), which is reflected by the r and K phases and the Ω and α phases, respectively, of the adaptive cycle explored above.

The adaptive cycle was proposed for the patch scale (Holling & Gunderson 2002). In general, there cannot be one single appropriate scale that covers the whole range of dynamics of complex systems (Levin 1998, Holling et al. 2002). Dynamics occur at every scale of the hierarchy. Therefore, each spatial and temporal level – from plant to patch, to stand, to ecosystem, to landscape – has its own adaptive cycle (Holling et al. 1995, 2002). As illustrated in *Figure 4*, there are nested sets of such four-phase cycles which interact with each other. These interacting hierarchies are dubbed *Panarchy* (Gunderson & Holling 2002, cf. also Gunderson, Holling & Light 1995b).

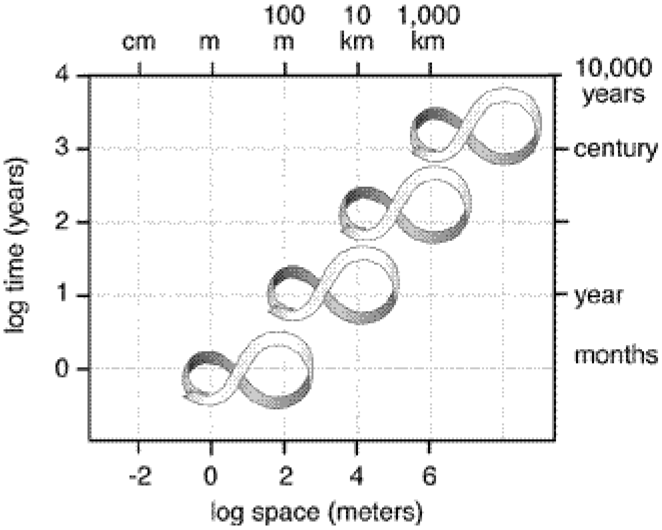

Figure 4: Panarchy: nested sets of adaptive cycles over various scales
(from online URL: http://www.resalliance.org, resource library)

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 function (Holling et al. 2002). 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 (Holling 2001). They are labelled *Revolt* and *Remember* (Holling 2001, Gunderson & Holling 2002, Gunderson & Pritchard Jr. 2002).

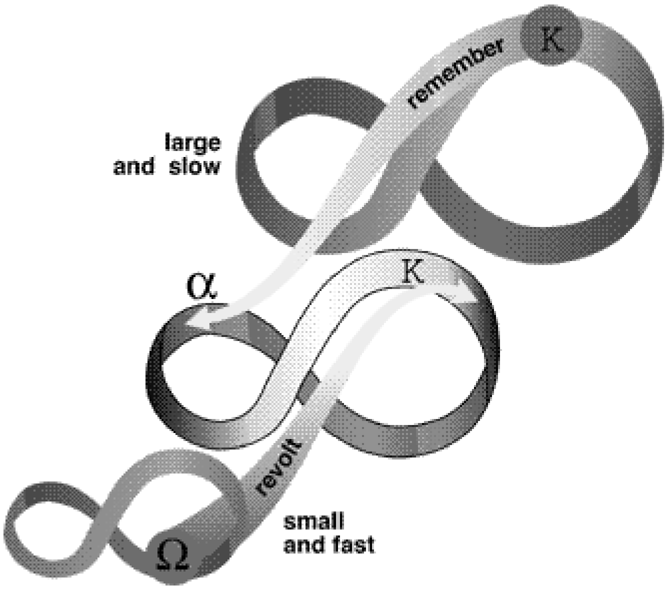

Figure 5: Interactions between scales: Revolt and Remember
(from [online] URL: http://www.resalliance.org, resource library)

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 ecosystem resilience is low. The *Revolt* arrow in *Figure 5* suggests this effect – where fast and small events overwhelm slow and large ones (Holling et al. 2002). 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.

The arrow labelled *Remember* in *Figure 5* 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. A opened patch in a forest due to a storm disturbance, for instance, is re-colonized by surviving species (*biotic legacies* [Franklin & MacMahon (2000)], 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* 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.

It is important to note that the *Resilience Alliance* considers both the adaptive cycle and the *Panarchy* as being useful to describe not only ecological systems but also social-ecological systems. According to the *Alliance* one of the main mistakes in examinations of ecosystems consists in the separation of the *human system* and the *natural system*. The interaction of the two realms create emergent properties without which system dynamics cannot be understood (Folke et al. 2002). A fundamental task of environmental management consist in the examination of the interactions of these two realms within large-scale social-ecological systems. In this section my main focus is on ecosystem dynamics, however.

The adaptive cycle and the *Panarchy* represents a metaphor for the dynamic of complex adaptive systems (Holling & Gunderson 2002). It is, however, meant not only to function as a metaphor but also to generate fruitful hypotheses for the analysis of complex systems. For an appropriate examination of ecosystem dynamics one has to specify *what exactly cycles* as the cycle proceeds. In other words, it is crucial to identify the self-identity of the ecosystem that is being considered [following Jax, Jones & Pickett (1998) and Jax (2002)], the *essential variables*, i.e. variables that control system behaviour, as well as the spatial and temporal scale [according to Grimm & Wissel (1997) and their ecological checklist] (cf. section 3.1.3).

In this respect, the *Resilience Alliance* conceives the hierarchical structure of ecosystems to be primarily regulated by a small set of plant, animal, and abiotic processes (Holling 1992, Holling et al. 1995, Holling 2001, Gunderson & Holling 2002, Gunderson & Pritchard Jr. 2002) that can be identified as the essential or key variables. Holling (1992) terms this the *extended keystone hypothesis*. Each of the key processes operates at characteristic periodicities, i.e. fastest, slower, and slowest temporal scales and specific spatial scales - either small, meso- or large spatial scales. These variables establish a persistent [*sensu* Grimm & Wissel (1997)] 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 (Hollling 2001).

Holling et al. (2002) state that three decades of studies or regional ecosystems from northern forest, southern wetlands, dry grasslands, lakes, and seas show that the key features of ecosystems are established by the interaction between these fast and slow essential variables. Biophysical processes concerning plant physiology, inter-specific plant competition processes and meso-scale disturbance processes of fire, storm, or herbivory, for instance, dominate fast and slow variables, whereas at the largest landscape scale, climate, geo-morphological, and bio-geographical processes alter ecological structure (Holling et al. 2002). The essential processes produce patterns and are in turn reinforced by those patterns, i.e. they are self-organized (Kauffman 1993). 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. Peterson (2002) suggests, that this extended keystone hypothesis is most likely to be demonstrated in ecosystems that exhibit strong *ecological memory* [56].

Thus, three to five fast/slow sets of variables, the nonlinear relationships between them, and stochastic processes generate the behaviour of complex systems (Holling et al. 2002). These key variables may be mainly driven by *keystone*

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[56] Ecological memory is defined as “the composition and distribution of organisms and their interactions in space and time, and includes the life-history experience with environmental fluctuations” (Nyström and Folke 2001, cf. section 3.3.3.4).

process species while the remaining species exist in the niches formed by these keystone process species (Folke et al. 1996).

What, however, is the evidence for this small set of variables?

Ecosystem dynamics is seen to provide a discontinuous template in space and time that entrains attributes of variables into a number of distinct *lumps*. The attributes of all size, speed, and function of ecosystem structures, such as species, populations, communities, and abiotic conditions are distributed in a lumpy manner (Holling 1992, Holling et al. 2002, Allen & Holling 2002).

The *Panarchy* which includes the essential variables each operating at specific temporal and spatial scales creates landscape structures with scale-specific pattern. This is termed *textural discontinuity hypothesis* (Allen & Holling 2002). These spatial pattern can be produced by simple interactions between contagious disturbance processes and vegetative dynamics, if the landscape holds a memory (Peterson 2002). 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 (Peterson 2002, cf. Allen & Holling 2002).

This lumpy ecosystem structure is echoed by a discontinuous distribution of species body masses showing body mass clumps and gaps which corresponds to the *world-is-lumpy-hypothesis* (Holling 1992)[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). Concerning this, Holling et al. (2002) suggest a strong correlation between complexity of lump structure and productivity or other correlates of net energy flux through terrestrial ecosystems. Boreal landscapes, 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 clumps. These lumps are conceived as constant [*sensu* Hansson & Helgeson (2003)], populations within them, however, are not (Forys & Allen 2002).

Additionally, Allen, Forys & Holling (1999) suggest that this discontinuous pattern has predictive power. First, invasions and extinctions of species in

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[57] There are sceptical views, however, doubting whether such lumps are real (e.g. Manly 1996).

landscapes subject to human transformation (Allen, Forys & Holling 1999) and, second, nomadic birds (Allen & Saunders 2002) tend to be located at the edge of body-mass lumps, which in turn suggests the existence of these body-mass lumps. The distribution of lumps and gaps is a kind of *bioassay* of the structure of the *Panarchy* and is seen as evidence for both the hierarchical structure of ecosystems and the existence of essential variables operating at distinct speeds and scales (Allen & Holling 2002, Holling et al. 2002).

3.2.3 Alternative Stable Regimes

It is easy to understand that gradual change in environmental variables may lead to gradual change in ecological systems, which is a case of *linear behaviour*. From this, it may appear to follow that abrupt changes in ecosystem structure or processes are attributable to abrupt changes in the environment. The obvious intuitive explanation for a sudden dramatic change in nature is the occurrence of a sudden large external impact. However, after 25 years of research, ecologists have recognized that gradual changes in environmental variables can sometimes cause abrupt changes in ecosystems which is a case of *non-linear dynamics*. Even a tiny incremental change in conditions can trigger a large shift in some systems (Carpenter 2001, Scheffer & Carpenter 2003, cf. Gorke 1996). Sharp changes in dynamics from gradual changes in parameters are an important reason for ecologists' interest in alternate states (Carpenter 2001).

Since Holling's paper *Resilience and stability of ecological systems* (Holling 1973) *alternative stable states* are seen as one of the phenomena that can lead to massive changes in ecosystems from only minor changes in the environment. It has been 30 years when Sutherland concluded that “multiple stable points are an undeniable reality” (Sutherland 1974, quoted from Wissel 1984, 101). Hence, proposition of the existence of multiple stable states is not something new on the horizon in ecology.

Building on that, the concept of ecosystem resilience presupposes the existence of *alternative stable regimes* and scientific inquiry stresses slow dynamics in a region that separates alternative basins of attraction. The concept of engineering resilience, on the other hand, presupposes global stability, i.e. an ecosystem has only one equilibrium or steady state (Gunderson 2001) and the focus of study is on slow dynamics near a stable equilibrium (Ludwig et al. 1997, 2002). The opposing views constitute what is called the *alternative-stable-state-controversy* which presents the heart of the two different views of resilience (Holling et al. 1995) and is regarded as the *key distinction* between them (Gunderson 2000).

In the following, I will use Scheffer & Carpenter's terminology (2003). Rather than *state* and *equilibrium*, the terms *regime* and *attractor* seem more appropriate to describe reality, since fluctuations within populations and communities are obviously the rule and ecosystems appear to be never constant [*sensu* Hansson & Helgesson (2003)] (Scheffer & Carpenter 2003)[58].

The idea of *alternative stable regimes* was introduced to ecology through theoretical papers including models. The first experimental examples that were proposed were criticized strongly. Nonetheless, recent studies have provided a strong case for the existence of alternative stable regimes in various important ecosystems such as in lakes (Carpenter 2001, Scheffer et al. 2001), marine systems (Done 1991) and oceans (Hare & Mantua 2000), in deserts (Foley et al. 2003) as well as rangelands (Perrings & Walker 1997, Janssen et al. 2004), in woodlands (Dublin, Sinclair & McGlade 1990) and forests (Holling 1986, Peterson 2002, Peterson et al. 2002)[59]. Such regime shifts 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). Moreover, Levin (1999) points out that several scientists emphasized the non-equilibrium nature of local population dynamics.

Seen from a model perspective alternative stable regimes can be described by a *ecosystem respond curve* that is folded backwards which is illustrated in *Figure 6* (Scheffer et al. 2001, Scheffer & Carpenter 2003). This curve implies that, for certain environmental conditions, the ecosystem has two alternative stable regimes (the solid lines), separated by an unstable attractor (the dashed line) that marks the border between the alternative *basins of attraction* [60].

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[58] There are many terms in use within relevant literature. Following Scheffer & Carpenter (2003), I will employ the term *alternative stable regimes*. Some authors, however, favour the terms *multiple stable states* or *alternative stable states* for the same phenomena. As *state* has static connotations I will rather use *regime* (Scheffer & Carpenter 2003) or *configuration* (Walker et al. 2002) to express the dynamic character of ecosystems. Since most ecosystems have only two stable regimes I prefer alternative stable regimes as opposed to *multiple stable regimes*.

[59] For a good overview of the evidence for multiple stable regimes confer Carpenter (2001), Scheffer et al. (2001)and Folke et al. (2002).

[60] I use the terminology from Scheffer et al. (2001) and Walker et al. (2004) in contrast to *domain of attraction sensu* Grimm & Wissel (1997).

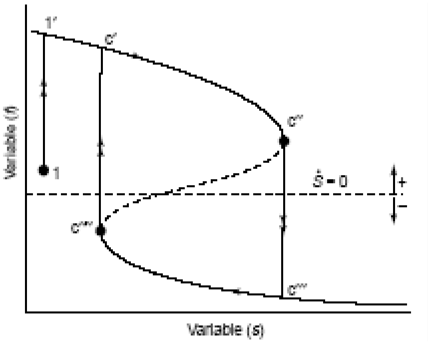

Figure 6: Bifurcation diagram with slow and fast variable. The ecosystem can exist within two alternative stable basins of attraction
(from Scheffer & Carpenter 2003)

When the system is in a state on the upper branch (between c' and c'') of the folded curve in *Figure 6*, it can not pass to the lower branch (between c'''' and c''') smoothly. Instead, when conditions change sufficiently to pass the critical value[61] (c'') 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 (c''''), where the system recovers by shifting back to the upper branch – a pattern known as *hysteresis* (Scheffer et al. 2001).

For example, shallow lakes can exist in a clear-water regime with aquatic plants and a turbid regime without vegetation (Scheffer et al. 2002). If the lake is in a clear 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 (Scheffer et al. 2002).

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[61] Surprising jumps in the state of ecosystems often are due to interactions between slow and fast variables or change of slow variables, respectively (Rinaldi and Scheffer 2000, cf. section 3.2.4).

Within ecological theory on alternative stable regimes, the term *basin of attraction* represents a region in *state space*, i.e. the variables considered and their relation, in which the system tends to remain (Walker et al. 2004). It is not static, rather it expands and contracts, and disappears in response to changes in slow variables (Holling 1986, Walker et al. 2004).

Both the Sahara and the Sahel regions (the transition zone between the desert and forest) in Northern Africa provide good examples for regions with alternative stable regimes and regime shifts. As Foley et al. (2003) point out strong nonlinear feedbacks between vegetation and the atmosphere can dramatically amplify the effects of orbital variations and create two alternative stable regimes in the climate and ecosystems of the Sahara – a *green Sahara* (regime 1) or a *desert Sahara* (regime 2). A regime shift occurred from regime 1 to the regime 2 approximately 5500 years ago.

The three-decade-long drought in the Sahel region from 1969 on may be the result of complex interactions among the atmosphere, land, and ocean (Foley et al. 2003). Dry conditions were initiated either through changes in sea-surface temperatures or increases in degraded land cover. In each case, the strong persistence of the drought results from the strong coupling between vegetation and monsoon circulations over the Sahel regions. These conditions help to reinforce either a *wet Sahel* or a *dry Sahel*. A climate transition to the dry Sahel took place abruptly once the coupled system passed a threshold and reached the domain of attraction of the alternative regime. Thus, in both the Sahara and the Sahel regions, it appears that environmental systems are predisposed toward having at least two alternative stable regimes (Foley et al. 2003).

Disturbance is an important and widespread phenomena in nature (Pickett & White 1985). Bengtsson et al. (2003) state that most ecosystems are subject to pulse disturbances at various spatial and temporal scales. Additionally, earth's ecosystems experience an increasing impact from human growth.

Suppose a large-scale human perturbation that reaches a certain magnitude. Clearly, the response of ecosystems to disturbance regimes and perturbations depends on the severity of the disturbance (Frelich & Reich 1998). Natural systems have a capacity to absorb change without dramatically altering their regime. But this "resilient character" (Holling 1973) has its limits. Then, a critical level of a controlling variable of the system is passed and the nature and extent of internal feedbacks alters (Walker & Meyers 2004) - a *regime shift* occurs, i.e. the ecosystem reorganizes around another set of controlling key variables and processes (Holling & Gunderson 2002) which includes its ecosystem resilience mechanisms (Carpenter & Cottingham 1997)[62]. This shift also alters an ecosystem's scaling, as for the other regime new ecological processes operate over different scales (Peterson 2000). The breakpoint 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).

In the paper *Thresholds in Ecological and Social-Ecological Systems: a Developing Database*, Walker & Meyers (2004) give some initial insights about ecological 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 on the landscape scale. Fourth, during a regime shift changes in system *feedbacks* [63] occur. Alternative regimes are, thus, characterized by a different set of system feedbacks. The most common kind of feedback change is a) a quantitative change in feedback from the biota to the environment (e.g. precipitation in cloud forests, albedo in West Africa, atmospheric water vapour or oxygen concentration in lakes). This primary effect is then often accompanied by a secondary effect in terms of species composition. The second kind of feedback changes are b) changes in direct feedbacks from the environment on species performance (e.g. below a threshold level of some environmental variable, one species or group of species is competitively superior to another). The third kind are c) feedbacks on intra-population processes (e.g. on one side of a threshold environmental level, population growth is positive, whereas beyond the threshold it is negative).

In summary, one can say that the existence of alternative stable regimes within ecosystems is rather common within nature. Hereby, in ecosystems with little human impact, the dynamics of the adaptive cycles in different patches will, in general, be partly unsynchronized in time and space, leading to a mosaic of patches

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[62] When there is a regime shift but no feedback change, there is no threshold effect, (e.g. massive loss of species) (Walker & Meyers 2004).

[63] Feedbacks can be defined broadly as "the result of any behaviour which may reinforce (positive feedback) or modify (negative feedback) subsequent behaviour" (Berkes and Folke 1998, 6).

and successional stages (White & Pickett 1985, Bengtsson et al. 2003). Moreover, alternative regimes, i.e. regimes of the same ecosystem that are located in different basins of attraction, can coexist side-by-side on a landscape scale (Scheffer & Carpenter 2003). 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 stable vegetation types (Peterson 2002)[64]. In this respect, White & Pickett (1985) suggest that biological systems, on some level, are patchy.

The contrast between gradual, stable dynamics (*equilibrium view*) and sharp shifts among very different states (*multiple stable regimes* [65]) has deep implications for our understanding of ecosystem change, methods for predicting ecosystem change, and policy choice (Carpenter 2001). In order to check if a system inhibits alternative regimes several tests and experiments can be launched[66]. But it still remains remarkably difficult to prove the existence of alternative basins of attraction in the field (Scheffer & Carpenter 2003). One can question where the burden of proof should be. Assuming that alternative attractors are absent when they are actually present could lead to dangerous false assumptions (Scheffer & Carpenter 2003) and following the *avoid-false-positives criterion* it would be reasonable to avoid this scenario. For instance, sudden regime shifts – or *threshold responses* (Scheffer et al. 2001) from a desirable regime of a system to an undesired regime can have far-reaching impacts on the social and economic welfare of the population living in this area but these impacts are in principal context-dependent (Walker & Meyers 2003). It could be of great value to be able to predict when a system is losing resilience and is in danger of collapse (Levin et al. 1998).

However, the existence of alternative stable states can be – and has been – predicted (Foley et al. 2003). This presents a vital option and a strong tool for environmental management. By examining the underlying state of environmental systems and their degree of nonlinearity, scientists may be able to *predict which regions of the world are susceptible to regime shifts and which are not* (Foley et al. 2003, Scheffer & Carpenter 2003).

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[64] This is termed *patch dynamics* [see White & Pickett (1985) for an overview]. The patch dynamics concept is closely related but not identical (Jax 2002) to the *mosaic cycle concept* put forward by Remmert (1992).

[65] 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 perspective (quoted in Holling & Gunderson 2002). Semiarid grazing systems are an example. It might not be easy to separate *quasi-alternate states* from *alternative stable regimes* clearly.

[66] Cf. Carpenter (2001) and Scheffer & Carpenter (2003) for an overview.

The exact timing and circumstances of regime shifts may be unpredictable. However, Wissel (1984) employs theoretical considerations that show that the characteristic time which an ecosystem needs to return to an equilibrium increases when a threshold is approached. This “universal law” may be used to predict the position of a threshold by extrapolation of empirical data which are recorded at a safe distance from this threshold. Similarly, Ludwig, Walker & Holling (1997, 2002) explore the relation of ecosystem resilience and return time and they conclude that long return times in their models represent a diagnostic tool for low ecosystem resilience or for disturbances that are large enough to take the system near an unstable equilibrium, i.e. an ecological threshold. Undoubtedly, prediction of the position of ecological thresholds would be of high value for an environmental management.

3.2.4 Stability Landscape

The ingredients *multiple stable regimes*, *basin of attraction* and *ecological thresholds* explored in the previous sections (3.2.2 and 3.2.3) have been summarized in the metaphor of a stability landscape. In sum, it provides a picture in order to understand the view *Nature Evolving* (cf. section 3.2.1) more appropriately.

Let us assume an ecosystem that exhibits alternative stable regimes within the boundaries of alternative basins of attraction. These alternative basins of attraction (illustrated as valleys) together with the ecological thresholds that have to be passed before shifting to another stable regime (illustrated by the boundaries) constitute a *stability landscape* which is visualized in *Figure 7* (Holling 1978, Scheffer & Carpenter 2003, Walker et al. 2004).

Ecosystem resilience is defined - in its *extended-ecological sense* – as “the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks” (Walker et al. 2004). Walker et al. (2004) suggest that this notion of ecosystem 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 ecosystem resilience (Holling 1973, Scheffer & Carpenter 2003).

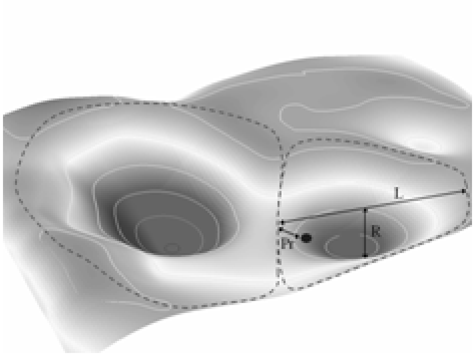


Figure 7: Stability landscape with two alternative basins of attraction and an ecological threshold; L: latitude, R: resistance and P: precariousness; the black dot illustrates the current regime of the ecosystem (from Walker et al. 2004)

A more detailed analysis is given by Walker et al. (2004). The authors distinguish four crucial *aspects of ecosystem resilience* three of which correspond to different characteristics of a basin of attraction (cf. L, R and Pr in *Figure 7*).

(1) *Latitude* (L) is defined as the maximum amount the system can be changed before losing its ability to recover which corresponds to the width of the domain of attraction. In my view, latitude matches more or less the meaning of basin of attraction we identified in section 3.2.3 and is, thus, closely related to resilience *sensu* Hansson & Helgesson (2003) and to engineering resilience *sensu* Gunderson & Holling (2002).

(2) *Resistance* (R) is defined as the ease or difficulty of changing the system which is related to the topology of the domain. This, in my opinion, is related to the *original-ecological definition* of ecosystem resilience which matches the “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior” (Gunderson & Holling 2002, 4) and to robustness *sensu* Hansson & Helgesson (2003).

(3) *Precariousness* (Pr) is 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).

The fourth aspect of ecosystem 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 ecosystem 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 ecosystem resilience.

This notion of ecosystem resilience represents a holistic and qualitative concept which corresponds to Grimm & Wissel's (1997) persistence. It is holistic since it refers to the entire ecosystem and qualitative because the focus is no longer on dynamics with its quantitative details, but on the qualitative question of whether the whole set of variables of interest used to characterize the system remains within certain boundaries (cf. Grimm et al. 1999 and Walker et al. 2004).

The components of the stability landscape represent an useful metaphor for ecosystem dynamics in order to understand the basic concepts related to ecosystem resilience. However, in my view, scientific understanding as well as operational purposes (i.e. estimation) and ethical objectives (e.g. maintenance of natural capital) require to specify *what exactly* the stability landscape refers to. This is one occasion where to appropriately use our critical tools of section 3.1.3.

There is the need, first, to define and specify the self-identity of the given ecosystem *sensu* Jax, Jones & Pickett (1998). Second, there is the need to identify the level of description, the variable of interest, the reference dynamic, the disturbance regime and the spatial as well as the temporal scale, i.e. to specify the *ecological situation sensu* Grimm & Wissel (1997) that the stability landscape applies to. Otherwise the concept of a stability landscape has heuristic value only.

Similarly, the distinct aspects of ecosystem resilience, i.e. latitude, resistance, precariousness and panarchy, have meaning only if they refer to a specified self-identity of an ecosystem and to a specific ecological situation.

Carpenter et al. (2001) suggest an operational, measurable concept of ecosystem resilience by clearly defining ecosystem resilience of what to what. Examinations of ecosystem resilience have to specify the time scale as well as the spatial scale, the reference dynamic (which stable regime is being considered) and the disturbances of interest (Carpenter et al. 2001).

Building on that Walker et al. (2002) propose a procedure for the analysis of ecosystem resilience. They distinguish a *of-what part* from the *to-what part* of an *ecosystem resilience analysis*. The to-what part of the analysis refers to the disturbance regime considered. The of-what part on the other hand includes the specification of the spatial and the temporal scale as well as the reference state (Walker et al. 2002, cf. section 4.2).

The crucial point I want to make, however, is to point to the focus of the of-what part of this analysis. Walker et al. (2002) state that it is crucial to identify, first, the key ecosystem services that are of concern to the people within the social-ecological system. Second - and this is of concern within this section – an ecosystem resilience analysis has to unfold the important controlling variables that act as drivers of the key ecosystem goods and services. These variables tend to have slower dynamics than the ecosystem goods, thus, they are termed *slow variables* as opposed to *fast variables* (Carpenter & Turner 2001, Walker et al. 2002). These slow variables are seen as crucial as they determine in which basin of attraction the ecosystem stays and performs its function including the ecosystem services people want. The stability landscape, thus, is seen to reflect these slow variables, within which the faster variables move around, reacting to the changes in the “topography” (Jansson & Jansson 2002, Gunderson & Walters 2002).

This corresponds to the view on complex systems that appear to be regulated by the interactions among three to five sets of variables, each operating at a qualitatively distinct speed and scale put forward by the *Resilience Alliance*. Ecosystem dynamics are conceived to be episodic caused by the interactions between fast and slow variables (Holling & Gunderson 2002, cf. section 3.2.2.2).

Seen from this system perspective, ecosystem resilience is affected by the interaction of fast and slow variables (Rinaldi & Scheffer 2000, Gunderson 1999, Gunderson & Walters 2002). Holling et al. (2002) suggest representative key or essential variables (fastest, slower and slowest) to illustrate the concept of fast and slow variables for various ecosystem types (cf. *Table 5*).

Note that the slowest variables can refer to a certain community (e.g. fish community) or even to a species or population, respectively (e.g. tree population) but also to a nutrient storage (e.g. phosphate in mud in lakes), i.e. to *abiotic factors*.

Janssen & Janssen (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 1999).

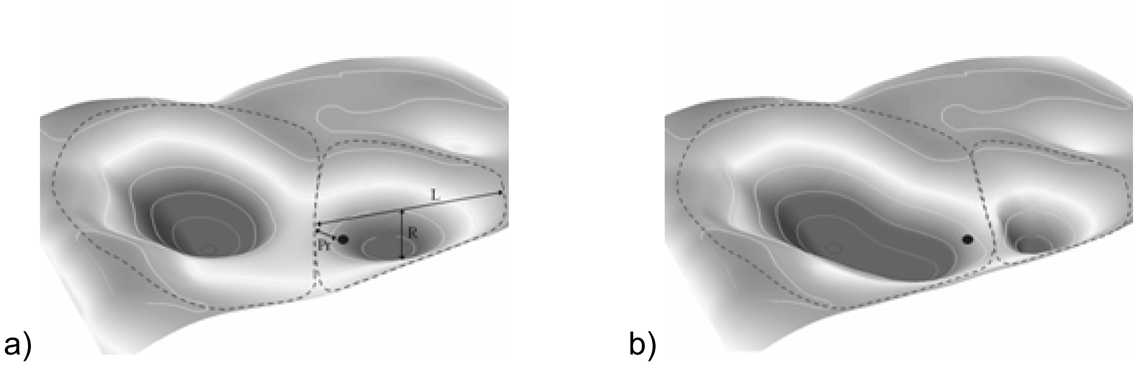
Table 5: Variables on disjunct time scales (fastest, slower, slowest) for different types of systems
(according to Holling et al. 2002)

	Variables			
System	**Fastest**	**Slower**	**Slowest**	**References**
Forest-pest dynamics	Insect	Foliage	Tree	Clark et al. 1997; Ludwig et al. 1978
Forest-fire dynamics	Intensity	Fuel	Trees	Holling 1986
Savannah	Annual grasses	Perennial grasses	Shrubs and grazers	Walker 1981
Shallow lakes and seas	Phytoplankton and turbidity	Sea grasses	Grazers	Scheffer et al. 1993
Deep lakes	Phytoplankton	Zooplankton	Fish and habitat; phosphate in mud	Carpenter et al. 1999
Wetlands	Periphyton	Saw grasses	Tree island; peat accreation	Gunderson 1994, 1999

The three-dimensional space of all possible combinations of the amount of the fast and slow variables constitute the *state space*. “We can conceive of the slow variables as defining the underlying structure of the system, while the fast variables reveal the dynamics of this underlying structure” (Carpenter et al. 2001, 778). The ecosystem resilience concept, thus, always refers to underlying slow dynamics of an ecosystem, hence, more or less the *capacity (i.e. the underlying mechanisms) of an ecosystem to maintain services in the face of a fluctuating environment* (cf. Carpenter et al. 2001, Folke et al. 2002, Deutsch, Folke & Skanberg 2003).

Slow variables do not function isolated within an ecosystem, rather they are 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 & Scheffer (2000), for instance, use models dubbed *singular perturbation approach* or *slow-fast-analysis* [67]. They 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, characteristic for the functioning of ecosystems.

These interactions between fast and slow variables are conceived to be one of the most important topics in ecology for a wise environmental policy (Carpenter & Turner 2000) and the most important scientific information for sustainable management, respectively (Carpenter, Brock & Hanson 1999). As Holling, Berkes & Folke 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, Berkes & Folke 1998, 354). This constitutes a crucial aspect of the *of-what part* of *ecosystem resilience analysis* (Walker et al. 2002, cf. chapter 4.2.3).



a) b)

Figure 8: 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)

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[67] 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 of-what part of an ecosystem resilience analysis *sensu* Walker et al. (2002) (cf. section 4.2).

Due to the interactions of slow and fast variables and the alteration of slow variables a stability landscape is not conceived as static. Rather, it behaves dynamically and alters in the face of exogenous drivers (rainfall, exchange rates) and/or endogenous processes (plant succession, predator-prey cycles, management practices) (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, while being mostly related to a *change in slow variables* (Holling 1986, Gunderson 2003, Walker et al. 2004). *Figure 8* illustrates a change in the position of an ecological threshold or the size of a basin of attraction, respectively. Dynamic stability landscapes are, in my view, the best heuristic in order to outline the meaning of *Nature Evolving* (cf. section 3.2.1).

As an example of a change in slow variables, *command and control resource management* [68] often leads to gradual loss of resilience and a change in slow variables. This results in regions that are more vulnerable to disturbance regimes and more prone to sudden switches to alternative regimes, a phenomena which is called the *pathology of resource management* (Holling & Meffe 1996). 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 functions such as productivity (Levin et al. 1998, Folke et al. 2002). Vulnerable and disturbed ecosystems tend to change not gradually, but non-linear, whereby outcomes differ from linear predictive models not only quantitatively but qualitatively. This is related to the existence of alternative stable regimes which are often fundamentally distinct in structure and function, respectively.

Considering the relevant literature, I conceive the examination of slow variables as a hopeful approach to grasp essential ecosystem behaviour and analyze the different aspects of ecosystem resilience which comprise resistance, latitude, precariousness, and panarchy.

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[68] Command and control approaches try to reduce the variability of a target variable (e.g. abundance of fish population) for an efficient human exploitation. Problems of this approach are discussed in section 4.3.1).

3.3 Resilience Mechanisms & the Ecosystem Functioning Debate

The background theory of ecosystem resilience has been explored in the previous sections (cf. section 3.2). Ecosystem resilience appears to be a diverse concept comprising several facets. The subsequent sections will show how the concept of ecosystem resilience is embedded within the ecological debates on biodiversity-ecosystem functioning and biodiversity-stability, respectively (section 3.3.1 and 3.3.2). Additionally, it will explore the mechanisms that are seen as responsible for the emergence of the stability property ecosystem resilience on the ecosystem level (section 3.3.3). Finally, concluding remarks on ecosystem resilience mechanisms attempt to paint a comprehensive picture (section 3.3.4).

3.3.1 Biodiversity-Ecosystem Functioning Debate

The question whether *either* abiotic conditions define biodiversity *or*, in contrary, biodiversity governs ecosystem function and influences abiotic conditions passes through at least the previous 60 years of ecological science and has challenged many of the highly acknowledged scholars. Guiding questions are which aspects and what level of biodiversity ensures processes at the ecosystem scale, i.e. ecosystem functioning.

Table 6: Paradigms in ecology with respect to ecosystem functioning: community ecology and ecosystem ecology
(according to Naeem 2002)

Paradigm	**Content**
Community ecology *(paradigm 1)*	Abiotic factors set regional patterns in distribution and abundance while biotic factors secondarily modify regional patterns; biodiversity is a passive consequence of intrinsic structure (interactions) and extrinsic factors (e.g. climate, geology)
Ecosystem ecology *(paradigm 2)*	Environment is primarily a function of diversity ; biodiversity plays an essential role in ecosystem processes

Discussing recent history of ecological science, Naeem (2002) distinguishes between two *paradigms*[69] (cf. *Table 6*) that emerged in the face of these questions, which, at the same time, show the separation of two ecological sub-disciplines, namely community ecology and ecosystem ecology[70]. The central thesis that currently guides community ecology (*paradigm 1*) in ecologists' attempt to explain biodiversity sees patterns in the distribution and abundance of species as a function of abiotic (physical and chemical conditions) and biotic factors (interactions among species). Abiotic factors set regional patterns in distribution and abundance while biotic factors secondarily modify regional patterns (e.g. Krebs, C.J. 2003). Thus, community ecology does not admit to a strong role for ecosystem processes in understanding nature. According to Naeem (2002) the main limitation of community ecology is that it considers biodiversity as a passive consequence of intrinsic structure (interactions) and extrinsic factors (e.g. climate, geology) – basically as a slave of the environment. The recently emerging paradigm (*paradigm 2*) conceives the environment primarily as a function of diversity promoting an active role for the biota in governing environmental conditions.

Two factors encourage adopting this alternative view of nature.

First, ecologists now focus less on diversity conceptualized through taxonomy (species richness only) in favour of a more inclusive concept of biodiversity. In general, the term *biodiversity* does not have a universally agreed on definition and it is often re-defined on each occasion according to the context and purpose of the author (Swingland 2001). Following the *Encyclopedia of Biodiversity* the more inclusive concept of biodiversity comprises *genetic diversity* (variation of gene sequences within and between populations, amount of DNA per cell, chromosome structure and number), *species diversity* (see below) and *ecosystem diversity* (relative abundances of species, trophic levels, taxonomic groups and alike) (Swingland 2001).

Similarly, in the *Dictionary of Ecology, Evolution and Systematics* biodiversity is defined as "the variety of organisms considered at all levels, from genetic variants of a single species through arrays of species to arrays of genera,

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[69] Naeem (2002) uses the term *paradigm* for "an unprecedented scientific achievement that is compelling enough to convince adherents from traditional perspectives to shift their allegiance, regroup around the new paradigm, and tackle problems anew" in accordance with Kuhn's initial definition in his influential book *The structure of scientific revolutions* (Kuhn 1962).

[70] Naeem (2002) conceives the separation of community ecology and ecosystem ecology as the largest and most recent fragmentation in ecology.

families and still higher taxonomic levels; includes the variety of ecosystems, which comprise both the communities of organisms within particular habitats and the physical conditions under which they live; the totality of biological diversity" (Lincoln, Boxshall & Clark 1998, 39)[71].

Walker (1992) lists numerous aspects of *ecological complexity* that are closely related to the term biodiversity. Decline in biodiversity includes all changes that have to do with reducing or simplifying biological heterogeneity, from individuals to regions, such as *phenotypic plasticity*, *genetic variability* within a population, *ecotypic variation*, *species richness*, *species (alpha) diversity*, *functional diversity*, *gradient (beta) diversity*, *community diversity*, and even the diversity of scales of patchiness (*landscape diversity*)[72]. In the following I will use biodiversity in this comprehensive sense.

The second reason for adopting *paradigm 2* consists in the recognition that physical as well as chemical conditions of the environment are driven, at least in part, by ecosystem function (e.g. nutrient cycling and energy flow), i.e. rates of ecosystem or biochemical processes affect physical and chemical state conditions (Naeem 2002). *Paradigm 2* is termed the *Biodiversity-Ecosystem Function Paradigm* (BEFP), which launched an explosion of research on the relationship between ecosystem function and biodiversity[73]. The main issues of the BEFP are that biota plays an essential role in ecosystem processes, and in particular, that diversity plays a significant role in such processes.

The view of diversity leading to stabilizing mechanisms (according to *paradigm 2*) is fundamentally distinct from a view that regards stability properties as a product of abiotic conditions (according to *paradigm 1*). However, according to Naeem (2002) both paradigms are not correct in an absolute sense. They rather reflect an dialectical process between theses (*paradigm 1*) and antitheses (*paradigm 2*) that leads to syntheses and creates further cycles of ecological dialectics. This dialectic nourishes ecological progress – constituting steps in the evolution of understanding nature – rather than reflecting accurate representations of nature.

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[71] This is also the position taken by the Convention on Biological Diversity of the United Nations Conference on Environment and Development in Rio de Janeiro 1992, signed by more than 150 nations (UNEP 1992).

[72] *Ecotypic variation* comprises genetic variability between populations within a species, *species richness* corresponds to the number of species in a community, *alpha diversity* involves both the number of species and the relative number of individuals per species, *functional diversity* matches the relative abundances of functionally different kinds of organisms, *beta diversity* extends to diversity resulting from speciation of ecological equivalents, *community diversity* corresponds to the number, sizes, and spatial distribution of communities (Walker 1992, 19).

[73] Cf. for an overview Naeem (2002) and Loreau et al.(2001, 2002)

3.3.2 Biodiversity-Stability Debate

What is the relevance of biodiversity-function research within a discussion about ecosystem resilience however? In my view, the biodiversity-ecosystem function research sets the framework for the relation of biodiversity in its comprehensive sense and various stability properties such as ecosystem resilience. This relation represents a fundamental issue throughout history of ecological science. Mooney et al. (1996), and following them Mooney (2002), identify two guiding questions for biodiversity-ecosystem functioning research: “Does biodiversity ‘count’ in system processes (e.g. nutrient retention, decomposition, production, etc.), including atmospheric feedbacks, over short- and log-term time spans, and in face of global change (…)?” and “How is system stability and resistance affected by species diversity, and how will global change affect these relationships?” (Mooney 2002, 13).

The fundamental question here is which level of biodiversity will secure the ability of the ecosystems on which human consumption and production depends (‘functions for’ humans) to continue to function under a range of as yet unknown conditions (‘functions of’ natural capital and ecosystem resilience) which is dubbed the *problem of biodiversity conservation* (Folke et al. 1994, Perrings et al. 1995). The debate whether biodiversity either is conducive or more of a hindrance for stability properties of ecological systems is dubbed the *diversity-stability debate*.

Four views can be identified constituting the debate (Loreau et al. 2002).

The early view until the 1960s held that diversity (or complexity) begets “stability” (*View 1*). In 1958, Elton observed that simple communities are more easily upset to destructive population oscillations and invasions. In 1955, MacArthur proposed that the more pathways there are for energy to reach a consumer, the less severe is the failure of any one pathway. 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 early 1970s by theorists such as Levins and May, who borrowed the formalism of deterministic autonomous dynamical systems and showed 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 the system, leading to either its collapse or its explosion.

These models were limited, however, as more realistic food webs incorporating thermodynamic constraints and observed patterns of interaction strengths do not necessarily have the same properties. Nevertheless, in the 1970s and 1980s the new paradigm emerged that diversity and complexity beget instability, not stability. There are other crucial limitations to this view. First, stability is seen to be a meta-concept that covers a range of stability properties. The relationship between biodiversity and each of them need not be the same. Second, each of these stability properties can be applied to a number of variables of interest at different hierarchical levels, such as individual species abundance, community species composition, or ecosystem level processes or properties. Again the relationship between biodiversity and any stability property may be different for different variables (Loreau et al. 2002). Moreover, different disturbances regimes affect different stability properties in a different way (Grimm & Wissel 1997, Frelich & Reich 1998, Paine et al. 1998). As explored in section 3.1.3 this results in a large matrix of potential combinations of stability properties and variables of interest and temporal and spatial scales and disturbances. Additionally, Bengtsson et al. (2002) state that results from one ecosystem do not necessarily inform us about other ecosystems.

According to Loreau et al. (2002), most of the theory on the complexity and stability of ecological systems has focused on deterministic equilibria, hence, on the concept of engineering resilience, and has ignored much of the potential for *functional compensation* [74], both within and between species, which represents the basis for the “stabilization” of ecosystem properties. During the 1970s and 1980s scientists emphasised functional compensation between species as the mechanism that stabilizes ecosystem processes against a background of wider variability of individual populations (*View 3*).

These ideas are the basis of the new wave of theoretical, experimental, and observational work that developed in the late 1990s. The focus moved from populations, communities and food webs to ecosystems and the interplay between community-level dynamical processes and ecosystem-level functional processes. New approaches address the link between the variability of individual species and that of aggregate ecosystem properties, and explicitly incorporate population

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[74] *Functional compensation* corresponds to structural change within communities that stabilizes ecosystem process rates (Frost et al. 1995, Carpenter & Cottingham 2002)

dynamical responses to environmental fluctuations. New work is focusing on ecosystem-level properties as the variable of interest and on systems subject to environmental fluctuations, in which the species' responses to these external fluctuations interact with the internal forces of species interactions (*View 4*). *View 4* addresses the distinction between taxonomic diversity and *functional diversity* defined as the relative abundances of functionally different kinds of organisms (Walker 1992), and its influence on stability properties in general, and ecosystem resilience, in particular (e.g. Walker 1992, 1995, Gitay et al. 1996). The important point here is that the focus of interest shifts to functional diversity which is seen as a fundamental concept for understanding the emergence of ecosystem resilience (cf. section 3.3.3.1).

The findings of current biodiversity-ecosystem functioning debate are presented in the synthesis paper *Biodiversity and Ecosystem functioning: current knowledge and future challenges* (Loreau et al. 2001).

The authors state that they can not reject the hypothesis that a few dominant species suffice to provide the functional diversity that is necessary to explain the level of primary production observed in grassland ecosystems at the small spatial and temporal scales. There is likely to be some, and perhaps considerable redundancy, which means that not all species that occur in a given habitat are actually critical to the functioning of that habitat (Walker 1992, 1995, Ekins et al. 2003). The challenge is to examine what kinds of biodiversity and which species are being most significant to the ways ecosystem function.

Moreover, as diversity increases, the variability of individual populations may increase as a result of the destabilizing influence of strong species interactions internal to the system, but the variability of aggregate ecosystem properties often decreases because of the stabilizing influence of asynchronous species responses to intrinsic or extrinsic environmental fluctuations (Tilman 1996, Yachi & Loreau 1999). What remains unclear is whether that stabilizing effect saturates at low or high diversity. Thus, at least some minimum number of species is essential for ecosystem functioning under constant conditions and a larger number of species is probably essential for maintaining the "stability" of ecosystem processes in changing environments – dubbed *insurance hypothesis* (Yachi & Loreau 1999, Loreau et al. 2001).

Specific knowledge of *functional types* within ecosystems may be critical to predict ecosystem response under different global change scenarios (Loreau et al. 2001). Relative effects of individual species and species richness may be expected to be greatest at small-to-intermediate spatial scales, but these biological factors should be less important as predictors of ecosystem processes at regional scales, where environmental heterogeneity is greater. At larger scales species diversity itself is a dynamical variable and adjusts to changes in environmental conditions. Abiotic factors then tend to the main drivers of variations in ecosystem processes across environmental gradients.

The maintenance of a given level of diversity at local scales is seen to require much higher diversity at regional scales. One of the most decisive effects of declining biodiversity could be the decline in the rate at which appropriate potential dominants are recruited during ecosystem assembly (Walker et al. 1999, Loreau et al. 2001).

In general, however, significant differences in the relationship between diversity and ecosystem functioning might be expected between ecosystem types (Loreau et al. 2001).

There is an important point here.

Most classical and recent research on stability properties has been performed at small spatial and short temporal scales. These classical equilibrium approaches are seen to be inadequate to understand stability properties such as ecosystem resilience and shifts between alternative stable regimes at larger scales (Loreau et al. 2001). 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 (Bengtsson et al. 2002). When the scale of investigation changes properties of communities and ecosystems do not just change in any coherent fashion. In this respect, Bengtsson et al. (2002) question the operational relevance of small-scale studies for a Sustainable Development. They argue for investigations on different scales and in particular on large-scale levels, comparisons of lands under different management practices and ecosystem models in order to be able to answer whether diversity really is important for the ecosystem resilience and maintenance of ecosystem services at larger scales in space and time. In my view the *scale issue* is crucial likewise for the *operationalization* and *implementation* of ecosystem resilience which will be considered in detail in section 4.2.4.

To sum up, a new approach, that claims to be not only scientifically sound but also to be relevant with regard to recommendations for a Theory of Sustainable Development, has to take into consideration the new notion of the relationship between biodiversity and ecosystem functioning (*View 4*), larger scales of investigation and new insights in ecological science, such as the notion of complex adaptive systems, non-linear behaviour of ecosystems and alternative stable regimes.

As a matter of course it is beyond the scope of this thesis to explore the whole discussion on the relationship between biodiversity and ecosystem functioning[75]. The ideas that were unfolded in this section provide an humble and provisional outline. In the following I will explore the *Resilience Alliance'* view on mechanisms that underlie the pattern and property of ecosystem resilience (termed *ecosystem resilience mechanisms*), and, in particular, their view on the relationship between biodiversity and ecosystem resilience. What mechanisms are conceived as responsible for ecosystem resilience? What ecosystem and species representation is needed to ensure ecosystem resilience in the face of fluctuating conditions?

3.3.3 Ecosystem Resilience Mechanisms

The three fundamental stability properties – constancy, resilience and robustness [*sensu* Hansson & Helgesson (2003)] - and the facets of biodiversity (e.g. genetic, taxonomic, functional, within and across ecosystems, landscape diversity) create a large matrix of possible combinations. Note that, hereby, ecosystem resilience itself comprises several aspects (latitude, resistance, precariousness) which could be influenced by separate aspects of biodiversity and ecosystem dynamics in a different manner. I will use the different aspects of the ecosystem resilience definition *sensu* Walker et al. (2004) in order to unfold various connections.

As mentioned before, biological sciences at the level of organisms and above are plagued by multiple causation (Paine 2002). It can be difficult if not

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[75] Eventually, the continuing debate in the face of biodiversity loss may lead to fundamental issues in ecology. Naeem proposes following research questions: “What constitutes a diversity effect? What constitutes a mechanism? How is evidence marshalled to support invoked mechanisms? What constitutes a parsimonious explanation?” (Naeem 2003, 619), that point to the heart of the biodiversity-stability debate.

impossible to find the fundamental cause of an interaction within a complex system. Searching for mechanisms for the properties that are located on the ecosystem and landscape level, in general, ecologists will focus on lower levels of integration (Krebs, C.J. 2003). On this level, however, several theories might be necessary in order to explain a certain phenomena or process (Mayr 2000). In his fundamental *MacArthur Award Lecture* about pattern and scale Levin states that “in general, there will be many conceivable mechanisms that could give rise to any set of patterns. All that theory can do is to create a catalogue of possible mechanisms” (Levin 1992). This is exactly what scholars considering the relation of biodiversity and ecosystem resilience did. Similarly, Jeltsch, Weber & Grimm (2000) propose that for statements to apply in general for ecological situations, may mean the development of a general framework for asking the right questions to analyze ecological systems. The general question in this case is: which mechanisms are responsible for the emergent property ecosystem resilience? Therefore, in the following sections, six ecosystem resilience mechanisms are proposed which are seen as being responsible for the stability property. Eventually, these mechanisms might be conceived as distinct *components of ecosystem resilience*.

As we will see in chapter 4.1.2, the *Resilience Alliance* considers ecosystem resilience to be a concept that is useful not only for ecosystems but for social-ecological systems (SESs) as well. They actually state that it is crucial not to separate the two realms of SESs (*natural system* and *human system*) in order to take into consideration the properties that emerge from the interactions of these realms. For a detailed analysis of the ecosystem resilience concept, however, it seems appropriate to separate the two realms analytically knowing that this separation is valid analytically only. The understanding of large-scale SESs necessitates the relation of the two realms in a second step. In this respect, Walker et al. (2002) suggest the terms *biophysical components of ecosystem resilience* and *social components of ecosystem resilience* in order to highlight this analytical distinction.

The next section will be concerned with resilience mechanisms that occur in the natural systems of ecosystems only, thus, with the *biophysical components of ecosystem resilience*.

3.3.3.1 Ecological Redundancy

Drawing from both the concept of *guilds* (group of species that perform the same function) and the concept of *ecological equivalents* (group of species that share a *niche* [76] or that have similar competitive ability), Walker (1992) introduced the concept of *ecological redundancy* (or *functional redundancy*). There is ecological redundancy if more than one species exist within a precisely separated guild that performs the same ecological function[77].

Hereby, Walker's (1992, 1995) purpose is to identify kinds of biodiversity that are most significant to the ways ecosystems function, because this is how to best focus the conservation efforts. Walker (1995) puts forward an *ecosystem view of conservation* which suggests that the best way to approach the problem of conserving biodiversity is to ensure that the system continues to have the same overall structure and function, i.e. to preserve its ecosystem resilience *sensu* Gunderson & Holling (2002).

Within the broad spectrum of aspects of biodiversity functional diversity is considered to be of high importance with respect to ecosystem resilience. A functional approach is favoured to describe biological composition, rather than sole reliance on the conventional taxonomic approach (Walker 1992). *Functional diversity* can be identified with "the relative abundances of functionally different kinds of organisms" (Walker 1992, 19), where function is related to ecosystem processes or ecosystem services such as predators, herbivores, pollinators, decomposers, water-flow modifiers or nutrient transporters.

In order to examine ecological redundancy Walker (1992) suggests a procedure in four steps. The first step is to identify functionally different kinds of organisms within an ecosystem with respect to the way the biota regulates ecosystem processes. In other words, step one includes an analyses of dominant

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[76] The *Dictionary of Ecology, Evolution and Systematics* defines niche as „the ecological role of a species in a community; conceptualized as the multidimensional space, of which the coordinates are the various parameters representing the condition of existence of the species, to which it is restricted by the presence of competitor species" (Lincoln, Boxshall & Clark 1998).

[77] The term *redundancy* has problematic connotations since redundant literally means unnecessary. With regard to ecosystem-functioning, redundant species are not conceived as superfluous but as *natural insurance capital* (Folke, Holling & Perrings 1996) that represents a critical component of ecosystem resilience. Referring to the terminological problem of *redundancy*, Naeem (1998) suggest to focus on the term *ecosystem reliability* in order to emphasis the importance of ecological redundancy in providing functional compensation and response diversity.

ecosystem functions[78] that are considered as important (according to the purpose and desired regime) coupled with a functional classification of the biota through guild analysis[79]. The objective should be to further subdivide the species in a guild on the basis of functional attributes that are related to dominant ecosystem processes. ”If this cannot be done and there are still several different species in the group, then on the basis of current knowledge, there is some ecological redundancy within the guild concerned” (Walker 1992, 21). Step two is to determine the number of species within each guild. Those represented by only a few or even a single species are clearly unable to withstand any loss of species and constitute an obvious, immediate conservation focus. Step three is to further examine the interactions among the species in each guild. Complete functional redundancy only occurs if, following the removal of one species, there is density compensation among the remaining species[80] (Walker 1992). Or as Gitay et al. put it: “the basic concept of redundancy is that if a species is removed, and the community remains constant, then that species was redundant”[81] (Gitay et al. 1996, 122). Step four is to consider the relative importance of the functional groups, i.e. how a change in abundance of a functional group directly affects ecosystem and community processes, and how such a change influences the net effect of the biota.

Another aspect of the ecosystem view of conservation, i.e. to ensure ecosystem resilience, is to examine which kinds of species and functional groups are most important to ecosystem function. Walker (1992) proposes the *driver and passenger analogy* [82]. At one extreme, some species are determinants of the ecosystem, they are *drivers*. At the other extreme are those that are *passengers* that exist at the whim of the determinant species. Loss of passengers leads to little change in the rest of the ecosystem. A crucial point, however, is that apparent passengers at one time scale, may turn out to be infrequent determinants. Hence, there are three categories of species: the *existing drivers*, the *true passengers* and

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[78] Following the view of the *Resilience Alliance* explored in sections 3.2.2.2, ecosystems can be described by a small set of variables each operating at distinct temporal and spatial scales. Hence, the *Resilience Alliance* would concur that this part of step one in Walker’s procedure is feasible.

[79] Conduction of step 1 in Walker’s analysis might be difficult. There are no standardized sets of functional groups that are recognized for the numerous ecosystems that exist in nature. Davic (2003), however, in his sound review of the keystone species concept and its relation to functional groups, suggests a route toward the discovery of functional groups using evolutionary history, feeding guilds and foraging patterns.

[80] *Density compensation* means that the loss of some species may lead to an increase in abundance of others (Walker 1992).

[81] What „constant“ means and refers to in this respect is pointed out below [cf. Gitay et al. (1996)].

[82] Gorke (1996) reminds us that these analogies do not reflect reality as such, rather analogies are methodological projections that can be more or less accurate.

passengers that are *potential drivers* (Walker 1995). The focus of conservation is then directed to drivers and potential drivers within ecosystems.

According to Walker (1992), the species-centred conservation approach, however, is not superfluous, but can be used as complementary tool depending on the nature of the system.

Whereas the *rivet hypothesis* (Ehrlich & Ehrlich 1981) assumes that ecological function is evenly partitioned among species, 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. Peterson, Allen & Holling (1998) suggest that both models that stress ecological redundancy 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 function among species.

Gitay et al. (1996) provide a constructive critique of the Walker-approach to ecological redundancy and conservation. The authors claim that one has to take into consideration *what* has to remain constant before we can declare redundancy. Suppose a redundant species is lost. Does that mean that there is no effect on abundance of remaining species in the community, or, solely that remaining species should all remain present, or, that some measure of ecosystem function should stay constant? Moreover, Gitay et al. (1996) pose the question how redundancy should be determined and doubt the meaningfulness and feasibility of the methods concerned. Reacting on this, scientist now often 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 (WBGU 2000).

There is, however, empirical evidence of some type of ecological redundancy within various ecosystems. Reviewing the evidence relevant to the subject, Peterson, Allen & Holling (1998) conclude that some ecosystems possess considerable redundancy. Investigating the history of human impact on coastal ecosystems, Jackson et al. propose that

> "ecological diversity and redundancy within trophic levels is probably the most important reason for the delay or time lag between the onset of fishing and the subsequent threshold response" (Jackson et al. 2001, 636).

Walker (1995) terms species of functional groups that have little or no redundancy *keystone species* since their loss has great impact on the abundances of many other species. This view contributes to a decade-long controversy over the use of the *keystone species concept*. Davic (2003) links the original keystone species concept of Paine (1969)[83] and Bond (1994), respectively, to functional groups and specific ecosystem processes in order to approach a concept that is both operational and falsifiable. An operational definition identifies a *keystone species* 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, 3). Davic (2003) considers both functional groups as identifiable and his concept as appropriate to identify keystone species *a priori*.

Similarly, functional groups that are represented by a single or only a few species are dubbed *keystone functional groups*. Conservation focus shifts to the species in functional groups with low redundancy since the loss of these species would result in loss of ecosystem function. According to Walker (1995), however, not all one-species functional groups are keystone functional groups and, thus, require priority conservation attention since some dominant keystone species are themselves adapted to a wide range of environmental conditions.

When considering the relevance for a Theory of Sustainable Development, a focus on key processes, keystone process species and little-redundancy-parts of the functioning of ecosystems in my view represents a clear mean in order to identify critical natural capital.

3.3.3.2 Response Diversity & Insurance Hypothesis

Ecosystems are seen to exhibit functional groups that contain few or many redundant species (Walker 1992, 1995). If a functional group comprises more than one species, responses of these species to various disturbances are likely to be different. The variability in responses is termed *response diversity* and defined as "the diversity of responses to environmental change among species that contribute to the same ecosystem function" (Elmqvist et al. 2003, 488). In this respect, it is

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[83] Paine (1969) defines a keystone species as "a species of high trophic status whose activities exert a disproportionate influence on the pattern of species diversity in a community" (quoted in Davic 2003, 1)

noticeable that an aspect of ecosystem resilience corresponds to "the amount of disturbance a system can withstand and still retain the same controls of function and structure" (Walker et al. 2002) which I term the original-ecological meaning of ecosystem resilience and which matches the *original-ecological meaning* of ecosystem resilience *sensu* Holling (1973) (cf. section 3.4.3). If diverse species within the same functional group respond to disturbance regimes un-identical the possibility increases that this functional group is able to perform its function despite various disturbance regimes due to a mechanism termed *functional compensation*, i.e. structural change within communities that stabilizes ecosystem process rates (Frost et al. 1995, Carpenter & Cottingham 2002). Response diversity and compensation within functional groups is seen to be *critical for ecosystem resilience* (Walker 1995, Naeem 1998, Holling et al. 1995, Yachi & Loreau 1999, Elmqvist et al. 2003) as *buffering effect* to various disturbances. Each species can be critical and contributes to the ecosystem resilience (Gunderson 2000). This claim is dubbed the *insurance hypothesis* (Yachi & Loreau 1999).

In this respect, Folke, Holling & Perrings (1996) distinguish between *key process species* that seem to drive or control the critical processes necessary for ecosystem functioning [*drivers* according to Walker (1992, 1995)], and those species that guarantee the continuation of the ecosystem in a fluctuating environment that appear to be responsible for ecosystem resilience. "The vulnerability of key structuring processes is a function of the number of organisms that can take over and run such processes when the system is perturbed" (Folke, Holling & Perrings 1996, 1020).

Theoretically, the insurance effect is determined by three factors: (1) the way ecosystem processes are determined by individual species responses to environmental fluctuations, (2) the degree of asynchronicity of these responses, and (3) their detailed characteristic including their range of variation (Yachi & Loreau 1999). According to Elmqvist et al. (2003), the insurance metaphor, however, should focus on how to sustain ecosystem capacity to cope with and adapt to change in the context of alternative stable regimes and human-dominated environments.

Providing an empirical example, Walker et al. (1999) examined a savannah in Queensland, Australia, and propose that some groups of dominant and minor species within an ecosystem are functionally similar, and that this functional similarity provides "buffering" against perturbations or environmental variability. The species that dominate under a given set of environmental conditions serve to maintain ecosystem functioning under those conditions. Minor species, on the other hand, will be functionally similar to dominant species, but with different environmental requirements and tolerances. These species increase in abundance in response to a decrease in their functional equivalent due to a given disturbance regime. Thus, they maintain ecosystem resilience of the key processes and functions in the face of changing conditions.

The concept of response diversity does not imply that high species diversity necessarily entails high ecosystem resilience or *vice versa* - species-rich areas may be highly vulnerable to environmental change (Elmqvist et al. 2003). Summarizing the results of the Ecosystem Functioning of Biodiversity Program launched by the Scientific Commitee on Problems of the Environment (SCOPE) and the Global Biodiversity Assessment (GBA), Mooney et al. (1996) conclude that sensitive ecological systems *all* have low representation of key functional types, i.e. little functional redundancy within functional groups (cf. also WBGU 2000).

In this sense, ecosystem resilience depends both on the amount of redundancy within the functional groups and their response diversity rather than only on species richness. Both functional diversity and response diversity can be important for *ecosystem reliability* (Naeem 1998) since functional diversity increases the performance of the community in a complementary way[84] and response diversity enables the community to keep performing in the face of disturbances (Elmqvist et al. 2003).

However, it is conceivable that only a small part of world’s species might be responsible and sufficient for the maintenance of human welfare. The ecological insurance argument for species conservation is only convincing if there is a proof that the extinction of a given species represents a real danger to human welfare. Otherwise it appears to be not powerful enough to compete with “hard” economic arguments within practical discourses (Gorke 1996). However, it is not regularly possible to quantify the number of species which make a functional ecosystem and to predict the consequences of species loss. If species are lost and species complement is not at the maximum which the ecosystem can include, then the

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[84] Note, however, that there might be no single, generalizable relationship between species diversity and the productivity of an ecosystem, because the relative contribution of species to productivity change with environmental context (Cardinale, Nelson & Palmer 2000). Further, the mechanisms producing the relationship can change from sampling effects on a small scale to effects from resource partitioning, dispersal and disturbance on a larger scale (Cardinale, Ives & Inchausti 2004).

reduced richness at least lays the ecosystem open to invasion and disturbance (Woodward 1994).

For any given ecosystem property, determining the effects of reduced species diversity requires knowledge of (1) the functional roles played by individual species in governing the property of interest; (2) the degree to which species interactions affect functional roles, and (3) the likelihood of local extinction of individual species. The importance of biodiversity, thus, depends on both species identity and the order by which species are lost (Ostfeld & LoGuidice 2003).

However, even if species within functional groups are genuinely redundant, i.e. the functional group comprises ecological equivalents with respect to both ecosystem function and responses to disturbances (which is, of course, hard to identify), there can be still good reasons for protecting these species.

In philosophy, *axiology* represents the theory of values. In this respect, Barsch et al. (2002) distinguish between four *categories* of values: *anthropocentric instrumental value* (“good” as means for humans), *bio-related instrumental value* (“good” for non-humans), *eudaimonistic intrinsic value* (“good” as ends for humans), and *inherent moral value* (“end in itself”).

In this sense, the insurance-function of ecologically redundant species represents an anthropocentric instrumental value, and is often dubbed *insurance value*.

Eudaimonistic intrinsic values apply if human beings value some thing or some activity as being good for them *as such*. This category is related to the notion of a “good human life” and comprises several *types* of value. Gorke (1996) lists intuition, economic values, and aesthetic values as examples for eudaimonistic reasons for species conservation. For instance, although a species is considered to be “genuinely redundant” with respect to ecosystem function and response diversity it could be that beautiful that people still value it enough to preserve it.

Additionally, inherent moral value can be taken into account if the moral community is expanded beyond anthropocentrism which results that more and more natural entities and even whole species (depending on the environmental ethics theory) would be taken into consideration morally. Gorke (1996), for example, considers eudaimonistic reasons for species conservation as too weak and rather not convincing enough to justify restrictions to human growth. He argues for ethical holism which results in a position that values each species in the first place just because it exists (Gorke 1996, 1996b). Ott (2003d) considers sentientism as the best-founded position in environmental ethics and approves the application of eudaimonistic arguments for species conservation.

3.3.3.3 Imbricated Resilience

Biodiversity provides buffering capacity within functional groups (Walker 1995, Yachi & Loreau 1999). This ecological redundancy, i.e. the compensating overlap of ecological function *at the same scale* results in ecosystem resilience if it provides insurance capital for the performance of ecosystem functioning. Since those species perform at the same scale the result or pattern is termed *within-scale resilience* (Peterson, Allen & Holling 1998, Holling et al. 2002, Gunderson et al. 2002). Ecosystems, however, are structured hierarchically in a lumpy manner across various scales (Holling 2001, Holling & Gunderson 2002). Ecosystem processes, such as seed dispersal, can be replicated at different scales (Holling et al. 2002, Gunderson et al. 2002). Thus, biodiversity can influence ecosystem functioning across scales.

> “If species in a functional group operate at different scales, they provide mutual reinforcement that contributes to the resilience of a function, while at the same time minimizing competition among species within a functional group” (Peterson, Allen & Holling 1998, 13).

The stability property that arises of these mechanisms is dubbed *(a)cross-scale resilience* (Peterson, Allen & Holling 1998, Holling et al. 2002, Gunderson et al. 2002). Across-scale resilience, thus, represents a crucial aspect of ecosystem resilience since it provides ecological redundancy with respect to ecosystem functioning. Most disturbances, for instance, occur at specific scales, leaving similar functions that operate at other scales unaffected (Gunderson et al. 2002).

Within-scale resilience, thus, complements across-scale resilience and both are summarized in the concept of *imbricated resilience* (Holling et al. 2002). Ecosystem resilience, in this sense, derives from overlapping function within scales and reinforcement of function across scales (Peterson, Allen & Holling 1998) as a product of functional diversity and ecological redundancy.

It is difficult to envision how ecosystems without redundancy could continue to persist in the face of disturbance[85] (Peterson, Allen & Holling 1998). The way in general many biological processes are regulated seems to consist in overlapping influences by multiple processes, each one of which is inefficient in its individual effect but together operating in a robust manner (Gunderson 2000). The distribution of functional diversity within and across scales allows regeneration and renewal to occur following disturbances over a wide range of scales (Peterson, Allen & Holling 1998) which links to the *Panarchy* and adaptive cycle heuristics that include small-scale disturbances and renewal as essential parts of ecosystem dynamics.

Thus, along with other mechanisms explored below within-scale and across-scale resilience create the conditions that are robust [*sensu* Hansson & Helgesson (2003)] enough to allow for renewal and small-scale disturbances [the *renewal aspect of resilience* (cf. section 3.3.3.5)].

Eventually, Peterson, Allen & Holling (1998) state that ecological redundancy does not reside within functional groups only, but rather it emerges from the interactions of species. “Therefore, it is not possible to substitute species for one another; rather, there are many possible combinations and organizations of species that can produce similar ecological function” (Peterson, Allen & Holling 1998, 10)[86].

3.3.3.4 Ecological Memory & Spatial Resilience

Another key component of ecosystem resilience is represented by the concept of *ecological memory*, which is defined as “the network of species, their dynamic interactions between each other and the environment, and the combination

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[85] This represents an explanation of how functionally redundant species could evolve. If each species is selected for its survival attributes, how does a functionally complementary set eventuate (cf. Walker et al. 1999)? “A particular environmental pattern favors a particular suite of species, and the dominants among these are sorted out through performance, resulting in a complementary (rather than strongly overlapping, intensely competing) set. Those that lose out in the competition (and they would be species that *do* strongly overlap in performance with dominants) are either eliminated or relegated to minor species status. If an environmental change leads to a decline in the dominant, the minor species that emerges to replace it is one that can both thrive under the new environmental conditions and also complement the performance of the remaining dominants. A complementary pattern of functional attributes is therefore favored, leading to persistence of the existing levels of function. The continuous interplay between ecosystem form and function, between the players and the performance, ensures that the nature of the species composition of a community tends to a combination of functional diversity and redundancy” (Walker et al. 1999, 112).

[86] This is related the substitutability debate within sustainability science (cf. section 2.2.2). Ecological redundancy that emerges not only from the single species but also from the interactions among them provides a good argument against the substitution of natural capital through manufactured capital.

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 (…) [that] includes the life-history experience with environmental fluctuations” (Nyström & Folke 2001, 407). Ecological memory comprises within-patch memory as well as memory from surrounding habitats[87] (Bengtsson et al. 2003).

Internal memory comprises *biological legacies* which include 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) (Franklin & MacMahon 2000). Internal memory is limited by the assembly rules (e.g. facilitation, competition, trophic interactions) that determine which species proliferate despite or after disturbance.

External memory is provided by support areas of colonizing species and is restricted by the permeability of the matrix between the colonized patches[88] (Bengtsson et al. 2003). This “buffering capacity” depending on areas in the vicinity of the patch affected by disturbance is termed *spatial resilience* (Nyström and Folke 2001, Bengtsson et al. 2003).

In this respect, Walker (2002) points to the relevance of *patchiness*, i.e. spatial patterning or *landscape diversity*, with respect to rangelands. In this case, loss of patchiness leads to increased run-off surfaces and often to an increase in woody plants in the run-on areas, with increased net loss of soil and therefore a reduction in overall productivity. Walker (2002) suggests that decreasing spatial heterogeneity indicates declining ecosystem resilience.

Similarly, van de Koppel & Rietkerk (2004) propose that spatial interactions on coarser scales increase both engineering and ecosystem resilience of vegetation cover in arid regions facing variable rainfall levels. “Coarse-scale catastrophic shifts between ecosystem states are more likely in systems that have little spatial heterogeneity (…)” (van de Koppel & Rietkerk 2004, 119). In this case, spatial heterogeneity, thus, spatial resilience, reduces the probability of an ecosystem to shift to an alternative basin of attraction.

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[87] This phenomena has been described with respect to engineering resilience by Grubb and Hopkins (1986). The authors distinguish *in-situ resilience* from *resilience by migration* (Grubb and Hopkins 1986, 21ff).

[88] The role of surrounding habitats is emphasized by *metapopulation theory*. The permeability or *connectivity* of the matrix can also expose the ecosystem to invasive species and thereby reduce, or at least challenge, the ecosystem’s resilience (McClanahan et al. 2002).

Nyström & Folke (2001) illustrate the importance of spatial resilience with respect to coral reefs. Spatial resilience is here defined as “the dynamic capacity to cope with disturbance and avoid thresholds at spatial scales larger than individual ecosystems” (Nyström & Folke 2001, 407). The authors emphasis a matrix perspective in which the presence of a larger and more diverse species pool in which both primary and secondary successional organisms and the interactions between them are represented. The degree to which the mosaic of coral reefs can be modified by disturbance and chronic stress without losing its capability to maintain coral ecosystem functions and support the reorganization and reestablishment of deteriorated individual reefs is conceived as crucial. Nyström & Folke (2001) identify three basic and interacting parts of ecological memory that ensure spatial resilience: biotic legacies, support areas of potentially colonizing species, and mobile link organisms.

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) and represent a central component of ecological memory and, thus, spatial resilience. Examples are organisms that provide functions such as pollination, seed-dispersal, the translocation of nutrients, and grazing. Thus, mobile links can often have pivotal effects on ecosystem processes, especially following disturbance because they act as mediators of re-colonization[89].

Lundberg & Moberg (2003) distinguish between *resource linkers*, animals that transport and trans-locate essential resources, *genetic linkers*, species, that carry genetic information between habitats, such as seed dispersers and pollinators, and *process linkers*, organisms that connect habitats by providing, or supporting, an essential process. The authors conclude that mobile link organisms are crucial for ecosystem functioning and ecosystem resilience.

Despite historic debates about the relevance of the two components of ecological memory (internal and external), Bengtsson et al. (2003) find no conflict between the two components and state that, depending on disturbance intensity and landscape composition, their relative importance may vary.

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[89] Of course, mobile link organisms are not inherently good or bad for ecosystem resilience and function. In many cases mobile links support ecosystem resilience, whereas in other cases, they may be problematic (Lundberg & Moberg 2003). Alien species can be also considered as disturbance to ecosystems. There is a huge debate in ecology about issues related to invasive species. In the papers and books I used for this thesis, however, invasive species played a minor role.

The concept of ecological memory is closely related to the definition of ecosystem resilience *sensu* Walker et al. (2002) which comprises (1) the amount of disturbance a system can withstand and still retain the same controls of function and structure, (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. Ecological memory represents a critical component in the re-organization of ecosystems despite and after disturbance and represents sources for renewal from neighbouring patches. Thus, ecological memory can be crucial for ecosystem resilience.

Referring to theoretical work, Peterson concludes that “when ecological memory is strong, landscape pattern is persistent” (Peterson 2002, 329), i.e. ecosystem resilience is high. The key point of ecological memory and spatial resilience is that disturbances are seen as an intrinsic part of ecosystem development (Holling 1986, Gunderson & Holling 2002, Bengtsson et al. 2003) and therefore, the ecological memory of these systems, situated inside or outside the disturbed area, is of crucial importance for robustness and recovery (Bengtsson et al. 2003). Along with functional diversity and ecological redundancy within and across scales resulting in imbricated resilience, ecological memory and mobile links, summarized as spatial resilience, provide the conditions that are robust [*sensu* Hansson & Helgesson (2003)] enough to allow for renewal and small-scale disturbances [the renewal aspect of resilience (cf. section 3.3.3.5)].

3.3.3.5 Small-scale Disturbances

On the one hand, the resilience mechanisms explored above create a robust structure for the performance of ecosystem functioning and ecosystem resilience, respectively.

On the other hand, the *back loop of the adaptive cycle* (Ω and α phases) and the *Revolt-interaction* across scales as part of the panarchy takes into account the importance of renewal and variability in relatively small-scale patches or, more general, at each level of the ecosystem hierarchy. The entire back loop functions as small-scale disturbance and as engine for the *renewal of the ecosystem resilience of ecosystems*. As Walker & Abel point out:

> “The 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).[90]

This implies the recognition of the relevance of small-scale disturbances as *intrinsic parts* of ecosystem dynamics (Holling 1986, Gunderson & Holling 2002, Bengtsson et al. 2003), i.e. disturbance is endogenous to the cyclic process of ecosystem dynamics (Folke, Berkes & Colding 1998, Berkes & Folke 2003).

In this respect our provisional understanding of disturbance discussed in section 3.1.4 is being extended. Bengtsson et al. (2003), for instance, suggest that ecosystems are subject to pulse disturbances at various spatial and temporal scales. Ecosystems cannot be seen as static entities, rather, they represent always-changing, fluctuating, dynamic systems (Reichholf 1998) that show heterogeneity and patchiness on a landscape scale (Pickett & White 1985).

When considering complex adaptive systems the mechanisms for renewal and novelty represent a crucial aspect for ecosystem resilience on a larger scale. The concept of ecosystem resilience comprises both the capacity of ecosystems to absorb disturbance and, additionally, to provide the sources for reorganization, renewal, novelty, and development after disturbance (Nyström & Folke 2001, Gunderson 2000, Walker et al. 2002). 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 1998, 432). The structure and processes of the hierarchy can be reorganized. The adaptive cycle explicitly introduces rearrangements as a periodic process within each hierarchical level (i.e. small-scale disturbances) in a way that partially isolates the resulting experiments, reducing the risk to the ecosystem resilience of the whole structure (i.e. the landscape scale) (Resilience Alliance 2002). Bengtsson et al. (2003) term Holling’s four phase cycle rightly the *ecosystem renewal cycle*. 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

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[90] Note that Walker & Abel (2002) use resilience for the *extended-ecological meaning* of ecosystem resilience (cf. section 3.4.3).

> 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).

For Pickett & White (1985b) the greatest likelihood for persistence [*sensu* Grimm & Wissel (1997)] of landscapes is realized in systems in which disturbance is frequent and small in scale relative to an otherwise homogeneous area of habitat. In this sense, Gunderson states that the ecosystem resilience within a system "is generated by destroying and renewing systems at smaller, faster scales" (Gunderson 2000, 10).

This view on the persistence of ecosystem dynamics and structure depends on the notion that disturbances on smaller scales produce equilibrium or persistence at larger scales (Jentsch, Beierkuhnlein & White 2002). In this respect, Turner et al. (1993) predict both the presence and absence of equilibrium and variance in ecosystem states as a function of two ratios: the ratio of the disturbed area to the landscape area and the ratio of the disturbance frequency to the time needed for successional recovery. The smaller the patch relative to the recovery time, the greater the chance for dynamic equilibrium in all patches. Small-scale disturbance is seen to contribute to an increase in species richness, heterogeneity and function[91]. Jentsch, Beierkuhnlein & White suggest that "[d]isturbance increases species diversity producing redundancy in functions, and in turn functional redundancy ensures the persistence of functions in response to disturbance" (Jentsch, Beierkuhnlein & White 2002, 399).

I term this phenomena the *renewal aspect of ecosystem resilience* (similar Bengtsson et al. 2003). Disturbance regimes need to be incorporated in environmental management. As Berkes & Folke point out: "successful knowledge and resource management systems will allow disturbances to enter on a scale which does not disrupt the structure and functional performance of the ecosystem and the services it provides" (Berkes & Folke 1998, 21). Policies aimed at removing change and variation, i.e. disturbance, will cause an accumulation of such disturbances and a more widespread, large-scale crisis (Folke, Berkes & Colding 1998, Folke, Colding & Berkes 2003, cf. section 4.3.2.3).

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[91] The relationship between disturbance and biodiversity may change as one focuses on different levels of organization (Mooney et al. 1996).

3.3.3.6 Relevance of Abiotic Conditions

Abiotic conditions can be highly relevant for ecosystem dynamics and function. Grimm et al. (1999), for instance, point to the relevance of abiotic conditions in the Wadden Sea, Germany. “To enable the stability mechanism of the organisms to work, care must be taken that the abiotic processes which enable these mechanisms in the first place are a primary focus of conservation” (Grimm et al. 1999, 255).

Similarly, many of the case studies carried out by the *Resilience Alliance* show that abiotic variables play an important role for the provision of ecosystem resilience. Many of the *slow variables* chosen are abiotic factors (cf. section 3.2.4 and 4.2.3.3).

For example, Carpenter et al. (2002) identify soil and lake mud phosphorus as controlling slow variable for lakes in the Great Lake Region. Jansson & 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 & Walters (2002) specify also the slowly changing soil phosphorus level as controlling factor.

Walker & 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 an important point here.

The loss of several ecosystem resilience mechanisms such as ecological redundancy, and ecological memory that is related to biotic variables of the ecosystem results in systems that are more vulnerable to change. But, finally, it is then the changes in the slow variables that define the loss of ecosystem resilience (similar Gunderson & Walters 2002, 177) which corresponds to a decrease of the size of the basin of attraction in a stability landscape. In many cases, these slow variables tend to be abiotic variables. Obviously, controlling factors can be highly relevant for environmental management (cf. section 4.2.3.3).

Note that the examples of phosphorus in lakes or the Baltic appear to be more or less flogged to death. To me, one of the most important research topics of an environmental management approach for ecosystem resilience represents the identification of these slow variables that control ecosystem behaviour for other ecosystem types than aquatic ones.

3.3.4 Concluding Remarks on Ecosystem Resilience Mechanisms

Most of the mechanisms that were outlined in the previous sections and are responsible for ecosystem resilience occur on the community or ecosystem level. However, in theory, ecosystem resilience mechanisms can be sought at several levels of an ecosystem’s hierarchy. Lugo et al. (2002) suggest five levels where resilience mechanisms can be found. (1) In individuals as part of their responses to their environment. Examples are high reproduction, high mobility, phenotypic plasticity, flexible feeding behaviour and physiological tolerance, which represent mechanisms Grimm et al. (1999) identified for the Wadden Sea, Germany. (2) In the cumulative effect of how organisms of different species react to their respective environments, e.g. functional diversity and response diversity as cumulative mechanisms to absorb disturbances. (3) In the effect of legacies after an event, e.g. biotic legacies. (4) As the consequence of inputs from, or effects of, processes from other levels in the hierarchy, e.g. internal and external ecological memory. (5) As inherent characteristics of ecological systems, such as negative feedback function of storages.

There is another important point here.

On the one hand, the concepts of ecological redundancy, response diversity, ecological memory, and mobile links, producing *imbricated resilience* and *spatial resilience* (cf. sections 3.3.3.1 - 3.3.3.4), are seen as *ecosystem resilience mechanisms*, hence as mechanisms that result in the emergent property of ecosystem resilience. These mechanisms that all are related to biodiversity in the inclusive sense can be considered as contributing to the “Biodiversity-Ecosystem Function Paradigm” (BEFP) (*paradigm 2*) since biodiversity is seen to play an essential role with respect to ecosystem processes and is essential for the emergence of stability properties.

On the other hand, slow variables that often correspond to abiotic variables are considered to be important for the ecosystem resilience of ecosystems (cf. section 3.3.3.6). The influence of abiotic conditions can be regarded as part of the paradigm of community ecology (*paradigm 1*) as abiotic factors are seen as essential for ecosystem processes and stability properties. The paradigm of community ecology does not admit to a strong role for biodiversity and their relevance for ecosystem processes in understanding nature.

However, what mechanisms, factors, and variables are in fact decisive for the stability property ecosystem resilience?

In my view, scientific examinations about ecosystem resilience mechanisms can be seen in the light of the *ecological dialectic* proposed by Naeem (2002) explored in section 3.3.1. To explain mechanisms that are responsible for ecosystem resilience is not about a decision of *either* biotic factors such as ecological redundancy and ecological memory *or* abiotic factors such as nutrient storages or soil structure. Rather, the two paradigms create an dialectical process between theses (*paradigm 1*) and antitheses (*paradigm 2*) that leads to syntheses and creates further cycles of ecological dialectics. This dialectic is seen to nourish ecological progress rather than reflecting accurate representations of nature (Naeem 2002). In fact, the property ecosystem resilience emerges through many interacting factors and mechanisms operating at various temporal and spatial scales.

3.4 Ecosystem Resilience: Concluding Discussion

On a theoretical basis we have identified the background theory of the concept of ecosystem resilience (cf. section 3.2) as well as the mechanisms that are seen as responsible for the emergence of the stability property ecosystem resilience on the ecosystem level (cf. section 3.3.3). The subsequent sections will, first, examine some attempts to model ecosystem resilience (section 3.4.1), then, question whether investigations should focus solely on ecosystem resilience (section 3.4.2), and finally, distinguish several distinct meanings of the ecosystem resilience concept (section 3.4.3). Together these sections provide a concluding discussion of the theoretical aspects of the concept of ecosystem resilience.

3.4.1 Models

The term resilience has been defined originally in two different ways referring to two distinct stability properties (cf. section 3.1.4). The emphasis on one of these properties is not only a conceptual marginality, rather it reflects two different *paradigms* in ecology (Gunderson & Holling 2002).

Ludwig, Walker & Holling (1997, 2002) propose ecological models in order to illustrate the implicit assumptions of these two properties. Hereby, slow variables are considered to be responsible for the emergence of basins of attraction and stability properties. For the reason that I am not especially familiar with mathematical modelling I will just present what Ludwig, Walker & Holling (2002) propose.

Engineering resilience defined as the time required for a system to return to a steady state following a disturbance event or the “tendency of a system to recover or return to (or close to) its original state after a perturbation” termed *resilience sensu* Hansson & Helgesson (2003), presupposes that the regime will maintain its structure and function under any sort of disturbance event. Such an assumption may be made when we make large modifications to natural systems. Our expectation is that things will proceed more or less as before, and that the response of the system will be approximately proportional (linear) to the perturbation. This reflects an implicit assumption of *global stability*, i.e. there is only one equilibrium and the system will always return to it (Gunderson 2000). Suppose that the dynamics are given by a relation of the form:

$$\frac{dx}{dt} = h(\alpha) - x \qquad (1)$$

Where h(α) is a smoothly varying function of an external variable α and x is the quantity of interest. Then dx/ dt = 0 if x = h (α) ; the system has a single equilibrium there. This equilibrium is stable, since dx/ dt > 0 if x < h(α) and dx/ dt < 0 if x >h(α). These relations imply that the system approaches the equilibrium, no matter what the starting point.

A system such as equation (1) cannot fail or surprise us. It returns to an equilibrium, no matter how far it is displaced, and the position of the equilibrium changes smoothly with the exogenous variable α. Such a system is not suitable for a discussion of possible collapses of natural systems, since such collapses are excluded by assumptions such as equation (1).

In contrast, ecosystem resilience defined in its original-ecological sense as “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior” (Gunderson and Holling 2002, 4) presupposes the existence of alternative basins of attraction, thus, several equilibria or attractors, and non-linear behaviour as ecosystems can exhibit drastic shifts from one basin of attraction to the other that is due to only tiny changes in slow or fast variables. To explore non-linear behaviour suppose the following equation which is an example that has three equilibria:

$$\frac{dx}{dt} = f(x) = x(x^2 - \alpha) \qquad (2)$$

Here, α is a parameter or a slowly varying quantity whose dynamics are not of immediate concern. The equilibria of the system are the states where f(x) = 0. These are the states where either of the two factors in equation (2) vanishes. Hence, they are points where

$$x = 0 \text{ or } x^2 = \alpha \qquad (3)$$

If α > 0, then there are three equilibria, namely,

$$x = 0,\ x = \sqrt{\alpha}\text{, or } x = -\sqrt{\alpha} \tag{4}$$

If α ≤ 0, then there is only the single equilibrium at x = 0.

In order to determine the *stability of equilibria* (Ludwig, Walker & Holling 2002), it suffices to examine the sign of velocity of x. For example, if α < 0, the second factor in equation (2) is always positive and, hence, dx/ dt > 0 if x > 0, and dx/ dt > 0 if x < 0. In this case, the system always moves away from the state where x = 0. Hence, Ludwig, Walker & Holling (2002) conclude that the equilibrium at x = 0 is unstable if α < 0. On the other hand, if α > 0, then dx/ dt changes sign at three places:

$$\frac{dx}{dt} > 0 \text{ if } x > \sqrt{\alpha} \tag{5}$$

$$\frac{dx}{dt} < 0 \text{ if } 0 < x < \sqrt{\alpha} \tag{6}$$

$$\frac{dx}{dt} < 0 \text{ if } -\sqrt{\alpha} < x < 0 \tag{7}$$

$$\frac{dx}{dt} < 0 \text{ if } x < -\sqrt{\alpha} \tag{8}$$

The equilibrium where x = $\sqrt{\alpha}$ is unstable, because the system always moves away from that point if nearby [according to equations (5) and (6)]. Similarly, the equilibrium where x = $-\sqrt{\alpha}$ is unstable [according to equations (7) and (8)]. On the other hand, the equilibrium at where x = 0 is stable [according to equations (6) and (7)], because the motion from nearby points is toward that point. However, if the system starts outside the interval ($-\sqrt{\alpha}$ < x < $\sqrt{\alpha}$), it moves away from the equilibrium at x = 0. Therefore, the equilibrium at x = 0 is *locally stable*, but not globally stable (Ludwig, Walker & Holling 2002). The system returns to x = 0 if small perturbations take the system into an unstable domain. The interval is called the *basin of attraction* of the point x = 0, because trajectories that start within that interval eventually return to x = 0, but not those that start outside[92]. If α decreases towards

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[92] Note that Ludwig, Walker & Holling (2002) use *domain of attraction* instead of basin of attraction for the same meaning.

zero, the basin of attraction shrinks and the three equilibria collapse into one where $\alpha = 0$, and only a single unstable equilibrium remains when $\alpha < 0$.

We have seen that if $\alpha > 0$ then this system approaches the stable equilibrium at $x = 0$ if it is started within the basin of attraction. If we envisage disturbances that displace the system a distance x_1 from the stable equilibrium, they will not affect the integrity of the system (its tendency to return to the 0 state) as long as $x_1^2 < \alpha$. Now, if we allow the parameter α to decrease slowly toward x_1^2, the system will take longer and longer to return to the state $x = 0$ when x is displaced to x_1 because motion is very slow near $x = \sqrt{\alpha}$, and a disturbance of magnitude x_1 may take the system into the region of slow dynamics. Ludwig, Walker & Holling (2002) suggest to think of the *decrease in α as causing a loss of ecosystem resilience*, because the integrity of the system is threatened more and more by disturbances of a given magnitude. A symptom of loss of ecosystem resilience may be that it takes longer and longer to return to the vicinity of $x = 0$ after disturbance. The *connection between* return times and ecosystem resilience, hence between *engineering resilience and ecosystem resilience*, is hereby not completely straightforward. This connection is, of course, of high interest for this thesis and we will refer to this issue below.

The preceding examples reflect not believable models for natural systems since they predict that the state variable may approach infinity under some circumstances. More plausible models contain two stable equilibria and are dubbed *bistable systems*. Ludwig, Walker & Holling (2002) obtain a simple prototype for such systems by changing the sign dx/ dt in equation 2. If the direction of time is reversed, the stable and unstable equilibria are interchanged.

$$\frac{dx}{dt} = -f(x) = -x(x^2 - \alpha) \tag{9}$$

If $\alpha > 0$ for this system, we have

$$\frac{dx}{dt} < 0 \text{ if } x > \sqrt{\alpha} \tag{10}$$

$$\frac{dx}{dt} > 0 \text{ if } 0 < x < \sqrt{\alpha} \tag{11}$$

$$\frac{dx}{dt} < 0 \text{ if } -\sqrt{\alpha} < x < 0 \qquad (12)$$

$$\frac{dx}{dt} > 0 \text{ if } x < -\sqrt{\alpha} \qquad (13)$$

If x > 0 initially, then x heads toward the equilibrium at $x = \sqrt{\alpha}$, but if x < 0 initially, then x heads toward the equilibrium at $x = -\sqrt{\alpha}$. Thus, the basin of attraction of the point $x = \sqrt{\alpha}$ is the positive x-axis, and the basin of attraction of $x = -\sqrt{\alpha}$ is the negative x-axis. Each of the stable equilibria is locally stable, but not globally stable. This system can be flipped from one stable state to another by crossing the unstable line where x = 0. Because this line separates the basins of attraction, it is termed a *separatrix* (Ludwig, Walker & Holling 1997) and corresponds to an ecological threshold.

If α < 0, then the system will return to the stable equilibrium at x = 0 no matter how large the disturbance. However, if α > 0, and the system starts near the lower basin of attraction, it will tend to return there is displaced by a small amount. As α decreases toward zero, the distance between the stable equilibria and the unstable one decreases. Hence, disturbances of a given magnitude take the system closer and closer to the unstable equilibrium. Dynamics are slow near the unstable equilibrium and, hence, the time to return to the vicinity of the lower basin of attraction increases sharply for trajectories that approach the unstable equilibrium. Again, return times play a role in models of ecosystem resilience.

For a higher level of disturbance and lower values of α the system may be moved across the separatrix more easily, whereas for larger values of α one would expect shifts from one equilibrium to the other to be extremely rare. Thus, *an increase in α may be associated with an increase in ecosystem resilience*.

The previous equations illustrate a *soft loss of stability* (Ludwig, Walker & Holling 2002). As the exogenous variable changes, the location of the stable equilibria changes smoothly. The state variable may move from one basin of attraction to another, but such changes are slow because dynamics are slow near an unstable equilibrium or a separatrix. The possibility of such behaviour would not ordinarily be cause for alarm, because slow dynamics may allow for adjustments to new behaviour. The following equations illustrate cases where more abrupt changes occur. Suppose the following equation:

$$\frac{dB}{dt} = r_B B(1 - \frac{B}{K_B}) - \beta \frac{\mathrm{B}^2}{\alpha + \mathrm{B}^2} \tag{14}$$

where r_B is an intrinsic growth rate at low densities, K_B is a carrying capacity for the budworm in the absence of predation, and the second term in equation 14 is a predation rate. The predators are assumed to have functional response with a maximum predation rate of β and a half-saturation budworm density of α. This functional form implies that predators have their greatest influence upon dynamics at intermediate ranges of budworm densities. The parameter α is proportional to a measure of foliage density, because the predators search foliage for the budworms and their response is mediated by the number of budworms per unit of foliage. Hence, α is actually a state variable that generally changes on a slower time scale than that of the budworm. For the moment we regard α as a constant.

There are either two or four equilibria for the budworm, depending on the size of the dimensions-less parameters R and Q, given by the equation

$$R = \frac{r_B \alpha}{\beta'} Q = \frac{K_B}{\alpha} \tag{15}$$

These equilibria satisfy equation,

$$R(1 - \frac{b}{Q}) - \frac{b^2}{1 + b^2} = 0 \tag{16}$$

where b = B/ α. The equilibrium b = 0 is always unstable, because db/ dt > 0 if b is small and positive. The highest equilibrium is always stable, because db/ dt < 0 if b is very large and positive. Thus, if there are only two equilibria, budworm density always moves toward the upper equilibrium. When there are four equilibria, they alternate in stability.

Imagine now that the parameter α begins at a low value and gradually increases as the forest grows. It turns out that Q does not change with forest growth. Because R is proportional to α, R will increase. At first (when $R < R_1$), budworm numbers will remain low, since the only stable equilibrium is the low one. Even when

R increases beyond R_1 the budworm numbers will remain low, because they lie below the unstable equilibrium, which determines the domain of attraction of the low equilibrium. The stability of the low equilibrium becomes precarious as R approaches R_2, because the basin of attraction shrinks. Finally, at $R = R_2$, the lower two equilibria disappear and budworm density jumps to the high value: an outbreak occurs. This abrupt change in the attracting state is coined a *hard loss of stability* (Ludwig, Walker & Holling 1997). It should be contrasted with the soft loss of stability. In the case of the budworm, once density has reached the high equilibrium there is no easy way to reduce it to the lower equilibrium. If the variable R is reduced below R_2, the budworm remains at the high equilibrium. As R is further reduced, there is a second hard loss of stability as R declines below R_1. In this case, there is a jump down to the equilibrium which is not reversed as R increases again.

If we now connect the dynamics of the trees and the dynamics of the budworm, a new phenomena appears. If the system starts with low foliage density and low budworm numbers, the foliage density slowly increases until it surpasses R_2. At this point, an outbreak occurs, as shown previously. High budworm numbers eventually cause death of trees, so R begins to decrease when the budworm has an outbreak. Budworm numbers remain high even though R declines, because budworm density lies above the separatrix. As R continues to decline to R_1, budworm density declines slowly and then jumps to a low value when R decreases below R_1. The different paths followed by the total system for increasing versus decreasing R constitute the *hysteresis effect* (Ludwig, Walker & Holling 2002, cf. section 3.2.3). The combination of budworm and forest dynamics produces stable cycles with long periods. Such stable cycles that are maintained through alterations of rapid transitions and slow changes are termed *relaxation oscillations* (Ludwig, Walker & Holling 2002).

If the objective of management is to keep budworm numbers and foliage damage low, the loss of stability as R increases beyond R_2 may be regarded as a loss of ecosystem resilience. This model suggests that small disturbances near the lower stable equilibrium may exhibit long return times if they approach the unstable equilibrium. However, because R increases as trees grow, a loss of stability accompanied by a budworm outbreak seems inevitable, which corresponds to the release phase of the adaptive cycle.

What, however, is the relation of return time and ecosystem resilience in these equations? What is, thus, the relation of ecosystem resilience and engineering resilience in these models?

According to Ludwig, Walker & Holling (2002), it is important to distinguish between behaviour near a stable equilibrium and behaviour near the boundary of a basin of attraction, which is an unstable equilibrium or separatrix.

For modelling engineering resilience, Pimm (1991) describes return to equilibrium by the equation

$$X_t - X^* = (X_0 - X^*)e^{-kt} \tag{17}$$

where X_t is the population density at time t, X_0 is the initial population density, and X^* is the equilibrium density. The differential equation for X_t that corresponds to this formula is given by

$$\frac{dX_t}{dt} = -k(X_t - X^*) \tag{18}$$

If we measure displacement from X^* by x, then x satisfies equation

$$\frac{dx}{dt} = -kx \tag{19}$$

Strictly speaking, Pimm's definition depends upon this simplicity, because the amount of time required for x to decay to some specified fraction of its initial value is only constant of the equation 17 is used. In fact, if the initial displacement is x_0 and the fraction is $p < 1$, then equation 17 implies the relationships of

$$x_1 = px_0 = x_0 \exp(-kt_r) \tag{20}$$

From equation 20 Ludwig, Walker & Holling (2002) conclude that the return time t_r is given by

$$t_r = \frac{1}{k}\log\frac{1}{p} \tag{21}$$

The remarkable feature is that the magnitude x_0 does not appear in this formula. In more general circumstances, such a result can be expected to hold only in the limit as x_0 approaches 0. Such results are called *local*. A common error is to extrapolate local results to global ones. In the present context, it amounts to replacing a complicated function by a linear approximation. Such approximations are certainly easy to work with, but they may miss essential features of the dynamics. In fact, failure to recognize the distinction between local stability and global stability can lead to unwarranted optimism about the likely consequences of interventions in natural systems. If we think that stability to small perturbations necessarily implies stability to large perturbations, then precautions are never required (Ludwig, Walker & Holling 2002).

In order to distinguish behaviour near the equilibrium at x = 0 from behaviour near an unstable equilibrium, we must use a model with more parameters than equation 18, such as

$$\frac{dx}{dt} = f_1(x) = \frac{x(x^2-\alpha)}{x^2\left[\frac{2}{k_1}-\frac{1}{k}\right]+\frac{\alpha}{\kappa}} \tag{22}$$

Equation 22 leads to an especially simple equation for the return time. The time to reach a position x_1 starting at x_0 is given by

$$t_r = \int_{x_0}^{x_1} dt = \int_{x_0}^{x_1} \frac{dx}{f_1(x)} \tag{23}$$

The form for $f_1(x)$ was chosen so that equation 24 can be verified algebraically.

$$\frac{1}{f_1(x)} = \frac{-1}{kx} + \frac{1}{k_1\left(x-\sqrt{\alpha}\right)} + \frac{1}{k_1\left(x+\sqrt{\alpha}\right)} \tag{24}$$

In view of equations 23 and 24, equation 25 emerges.

$$t_r = \frac{1}{k}\log\frac{x_0}{x_1} + \frac{1}{k_1}\log\frac{x_1-\sqrt{\alpha}}{x_0-\sqrt{\alpha}} + \frac{1}{k_1}\log\frac{x_1+\sqrt{\alpha}}{x_0+\sqrt{\alpha}} \tag{26}$$

Now, if we replace x_1 by px_0, equation 26 becomes equation 27.

$$t_r = \frac{1}{k}\log\frac{1}{p} + \frac{1}{k_1}\log\frac{1}{p_1} + \frac{1}{k_1}\log\frac{1}{p_2} \tag{27}$$

Here p_1 and p_2 are given in equations 28 and 29, respectively.

$$p_1 = \frac{x_0-\sqrt{\alpha}}{px_0-\sqrt{\alpha}} \tag{28}$$

$$p_2 = \frac{x_0+\sqrt{\alpha}}{px_0+\sqrt{\alpha}} \tag{29}$$

If the last two terms in equation 27 are omitted, this result is identical to Pimm's assumption according to equation 17. The more complicated dynamical assumption from Ludwig, Walker & Holling (2002) is the analogue of Pimm's assumption if there are three equilibria.

Now, under what conditions equation 27 implies long return times?

The first term, which corresponds to Pimm's model, implies a long return time if the ratio $p = x_1/ x_0$ is small or if k is small. In Pimm's discussion, p is a parameter that describes a probe or observation of the system. Ordinarily, p is fixed, and the return time provides an estimate for k.

The second term in equation in equation 27 implies a long return time if p_1 is small or k_1 is small. The previous discussion was concerned with a possibly variable α and disturbances that might take the system near an unstable equilibrium. That corresponds to x_0 near $\sqrt{\alpha}$ or x_0 near $-\sqrt{\alpha}$. In such case, t_r will be large even if the parameter k is large. That is, return time may be long, even for systems that show very rapid return when close to the stable equilibrium. According to this point of view, long return times may be diagnostic for a small α or for disturbances that are large enough to take the system near un unstable equilibrium (Ludwig, Walker & Holling

2002). If a disturbance takes the system beyond the unstable equilibrium, i.e. beyond the ecological threshold, there is no return at all.

Thus, according to Pimm (1991) and to Ludwig, Walker & Holling (2002), long return times may be a diagnostic tool for a loss of ecosystem resilience, but the meaning of the terms are quite different in the two cases. On the one hand, Pimm (1991) is concerned with behaviour near a stable equilibrium. In that case, a long return time for a given displacement from the equilibrium indicated a small coefficient k or, equivalently, a small derivative of log x. On the other hand, Ludwig, Walker & Holling (2002) are concerned with behaviour of a system with two or three equilibria, one of which is stable. Ecosystem resilience describes the tendency of the system to return to its stable equilibrium. A long return time is due to disturbances that bring the system near an unstable equilibrium.

Thus, engineering resilience and ecosystem resilience are closely related. The important distinction is the focus of study on slow dynamics near a stable point and an unstable point, respectively.

3.4.2 Focus on Ecosystem Resilience?

In contrast to the fact that there are three fundamentally distinct stability properties – constancy, resilience and robustness (or persistence) – the *Resilience Alliance* considers robustness, or, following their terminology, *ecosystem resilience* to be the concept that is more valuable both theoretically and with respect to the implementation of objectives related to Sustainable Development. As Peterson, Allen & Holling point out:

> “ecological resilience concentrates on the ability of a set of mutually reinforcing structures and processes to persist. It allows ecologists or managers to focus upon transitions between definable states, defined by sets of organizing processes and structures, and the likelihood of such occurrence. Engineering resilience, on the other hand, concentrates on conditions near a steady state where transient measurements of rate of return are made following disturbances. Engineering resilience focuses upon small portions of a system’s stability landscape, whereas ecological resilience focuses upon its contours. Engineering resilience does not help assess either the response of a system to large perturbations or when gradual changes in a system’s stability landscape may cause

> the system to move from one stability domain to another. For these reasons we concentrate on ecological resilience" (Peterson, Allen & Holling 1998, 11)[93].

Most authors participating in the *Resilience Alliance* state that the emphasis on one of the two different stability properties has very different consequences for evaluating, understanding, and managing ecological systems. The first definition focuses on efficiency, control, constancy, and predictability, the second definition focuses on robustness, adaptiveness, variability and unpredictability. These are fundamentally different views of science (e.g. Berkes & Folke 1998, Gunderson & Holling 2002).

Engineering resilience concentrates on constancy at a presumed steady-state, and stresses resistance to disturbance and the speed of return to the equilibrium point. This is the conventional, equilibrium-centred, linear, cause-and-effect view of a predictive science. In resource management this view leads to the assumption that resources are manageable and yields predictable (Berkes & Folke 1998).

A focus on ecosystem resilience encourages the view that determining causal effects and making predictions are not simple matters at all. Rather, systems are seen to be complex, non-linear, multi-equilibrium and self-organizing and permeated by uncertainty and discontinuities (Berkes & Folke 1998).

In the key book *Panarchy* published by the *Resilience Alliance*, Holling & Gunderson argue that

> "sustainable relationships between people and nature require an emphasis on the second definition of resilience, i.e., as the amount of disturbance that can be sustained before a change in system control and structure occurs – ecosystem resilience" (Holling & Gunderson 2002, 28).

The emphasis on engineering resilience may miss important aspects of a system's "stability". It leads to scientific examinations and results that often do not meet the appropriate landscape scale in order to be useful for objectives for Sustainable Development and environmental management (e.g. maintenance of

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[93] Note that Peterson, Allen & Holling (1998) use ecological resilience as synonym for ecosystem resilience and the term stability domain for the meaning of basin of attraction.

natural capital, preservation of biodiversity). Holling & Meffe suggest that, on the contrary,

> “an ecosystem-resilience perspective better reflects the reality of large-scale processes and dynamics and provides the most realistic foundation for addressing the challenging and complex resource management issues of the day” (Holling & Meffe 1996, 334).

The concept of engineering resilience does not take into account the existence of alternative basins of attraction and the non-linear behaviour of slow variables in the transition zone far from a stable attractor (Ludwig, Walker & Holling 1997). Thus, a focus on engineering resilience reinforces the pathology of equilibrium-centred command and control management which often leads to loss of ecosystem resilience on a longer time scale and to a shift to an often undesired basin of attraction (Holling & Meffe 1996). Similarly, Perrings (1998) suggests that the notion of ecosystem resilience is both more policy relevant and more testable in evolutionary and stochastic systems which, in turn, appear to be more relevant for sustainability. Thus, during the examination of this thesis I have stressed the background theory of *ecosystem* resilience (cf. sections 3.2).

However, the *Resilience Alliance* regards their emphasis on ecosystem resilience more as a paradigm rather than a reflection of reality in nature (Holling & Gunderson 2002, Gunderson & Pritchard 2002), i.e. they do not doubt that *other stability properties can be relevant for ecosystem analysis*.

Grimm et al., for instance, state that stability properties both of abiotic and biotic conditions, in the Wadden Sea, Germany, follow the same pattern:

> “on a short time scale (from days to a year) and small areas – low constancy (or high unpredictability); on an intermediate time scale (a few years to several decades) and larger areas – constancy (or low variability), with resilience in the case of disturbances; and on a long time scale (many decades or longer) and the entire Wadden Sea – persistence” (Grimm et al. 1999, 253).

Note that Grimm et al. (1999) use *persistence* for the meaning of the original-ecological meaning of ecosystem *resilience* and resilience for engineering resilience. The pattern illustrates an important point and that is that the relevance of different stability properties might depend on the temporal and spatial scale of concern. In this case, ecosystem resilience gets more important on larger scales.

From a different perspective, Carpenter & Cottingham state that

> “although return rate near a stable state may miss important features of a system’s resilience, empirical estimates of return rate may be useful for comparing responses of different systems to a given perturbation” (Carpenter & Cottingham 1997, 7).

As we have seen with respect to ecological thresholds, Wissel (1984) conceives return time and, hence, engineering resilience as appropriate tool in order to predict the position of ecological thresholds. Similarly, Grimm et al. (1999) regard resilience or engineering resilience defined as “returning to a reference state (or dynamic) after a temporary disturbance” (Grimm & Wissel 1997) as the most important stability concept to understand ecological systems. However, Grimm et al. (1999) use stability concepts in a different way, without connotations such as equilibrium. They consider assessments of stability properties as a diagnostic tool which helps organize our thinking about properties and processes.

In this respect, Lugo et al. (2002) use the engineering resilience concept to examine tropical wet and dry forests. Key sources of engineering resilience are represented by belowground nutrient storages, rapid fluxes of nutrients and biomass, biotic controls of nutrient loss, and ecological redundancy.

Additionally, the two properties of engineering resilience and ecosystem resilience often seem to be closely related as in Hansson & Helgesson’s (2003) analysis of stability concepts (cf. section 3.1.3) or as in the models of Ludwig, Walker & Holling (1997, 2002) (cf. section 3.4.1). According to the models long return times may be a diagnostic tool for small ecosystem resilience or for disturbances that are large enough to take the system near an unstable equilibrium.

In this respect, Scheffer et al. (2002) conclude that both interpretations of resilience, i.e. ecosystem resilience and engineering resilience are closely linked. In general, however, the relation between stability properties depends on the concrete ecosystem (Grimm & Wissel 1997).

Moreover, the definition of ecosystem resilience *sensu* Walker et al. (2004) includes “a measure of the maximum amount the system can be changed before losing its ability to recover”, which corresponds to the width of the domain of attraction and, in my opinion, to the ability to return to an attractor after disturbance, which is, in turn, closely related to engineering resilience. It is arguable whether the ecosystem resilience concept sensu Walker et al. (2004) includes the concept of engineering resilience or not. In this respect, Loreau et al. (2002) consider the view of deterministic autonomous systems at equilibrium, and the ecosystem resilience view as not necessarily contradictory.

To sum up, my emphasis on ecosystem resilience is justified only partly since other stability properties, engineering resilience in particular, can be highly relevant for certain ecosystem types.

The *Resilience Alliance* conceives the ecosystem resilience concept to be valuable not only for ecosystems (without the human sphere) but for social-ecological systems as well (Gunderson & Holling 2002). With respect to socio-economic systems, Batabyal criticizes the emphasis on the ecosystem resilience concept since "the concepts of persistence and resistance (…) can be just as important to our understanding of the behavior of socio-economic systems" (Batabyal 1998, 236)[94]. On occasion, engineering resilience, for instance, might be more relevant. Similarly, Hanley (1998) states that the ecosystem concept of resilience fails the cost-benefit test, when applied to the study of economic and social systems, because it offers no additional insights to those we have already, and appears to be poorly defined.

In my view the emphasis on ecosystem resilience is useful in order to recognize the importance of alternative basins of attraction, the non-linear behaviour and relevance of slow variables and ecosystem resilience as a changing quantity. It is useful in order to enlighten our notion of nature as pointed out in the view *Nature Evolving* put forward by the *Resilience Alliance* (cf. section 3.2.1 and 3.2.4). There is fundamental difference between the view that quantitative prediction is difficult and data intensive ('we need more research') and the view that nature is not equilibrium centred and inherently unpredictable (Berkes, Colding & Folke 2003). According to some authors, the linear, equilibrium-centred view of nature no longer fits the evidence, and is being replaced by a non-linear, multi-equilibrium view (Holling, Berkes & Folke 1998). As Carpenter et al. point out

> "the distinction of these two definitions of resilience has been useful in encouraging the managers of naturally variable systems to think about the persistence of such systems and has helped them to break away from their traditional preoccupation with management aimed at the unachievable goal of stability" (Carpenter et al. 2001, 766),

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[94] Batabyal (1998) uses persistence and resistance according to Pimm (1991). What is important here is that Batabyal considers other stability properties than ecosystem resilience to be important to understand socio-economic systems (1998).

i.e. to break away from an equilibrium-centred view on “stability” which is emphasised by engineering resilience.

On the other hand, the emphasis on ecosystem resilience is *not* useful, in my opinion, *if* it is to reflect reality of nature as if ecosystem resilience was the only observable stability property that ecosystems may perform. Other stability properties can be highly relevant for both mere ecological systems and socio-economic systems. I agree with Walker who points out that

> “too often in the development of ecology there has been a swing from one extreme to the other (the association vs. the continuum, equilibrium vs. disequilibrium, etc.) with the eventual realization that both approaches were valid and that the extent to which each was important depended on the nature of the system” (Walker 1992, 21).

All three fundamental stability properties may be relevant for specific ecological situations and ecosystem types.

3.4.3 Levels of Meaning

The term resilience was first established and defined by C.S. Holling in his paper *Resilience and stability of ecological systems* (Holling 1973). Since then, the term was used for two distinct stability properties, namely resilience *sensu* Hansson & Helgesson (2003) or engineering resilience *sensu* Gunderson & Holling (2002) and robustness *sensu* Hansson & Helgesson (2003) or ecosystem resilience *sensu* Gunderson & Holling (2002) (cf. section 3.1.4). During the examinations of this thesis we focused on ecosystem resilience since it is conceived as the more valuable concept both theoretically as well as operationally (cf. section 3.4.2).

Ecosystem resilience, in turn, is a concept that has multiple levels of meaning from the metaphorical to the specific (Carpenter et al. 2001).

In 1973, Holling originally defined (ecosystem) 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). Similarly, Holling & Gunderson defined ecosystem resilience as the “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour” (Holling &

Gunderson 2002, 4). This *original-ecological meaning* more or less corresponds to persistence *sensu* Grimm & Wissel (1997) and to robustness *sensu* Hansson & Helgesson (2003).

Due to the progress in ecological debates about related concepts [*Nature Evolving*, complex adaptive systems, alternative stable regimes and ecological thresholds (cf. section 3.2) as well as ecosystem resilience mechanisms (cf. section 3.3.3) the ecosystem resilience concept has been extended in its meaning which is reflected in its definition[95].

Referring to these debates, Walker et al. (2002) define ecosystem resilience as "the potential of a system to remain in a particular configuration and to maintain its feedbacks and functions, (…) [which] involves the ability of the system to reorganize following disturbance-driven change" (Walker et al. 2002). Similarly, Walker et al. (2004) identify ecosystem resilience as "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks" (Walker et al. 2004, 2).

In this *extended-ecological meaning*, ecosystem resilience comprises three defining characteristics: (1) the amount of disturbance a system can withstand and still retain the same controls of function and structure, (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). In my view, the last two characteristics extend[96] the meaning of the original definition from Holling (1973), as ecosystem resilience is seen to comprise not only the ability to absorb disturbances and to stay in the same basin of attraction but also to include the capability of self- and re-organization and renewal.

Note that all the investigations of ecosystem resilience conclude that ecosystem resilience is not something that is desirable as such (e.g. Carpenter et al. 2001, cf. section 4.2), i.e. it represents a descriptive rather than a normative concept. It is arguable, however, whether characteristic (3), i.e. the degree to which the system expresses capacity for learning and adaptation, is slightly normative.

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[95] Gunderson & Holling (2002) represents a book that develops the extended definition of ecosystem resilience, i.e. the extended notion is already comprised and specified within Gunderson & Holling (2002).

[96] Some authors suggest that the original definition from Holling (1973) already comprises the three defining characteristics (e.g. Carpenter et al. 2001). In my view the definitions suggested by Walker et al. (2002, 2004) provide at least an explication of the distinct aspects of the ecosystem concept if not an expansion of Holling's (1973) definition.

Referring to the expanded definition, Walker et al. (2004) distinguish four crucial aspects of ecosystem resilience three of which correspond to different characteristics of a basin of attraction (cf. Walker et al. 2004 and section 3.2.4). (1) *Latitude* is defined as the maximum amount the system can be changed before losing its ability to recover which corresponds to the width of the domain of attraction. (2) *Resistance* is defined as the ease or difficulty of changing the system which is related to the topology of the domain. (3) *Precariousness* is defined as the current trajectory of the system, and how close it currently is to an ecological threshold. The fourth aspect of ecosystem 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.

When proposing ecosystem resilience statements, one of the most crucial points is, in my view, that one has to refer to underlying, controlling, so-called *slow variables* (cf. section 3.2.4 and section 4.2.3.3). This corresponds to the suggestion that ecosystem resilience is more an emergent *capacity* of ecosystems which points to the relevance of underlying mechanisms that are responsible for this capacity. In this ecosystem sense, the *ecological-systemic meaning* of ecosystem resilience is, the capacity of ecosystems to maintain services in the face of a fluctuating environment and human perturbations.

In the ongoing debate on theoretical aspects of the ecosystem resilience concept and its relevance within a Theory of Sustainable Development, some authors recognized the importance of clear and measurable definitions. Carpenter et al. (2001), for instance, suggested an *operational meaning*, thus, a measurable concept of ecosystem resilience by clearly defining ecosystem resilience *of* what *to* what. Examinations of ecosystem resilience have to specify the time scale as well as the spatial scale, the reference state (which stable regime is being considered) and the disturbances of interest (Carpenter et al. 2001). Additionally, Jax, Jones & Pickett's (1998) self-identity of ecological units and Grimm & Wissel's (1997) ecological checklist provide tools in order to specify these sorts of stability statements appropriately. Various specifications are essential for a sound *ecosystem resilience analysis* (Walker et al. 2002, cf. section 4.2).

The focus of study within sustainability discourse is not on ecosystems and the environment only but rather on several other dimensions, such as the economic system and the social system. Therefore, corresponding to the concept of ecosystem resilience for ecosystems, some authors refer to social systems and identify *social resilience* as “the ability of human communities to withstand external shocks to their social infrastructure, such as environmental variability or social, economic, and political upheaval” (Adger 2000, 347). Clearly, social resilience refers to the original-ecological meaning of ecosystem resilience.

A crucial point in the work of the *Resilience Alliance* on ecosystem resilience is their claim that a sound ecosystem resilience analysis has to take into account not only the natural system, but rather whole social-ecological systems (Holling 1999, Folke et al. 2002, Gunderson & Holling 2002, Berkes, Colding & Folke 2003). The separation of social systems and natural systems is seen to be more of a recent mental artefact than an observation of the real world. Therefore, some authors identify *social-ecological resilience* as “the capacity of ecosystems to sustain societal development and progress with essential ecosystem services” (Folke, Colding & Berkes 2003, 354).

Definitions of social resilience and social-ecological resilience, respectively, are rather normative than descriptive concepts as they refer to states of nature that are valued due to human needs and wants and to terms such as *development* and *progress* that are obviously normative. In my view social-ecological resilience contains connotations with respect to the meaning of Sustainable Development (e.g. the Costanza et al.-definition, cf. section 2.2.1). Similarly, Ott (2001, 2003) uses the concept of (ecosystem) resilience as a guideline for a Theory of Sustainable Development (cf. section 2.2).

To sum up, the assessment of the different meanings of the concept of ecosystem resilience results in at least seven distinct but related definitions that emerged during the history of the term (cf. *Table 7*).

I regard the ecological definitions which specify ecological terms as useful descriptive concepts of ecosystem resilience. Additionally, the operational definition appears to be useful to measure and implement the concept of ecosystem resilience.

The social-ecological and the explicitly-normative definitions are, in my view, normative concepts. It is important to be clear about the descriptive part and the normative part of terms and to separate them carefully (Gorke 1996). For example, the terms *ecology* or *ecological* have been used for several meanings both descriptive and normative. This resulted in a confusion about the meaning of these terms in the public as they get mixed up with political objectives or notions of equilbria in nature.

Table 7: Ecosystem resilience: Seven levels of meaning
(my arrangement)

Level of meaning	**Definition**	**Reference**
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)
	Magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour	Holling & Gunderson (2002)
Extended-ecological	Capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks	Walker et al. (2004)
	(1) The amount of disturbance a system can withstand and still retain the same controls of function and structure, (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	Walker et al. (2002); Carpenter et al. (2001)
Ecological-systemic	The underlying capacity of an ecosystem to maintain ecosystem services in the face of a fluctuating environment and human perturbations	Carpenter et al. (2001); Folke et al. (2002); Deutsch, Skanberg & Folke. (2003)
Operational	Ecosystem resilience *of* what *to* what	Carpenter et al. (2001)
Sociological	The ability of human communities to withstand external shocks to their social infrastructure, such as environmental variability or social, economic, and political upheaval	Adger (2000)
Social-ecological	The capacity of ecosystems to sustain societal development and progress with essential ecosystem services	Folke, Colding & Berkes (2003)
Explicitly-normative	Maintenance of natural capital	Ott (2001)

Similarly, in the case of (ecosystem) resilience the ecological-scientific sense has been mixed up with the normative claim to preserve certain “states” in nature. In the international free encyclopedia *Wikipedia*, for instance, the anonymous author states that (ecosystem) resilience corresponds to the capacity to absorb and to reorganize after disturbance, to maintain the “vitality” of the ecosystem. The author suggests eroded forests as *counter*examples which lack the ability to reorganize, i.e. they have no or less (ecosystem) resilience (Wikipedia 2004). This view, in my opinion, is partly false in that it suggests that, in general, eroded, non-productive, low-diversity “states” of nature show low resilience including low reorganization after disturbances while human impact always results in loss of (ecosystem) resilience and, therefore, high-resilient states were desirable. In fact, eroded savannahs or polluted shallow lakes, for instance, can be highly resilient and highly-resilient “states” highly undesirable as eroded savannahs are for pastoralists (e.g. Carpenter et al. 2001, Walker et al. 2002, Gunderson and Holling 2002, Allison and Hobbs 2004). Thus, in this case, the descriptive ecological and the normative meaning of the ecosystem resilience concept have been mixed up.

It is important to be clear that the *choice* of a desired ecosystem regime depends on human values and on normative judgements, rather than on descriptive ecological facts (cf. section 4.2.3.2). Therefore, in my view, it would be better to describe the normative meanings (social-ecological and explicitly-normative definitions) with other terms (e.g. maintenance of natural capital, ecosystem, integrity, ecosystem health) in order to avoid confusion about different meanings of the term ecosystem resilience.

4 Operationalization and Implementation of Ecosystem Resilience

The ecosystem resilience concept assumes not only theoretical interest. For instance, both Ott (2001) and Kopfmüller et al. (2001) do not doubt the theoretical relevance of the ecosystem resilience concept, rather the authors are interested in the possibilities to operationalize it. The *Resilience Alliance* put ecosystem resilience on their agenda because it provides not only a good tool for the analysis of large-scale processes and dynamics but it is also able to address the resource management issues of the day (e.g. Holling & Meffe 1996, Gunderson & Holling 2002). In fact, some authors identified ecosystem resilience as a guideline for sustainable development (e.g. Ott 2001).

Ecosystem resilience is regarded as a potential tool to solve practical problems, in resource management for instance. This encourages that resource managers as well as sustainability scholars question whether there are *abilities to operationalize* the ecosystem resilience concept in order to decide whether and in how far a given regime is ecosystem resilient to given disturbances or not or whether it is prone to shifts to alternative basins of attraction, respectively (Arrow et al. 1995, Kopfmüller et al. 2001, Ott 2001). Carpenter et al. state that "[p]ractitioners have repeatedly asked how resilience, and trends in resilience, can be measured for particular socioecological systems" (Carpenter et al. 2001, 766) which will be the focus of section 4.2.

The task is not only to operationalize and measure ecosystem resilience. Rather, there is the need for an environmental management approach that is able to *manage for ecosystem resilience*, i.e. to increase the ecosystem resilience of regimes that provide desired services or, at least, to avoid reducing this stability property of a given ecosystem. Thus, section 4.3 will be concerned with the possibilities to *implement* the ecosystem resilience concept through measures within environmental management and nature conservation.

4.1 Conceptual Clarifications and Preliminaries

4.1.1 Conceptual Clarifications

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).

Operationalism as a concept was first introduced by P.W. Bridgman in 1927. Bridgman stated 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). Within theory of science operationalism in its original meaning is regarded as out-dated (Edwards 1967, Seiffert and Radnitzky 1989). Therefore, Jax (2002) 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. It is not necessary to examine this possibility in every single case, rather it appears to be decisive whether it can be carried out in principal (Jax 2002).

For some terms it is even possible to abstain from the claim to operationalize them since they provide a focal point for the development of ecological theory. The term “community” *sensu* Elton (1927), for instance, functions more as a perspective on nature, rather than as a reflection of empirical reality (Jax 2002). As we have seen in section 3.4.2 the concept of ecosystem resilience, in one sense, functions similarly more as a paradigm than as a reflection of empirical reality. The objective is more to switch attention to certain ecological insights, such as *Nature Evolving*, alternative stable regimes, or ecological thresholds, for instance. In this sense it is justified to abstain from the claim to operationalize ecosystem resilience.

On the other hand, there has been tremendous effort to operationalize ecosystem resilience (e.g. Carpenter et al. 2001, Carpenter et al. 2002, Walker et al. 2002, Peterson et al. 2003) since it is regarded as an useful concept for the maintenance of natural capital which represents a claim of strong sustainability and our obligations to future and present generations, respectively. Despite the philosophical difficulties, the possibility to operationalize is still of high scientific value (Poser 2001). With respect to stability properties, Grimm et al. point out: "[t]o make stability concepts operational, i.e. to make them applicable for practical purposes, stability properties have to be quantified" (Grimm et al. 1999, 230). Thus, some authors claim to *measure* the amount of ecosystem resilience an ecosystem may have to a given disturbance regime. However, due to principal difficulties, this amount often cannot be measured directly. Rather an in-depth analysis of ecosystem resilience tries to *estimate* the amount of ecosystem resilience indirectly through indicators of slow variables, for instance.

In order to describe the attempts to operationalize the concept of ecosystem resilience I will use the broad meaning of operationalization, i.e. the principal possibility to identify inter-subjectively for a concrete situation whether a term applies to empirical reality or not, hereby following Jax (2002).

What is crucial here is to distinguish the possibility to operationalize ecosystem resilience *sensu* Jax (2002) from attempts to put the insights provided by the concept of ecosystem resilience (e.g. importance of small-scale disturbances, relevance of ecological redundancy and ecological memory, applicability of slow variables) into concrete environmental management measures, i.e. its *implementation*. In my view, possibilities of implementation also reflect the relevance of ecosystem resilience for a Theory of Sustainable Development and, thus, represents a crucial part of this thesis. Thus, section 4.3 will explore the influence of the ecosystem resilience concept on natural resource management practices.

4.1.2 Social-ecological Systems

According to the *Resilience Alliance*, a fundamental error in past environmental policy has been the assumption that the *human system* and the *natural system* are separate and can be treated independently (Folke et al. 2002).

The nature-culture split is seen as arbitrary and artificial (Westley et al. 2002, Berkes, Colding & Folke 2003). Understanding the individual components of nature-society systems provides insufficient understanding about the behaviour of the systems themselves (Clark & Dickson 2003). Berkes and Folke (1998) and, building on them, Berkes, Colding & Folke (2003) consider social systems (including economic systems, institutions[97], and organizations) and ecological systems as fundamentally linked to each other through interactions on various scales. Examinations do not focus merely on environmental or on social change but rather on social-ecological system (SES) change. Ecosystem responses to resource use, and the reciprocal response of people to changes in ecosystems, constitute coupled, dynamic systems showing emergent properties of the whole SES (Berkes, Colding & Folke 2003). Purely ecological policy models, for instance, ignore essential human behaviours such as foresight potential, forward-looking institutions, reflexivity and sense-making (Westley et al. 2002).

As part of *sustainability science*, the new approach of *vulnerability analysis* considers the examination of coupled SESs as essential to understand the fundamental character of interactions between nature and society which, in turn, is needed to meet human needs while preserving the life-support systems (Clark et al. 2000, Kates et al. 2001, Polsky et al. 2003, Turner et al. 2003, cf. section 4.3.3).

The *Resilience Alliance* extends their heuristics of the adaptive cycle and the *panarchy*, respectively (cf. sections 3.2.2.1 and 3.2.2.2), in proposing that it is useful not only for mere *natural* systems but also for social, political, economic and social-ecological systems (Gunderson, Holling & Light 1995, Holling & Gunderson 2002). For instance, Gunderson, Holling & Light state that the “[c]oupled dynamics of nature society, and resource institutions appear to correspond to the four phase model” (Gunderson, Holling & Light 1995, 508), whereby the adaptive cycle provides a heuristic framework for organizing thoughts, especially around the notion of nonlinearities and the discontinuous and adaptive nature of complex systems. In their case study about the Everglades, USA, Gunderson & Walters (2002) attempt to establish the *Panarchy* heuristic to interpret coupled ecological and social systems. Similarly, Allison & Hobbs (2004) consider the whole social-ecological, agricultural region of Western Australia to follow the adaptive renewal cycle. In my view, it is at

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[97] Institutions are described rather general as “the set of norms, rules that people use to organize activities” (Gunderson 2003, 34).

least doubtable that this heuristic is useful for the examination of whole social-ecological systems.

In this holistic perspective, SESs act as strongly coupled, integrated systems (Folke et al. 2002, Gunderson & Holling 2002) and it leads to fundamental errors if sub-systems are treated isolated and used for recommendations for environmental policy and management (Folke et al. 2002). However, scientific concepts are seen as deficient in the description and analysis of such human-in-nature systems. There is no single, universally accepted way of formulating the linkage between social systems and natural systems (Berkes & Folke 1998).

With respect to ecosystem resilience, Berkes & Folke suggest that "maintaining resilience may be important for both resources and social institutions – that the well-being of social and ecological systems is thus closely linked" (Berkes & Folke 1998, 21).

To sum up, the focus on biophysical components of ecosystem resilience is only useful for analytical and scientific objectives. As far as implementation is concerned coupled SESs have to be examined.

4.2 Ecosystem Resilience Analysis

If we want to decide whether a self-defined system is ecosystem resilient to a given disturbance and estimate the amount of ecosystem resilience the system may have we have to specify clearly the ecosystem resilience *of* what *to* what. The result is an operational, measurable concept of ecosystem resilience (Carpenter et al. 2001) which represents the basis for an appropriate *ecosystem resilience analysis*.

The goals of an ecosystem resilience analysis are (1) to prevent a socio-ecological system from moving into undesirable basins of attraction in the face of external stresses and (2) to nurture and preserve the elements that enable the system to renew and reorganize itself following a massive change (Walker et al. 2002).

An important point here is that ecosystem resilience is - unlike sustainability – not something that is desirable as such (Carpenter et al. 2001, Carpenter & Cottingham 2002, Gunderson & Holling 2002, Walker et al. 2002). Resilience can be desirable or undesirable which depends on the regime of the system of concern. For example, a degraded savannah or a polluted lake can be highly resilient[98].

The following sections will outline five essential steps for an appropriate analysis of ecosystem resilience.

4.2.1 Specification of Ecosystem Resilience

Ecosystem resilience is a concept with multiple meanings (cf. section 3.4.3). Using the extended-ecological meaning, Walker et al. (2004) distinguish four crucial aspects of ecosystem resilience three of which correspond to different characteristics of a basin of attraction. (1) *Latitude* (L) is defined as the maximum amount the system can be changed before losing its ability to recover which corresponds to the width of the basin of attraction. (2) *Resistance* (R) is defined as the ease or difficulty

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[98] Aldo Leopold has already recognized that both normal and degraded regimes of ecosystems could be self-sustaining. His concept of *recuperative capacity* is seen to be similar to the modern concept of ecosystem resilience (Carpenter & Cottingham 2002).

of changing the system which is related to the topology of the domain. (3) *Precariousness* (Pr) is defined as the current trajectory of the system, and how close it currently is to a limit or ecological threshold. The fourth aspect of ecosystem 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.

Table 8: Four Aspects of ecosystem resilience: latitude, resistance, precariousness and panarchy
(according to Walker et al. 2004)

Term	**Definition**
Latitude	Maximum amount the system can be changed before losing its ability to recover
Resistance	ease or difficulty of changing the system
Precariousness	current trajectory of the system; how close it currently is to a limit or ecological threshold.
Panarchy	how the three aspects above are influenced by the regimes of the (sub)systems at scales above and below the scale of interest

Ignorance of the distinction of different stability concepts results in confusion about their meaning (Grimm & Wissel 1997). Therefore, for a sound ecosystem resilience analysis step one is to specify clearly which aspect of ecosystem resilience (cf*. Table 8*) is being analyzed.

4.2.2 To-what Part

The term *disturbance* was provisionally defined as “any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment” (White & Pickett 1985, 7) and is used by several authors considering the role of disturbances (Paine 1998, Turner & Dale 1998, Dittmann & Grimm 1999, Colding, Elmqvist & Olsson 2003).

The definition from White & Picket, however, does not take into account that many disturbances are a normal feature in nature, as ecosystems cannot be seen as static entities (Reichholf 1998). Carpenter & Turner point out that “there is no balance of nature; endless change and the ongoing creation of novelty are the rule" (Carpenter & Turner 2001). Especially, in the face of changing global processes and climatic change in particular, a fluctuating environment is more likely.

In this respect, the distinction of *natural small-scale disturbances* and *human large-scale disturbances* is relevant. Holling & Gunderson (2002) and Colding, Elmqvist & Olsson (2003) suggest that cycles of disruption and recovery (the *adaptive cycle* and the *Panarchy*) are the usual state of affairs in ecosystems whereas compounded human disturbances, i.e. perturbations, have more serious implications for long-term alterations of community state and are seen as extraordinary. Natural small-scale disturbances are seen as intrinsic parts of ecosystem dynamics and as engine for the renewal, novelty, and thus, ecosystem resilience of the ecosystem (Holling 1986, Folke, Berkes & Colding 1998, Gunderson & Holling 2002, Bengtsson et al. 2003, Berkes & Folke 2003, cf. section 3.3.3.5). Natural small-scale disturbances are, therefore, an essential part of a sound adaptive management strategy of natural resources (Folke, Berkes & Colding 1998, Colding, Elmqvist & Olsson 2003, cf. section 4.3.2.3).

Moreover, several features of the disturbance regime have to be identified in order to be able to predict its impact. The *severity* of the disturbance, i.e. its effect on the biota (distinguished from *intensity*, i.e. the energy released or force exerted) is seen as decisive for the ecosystem's response and whether it experiences a shift to another basin of attraction or not (Frelich & Reich 1998, Turner & Dale 1998). Romme et al. (1998) distinguish (1) a *scale-independent response*, (2) a *continuous response*, and (3) a *threshold response* which depends on the severity of the disturbance regime of concern.

Additionally, it makes a difference if the disturbance is *chemical* (acid rain), *physical* (fire, logging), or *biological* (pest outbreak). Some plants (e.g. rain forest) are sensitive to physical disturbances, some (e.g. desert plants) are sensitive to chemical stress (Gorke 1996). In this sense, each ecosystem type may be vulnerable to different disturbances. Moreover, disturbances can influence each other (e.g. fire and wind) which makes it hard to predict their impact (Remmert 1992, Gorke 1996).

Thus, to specify the *disturbance regime* is a fundamental part and represents step two of an ecosystem resilience analysis.

4.2.3 Of-what Part

In the milestone paper *Resilience Management in Social-ecological Systems: a Working Hypothesis for a Participatory Approach*, Walker et al. (2002) point to step three of the analysis of ecosystem resilience – the *of-what part* consisting of the specification of a temporal and spatial scale (section 4.2.3.1), an identification of desired ecosystem services (section 4.2.3.2) and underlying slow variables (section 4.2.3.3).

4.2.3.1 Scale

Concluding a dialogue of young scholars about progress and thoughts for the future in ecology, Starzomski et al. (2004) consider the specification of scale as fundamental for ecological theory formulation. Similarly, Grimm & Wissel (1997) identify scale as an essential part of their *ecological checklist*. In general, *scale* can be defined as “a range of spatial and temporal frequencies” (Peterson, Allen & Holling 1998, 11).

The relevance of different stability properties may vary for distinct temporal and spatial scales (Grimm et al. 1999). The specification of scale is also important since ecosystem resilience of a SES in one time period can be gained at the expense of a succeeding period of time or the ecosystem resilience at one spatial extent can be subsidized from a broader scale (Carpenter et al. 2001, Folke et al. 2002).

Thus, for the identification of the ecosystem resilience of *what* the first thing to do is to identify the temporal and spatial scale that is of concern (Carpenter et al. 2001, Walker et al. 2004) which represents step three of the ecosystem resilience analysis.

4.2.3.2 Ecosystem Services

Daily defines ecosystem services as “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life” (Daily 1997, 3) which include productivity, mitigation of floods and drought, detoxification and decomposition of wastes, pollination of crops and natural vegetation, for instance (cf. section 2.3).

Building on this, Ekins (2003) distinguishes between *functions-for* humans, i.e. those environmental functions that provide direct benefits to humanity, i.e. ecosystem services, and *functions-of* natural capital, i.e. the underlying processes and dynamics that ensure the functioning of ecosystem and provision of ecosystem services.

There is a very important point here.

Within regional management the ecosystem resilience of different ecosystem services – of different functions-for humans - can be in conflict (e.g. agricultural and aquatic ecosystem services). As Walker & Abel point out:

> “The notion of general resilience – that is, ecosystems 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).

Therefore, it is essential to decide which ecosystem services are of primary concern within a given area (e.g. ecosystem, landscape, region), i.e. what exactly *should* remain ecosystem resilient with respect to a given disturbance regime.

These sorts of decisions can not be made solely by using descriptive data provided by ecology since natural sciences only tell us what is the case but not what we should do (Honnefelder 1993, Gorke 1996). The concept of ecosystem resilience does not relieve us to reflect about which nature we want, what types of ecosystems we value enough to preserve, and what ecosystem services we need for a life in human dignity. This task is rather taken into account, first, by ethical reflections about our moral attitude and behaviour towards natural entities (*environmental ethics*) and, second, by evaluative judgements about *environmental quality objectives* (Wiegleb 1997), or, in general, by *environmental assessment* procedures or *wildlife conservation evaluation* [‘*naturschutzfachliche Bewertung*’ (Eser &

Potthast 1997)][99]. Within environmental assessments, several criteria are specified in order to evaluate a given ecosystem or wildlife reserve, such as biodiversity, size of reserve, rareness, naturalness or precariousness (Plachter 2000).

There is a bulk of literature about environmental assessment procedures which cannot be outlined here[100]. However, what is of concern here is that a focus on the ecosystem resilience concept shifts the attention from a *species-view of conservation* to a *ecosystem-view of conservation*, which proposes that the best way to preserve species is to ensure the performance of the underlying variables that are responsible for an ecosystem's structure and function, and, hence, its capacity to sustain its species. This ecosystem-view of conservation may lead to value some criteria for environmental assessments that focus on species conservation *per se* lesser than criteria that take care for the preservation of whole ecosystems. It might be interesting to examine the influence of the background theory of ecosystem resilience on environmental assessment procedures.

The most important thing is that the ecosystem resilience concept does not relieve us to reflect on what nature we want to have. Thus, to choose the ecosystem services that are of concern is an important part and step four of the ecosystem resilience analysis. In this sense, an ecosystem resilience analysis depends on the objectives of the given land-use (Ludwig, Walker & Holling 1997, Walker & Abel 2002).

4.2.3.3 Slow Variables

The continued performance of ecosystem services requires underlying dynamics of ecosystem functioning – the functions-of natural capital (Ekins 2003). The functions-of are constituted by the basic processes and cycles in the internal functioning of natural systems, which are responsible for sustaining and maintaining the ecosystem resilience of natural systems (similar Holling et al. 1995).

Following Folke et al. (2002), I regard the focus on underlying variables as crucial for the maintenance of ecosystem resilience and fundamental for the understanding of the ecosystem resilience analysis. Along with the recognition of the

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[99] These decisions are affected by our position with respect to the discourse about weak and strong sustainability (cf. section 2.2.2).

[100] Cf. Plachter (2000) and Eser & Potthast (1997) for an overview.

existence of alternative stable regimes and ecological thresholds and non-linear behaviour of ecosystems, the focus on slow variables is, in my view, the most important contribution of the *Resilience Alliance* in order to achieve sustainability. As Folke et al. point out:

> “[f]ocusing on production of ecosystem goods or valuation of ecosystem services will not lead to sustainable use by itself, because it does not address the dynamic capacity of ecosystems to uphold the supply of these goods and services” (Folke et al. 2002, 10).

We have explored the focus on slow variables in section 3.2.4. These slow variables are seen as crucial as they determine in which basin of attraction the ecosystem stays and performs its function. Carpenter et al. (2001) suggest that we can conceive of the slow variables as defining the underlying structure of the system, while the fast variables reveal the dynamics of this underlying structure. The supply of ecosystem services depends on which configuration the ecosystem is in and on slow variables of this configuration (i.e. regime), respectively (Walker et al. 2002). Seen from this perspective, ecosystem resilience is a measure of the capacity of an ecosystem to maintan services in the face of a fluctuating environment and human perturbations which points to underlying dynamics of ecosystems that ensure the provision of ecosystem services and foster the ability to stay in the same basin of attraction[101]. This perspectives elucidates that ecosystem resilience is a holistic and qualitative concept [similar to *persistence sensu* Grimm & Wissel (1997)].
As Folke, Colding & Berkes (2003) point out:

> The resilience of the ecological stability domain often depends on slowly changing variables such as land-use, nutrient stocks, soil properties, and biomass of long-lived organisms (…). These factors may be predicted, monitored, and modified. By contrast, stochastic events that can trigger threshold effects and shifts in stability domains (such as hurricanes, droughts, or floods) are usually difficult to predict or control. Therefore, building and maintaining resilience of desired ecosystem states may be most pragmatic and effective way to manage ecosystems in the face of increasingly human-driven environmental disturbances across scales from local to global levels (Folke, Colding & Berkes 2003, 378).

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[101] Herby, I consider expressions such as the “ecosystem resilience of the fish stock” to be fundamentally false, since the right expression would be “ecosystem resilience of the ecosystem to provide the fish stock”. In my view, these formulations are not almost similar, rather they reflect different views on nature and distinct management approaches (cf. section 4.3).

These slow and structuring variables are, therefore, where the priority should be placed in investing to renew, maintain, or restore ecosystems (Holling 1995). It corresponds to an *ecosystem view on conservation sensu* Walker (1992).

Ecosystem resilience in its extended-ecological meaning is defined as "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks" (Walker et al. 2004, 2). There is a principal problem to *measure* ecosystem resilience. As Peterson points out:

> "measuring resilience by altering an ecosystem until it reorganizes is inappropriate if ecological reorganization is costly or irreversible. Because ecological reorganization is what most ecological management is attempting to avoid, measuring resilience in this fashion is often impractical, which has led to attempts to estimate resilience" (Peterson 2002, 2).

In this respect, Carpenter et al. (2001) suggest to *estimate* ecosystem resilience by plotting the equilibria of the system on axes of the rapidly changing variable and a more slowly changing variable. The plot then shows upper and lower sets of stable states separated by an unstable set of equilibria – which corresponds to a *bifurcation diagram* (cf. *Figure 9*).

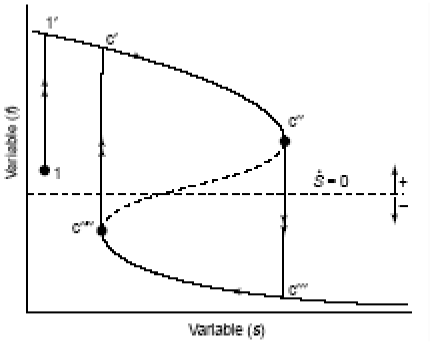

Figure 9: Bifurcation diagram; X-axis corresponds to a slow variable (s), Y-axis reflects a fast variable. The upper and lower solid line represent two alternative basins of attraction. Ecosystem resilience can be estimated as the size of the upper basin from c' to c'' or the distance from an imaginary current state to the critical amount of the slow variable (c'')
(from Scheffer & Carpenter 2003)

Similarly, Peterson, Carpenter & Brock (2003) propose difference equations of a slow variable as model to operationalize and estimate ecosystem resilience. Hereby (as illustrated in *Figure 9*), ecosystem resilience is tracked as the size of the attractor of the desired regime (e.g. c’ to c’’) or the distance between the current state of the slow variable and the critical state (the distance between an amount of the slow variable between c’ and c’’ and c’’), which has to be estimated empirically (Peterson, Carpenter & Brock 2003) or theoretically (Wissel 1984, Ludwig, Walker & Holling 1997, 2002). In practice, ecosystem resilience could be tracked by monitoring the slowly changing ecological variables that control the desired regime (Carpenter et al. 2001). To my view, it remains unclear, however, how the attractor size or the precariousness should be estimated empirically.

The metaphor of a stability landscape gets relevant here which includes the identification of ecological thresholds that separate alternative basins of attraction as well as the specification of the *precariousness* of regimes, i.e. how close they are to a certain ecological threshold. In this respect, Wissel’s (1984) and Ludwig, Walker & Holling’s (2002) theoretical analyses may help to predict the position of ecological thresholds with respect to environmental variables.

Referring to this point, Walker & Meyers (2004) provide a developing database including five classes and eleven categories of ecological thresholds. Data from areas that have already undergone a regime shift can be compared with data from similar environments that have not. Whether an ecological threshold can be identified before it has been crossed depends on data extrapolation from similar closely related systems. As Walker & Meyers point out:

> “So far, we have found no published examples where a new kind of threshold has been predicted before it has been experienced. How to do this is an obvious question for any future research agenda on thresholds. Is a threshold an emergent property of some underlying set of attributes of a system? Is there a relationship, for example, between the network topology of a system and the likelihood of a threshold? Which research avenues will allow us to best approach this question? Research on a typology of thresholds is a priority topic in the emerging area of “sustainability science”, and it requires a rich database of empirical data” (Walker & Meyers 2004, 12).

Even if the position of an ecological threshold cannot be predicted at the current knowledge (Muradian 2001) the existence of alternative basins of attraction can be and has been predicted (Foley et al. 2003). By examining the underlying state of environmental systems and their degree of nonlinearity, scientists may be able to predict which regions of the world are susceptible to regime shifts and which are not (Scheffer & Carpenter 2003). This clearly represent a strong tool for environmental management.

The identification of indicators represents a vital option for environmental management. *Indicators* can be defined rather general as auxiliaries chosen to represent complex circumstances that are often difficult to measure directly. Kopfmüller et al. (2001) suggest some standards that indicators ought to fulfil. Among other criteria, they should represent the ecological situation adequately, be reproducible, be sensitive to changes over time, and the data should be available regularly and in an appropriate way.

There are several methods to organize sustainability indicators. Hart (1999) lists *categories* (e.g. economic, ecological, social, institutional, cultural), *goal-indicator matrixes* and *pressure-state-response models* as examples.

Using these methods, various organizations and governments have proposed their indicator set for sustainability, such as the Commission on Sustainable Development, the Organization for Economic Co-operation and Development, the US Presidents Council on Sustainable Development, the International Institute for Sustainable Development, the World Bank or the World Resource Institute (Kopfmüller et al. 2001).

Several countries have launched *sustainability strategies* which include the identification of environmental. With respect to slow controlling variables, indicators clearly should reflect the internal dynamics of ecosystems and their ecosystem resilience, respectively. However, Deutsch, Folke & Skanberg (2003) suggest that it is difficult to judge the extent to which indicators sets capture internal dynamics, or whether they have been developed for such a purpose. As we have seen ecosystem resilience focuses on the variables that underlie the capacity of natural or social-ecological systems to provide ecosystem services, and obviously indicators for ecosystem resilience should reflect this.

In this respect, the *Resilience Alliance* has launched some case studies that have proposed indicators that represent the slow variables that are conceived as driving and controlling ecosystem behaviour. Carpenter & Cottingham (2002), suggest several potential indicators of a lake’s capacity to remain in the clear-water state, i.e. the ecosystem resilience of the desired clear-water regime to disturbances such as phosphorus pollution, airborne pollution and species invasions.

> “The livestock density in the watershed is a correlate of phosphorus imports (…). Wetland area per unit lake is an index of the landscape’s capacity to hold water and export humic substances (…). The proportion of the riparian zone occupied by forest and grassland indicates the potential attenuation of nutrient inputs (…). Lake color relates to humic content (…). Slow-to-moderate piscivore growth rates are associated with strong piscivore control of planktivores (…). Grazer body size correlates with the capacity to suppress algal growth (…). Partial pressure of carbon dioxide in surface waters may be a decisive indicator of ecosystem metabolism (…). Hypolimnetic oxygen depletion is a symptom of eutrophication and a driver of phosphorus recycling from sediments (…)” (Carpenter & Cottingham 2002, 61).

In this case, indicators for slow variables presume a detailed understanding of the ecosystem dynamics and their relation to the disturbance regime.

Jansson & Jansson (2002) identify phosphorus storages and the nitrogen/phosphorus ratio in the sediments, as well as the pool of organic matter as slow variables for the Baltic. These variables can be estimated and function as indicators for a given ecosystem regime that exhibits desired or undesired ecosystem services.

For rangelands in Australia, Walker (2002) considers abundance of woody plants, grasses, perennial grasses and annual grasses, and amount of herbivores as essential variables since the desired regime is defined by its productive potential for pastoralists and, thus, the proportional composition of the plant biomass in terms of woody plants, perennial grasses, unpalatable perennial grasses and annual grasses. Again, these variables can be estimated. To my knowledge there have been attempts to indicate these variables by the abundance of emus, a project termed correspondingly the EMU project. In this case, it is arguable, in my view, whether the slow variables *really* reflect the driving and controlling variables or rather faster variables that represent ecosystem services. In fact, Walker (2002) includes rainfall, fire and grazing and browsing as *drivers* of the rangelands.

Note that ecosystem resilience indicators appropriate for the current regime of an ecosystem may become irrelevant as ecological structures and social expectations shift and evolve (Deutsch, Folke & Skanberg 2003).

The identification and specification of slow variables and indicators that represent them is, in my view, one of the most important tasks within further investigations of ecosystem resilience. Such an estimation would apparently provide useful data for an adaptive, environmental management approach. It might be difficult, however, to find the controlling underlying variables for each ecosystem type. It seems to be relatively easy for aquatic types since nutrient storages play a key role in these ecosystems. According to Volker Grimm working at the UFZ ('*Umweltforschungszentrum*') Leipzig-Halle, this leads to the debate about modelling within theoretical ecology. Grimm (1999) argues for the relevance of bottom-up or *individual-based models* but insists on using top-down or *state variable models* in a complementary way in order to achieve an appropriate integrated view of ecosystem dynamics.

The identification of critical natural capital (CNC) is related to ecosystem services on the one hand, and to ecosystem resilience and underlying controlling variables on the other (Deutsch, Folke & Skanberg 2003). This often imprecise concept of CNC would get contours if it was related to slow and controlling variables. There are some attempts to specify CNC for regions and countries, such as Ekins & Simon (2003) for the UK. In my view, if related to ecosystem resilience and underlying slow variables, the estimation of CNC for as many regions and countries as possible would be of high value.

To sum up, the identification of slow variables represent the fifth and last step of an appropriate ecosystem resilience analysis.

4.2.4 A Most Appropriate Scale?

Specification of the spatial and temporal scale is crucial for an appropriate ecosystem resilience assessment. Ecosystem dynamics, however, occur at every scale, from the very small-scale via the biome scale to the world scale. Presuming the *Panarchy* heuristic, each level of the hierarchy is influenced by interactions from above (e.g. Remember) and from below (e.g. Revolt) (Holling et al. 2002). With respect to ecosystem resilience, this is reflected by pointing to the aspect *panarchy*, which is related to how the three other aspects of ecosystem resilience (*latitude*, *resistance*, *precariousness*) are influenced by the regimes of the (sub)systems at scales above and below the scale of interest. In particular, considering global change and climate change issues, it becomes clear that processes on the regional or even continental level can influence ecosystem dynamics on the landscape and patch scale over great distance. For each specification of scale it is, thus, obvious, that this scale of interest is influenced by other scales, either temporal or spatial.

According to Levin (1992), there is no single “correct” scale on which to describe populations or ecosystems since different processes are likely to be important on different scales. The investigation is contingent upon the window through which the system is viewed[102]. This point is crucial. Since ecosystem dynamics occur at each level of the hierarchy and processes are likely to be important on different scales there cannot be something as the most appropriate scale. Referring to a dialogue of young scholars, Starzomski et al. propose that

> „it seems that the solution to the multiscale dilemma may lie in our ability to assess and incorporate information across various scales rather than to collaps information into a single „most appropriate“ scale“ (Starzomski et al. 2004, 6).

On the other hand, there are practical objectives that should guide the application of the ecosystem resilience concept. With respect to a theory of strong sustainability, objectives such as the maintenance of natural capital (e.g. provision of the productive potential of a agricultural region, maintenance of pastoral land, preservation of fish resources in a lake region) point to the relevance of larger, in particular regional scales of investigation. For instance, assessments of the agricultural region in Western Australia (Allison & Hobbs 2004), lakes (Carpenter & Cottingham 2002), the Baltic Sea (Jansson & Jansson 2002), coral reefs (McClanahan et al. 2002), the Everglades, USA (Gunderson & Walters 2002), grazed rangelands (Walker 2002) or tropical wet and dry forests in Puerto Rico (Lugo et al. 2002) all point to the importance of larger scales of investigation.

Additionally, within ecosystem-functioning debate, various authors question the expressiveness of small-scale studies for large-scale issues. As Holling & Meffe point out:

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[102] Levin (1992) proposes, however, that there might be scaling laws that allow one to make comparisons among studies carried out on different scales.

> “much of the present ecological theory uses (…) small-scale quadrat experiments in nature (Tilman and Downing 1994) in which long-term, large-scale successional or episodic transformations are not of concern. (…) But these traditional concepts and techniques make the world appear more simple, tractable, and manageable than it really is. They carry an implicit assumption that there is global stability” (Holling & Meffe 1996, 333).

Classical equilibrium approaches are seen to be inadequate to understand stability properties such as ecosystem resilience and shifts between alternative stable regimes at larger scales (Loreau et al. 2001). Similarly, Bengtsson et al. (2002) state that it is not at all clear if small-scale approaches can be used to inform public and policymakers about the large-scale consequences of biodiversity loss. When the scale of investigation changes properties of communities and ecosystems do not just change in any coherent fashion. Bengtsson et al. (2002) argue for investigations on different scales and in particular on large-scale levels, comparisons of lands under different management practices and ecosystem models in order to be able to answer whether diversity really is important for the ecosystem resilience and maintenance of ecosystem services at larger scales in space and time.

Thus, a focus on larger spatial and temporal scales seems to be appropriate as far as SD objectives are concerned. Still the investigation is contingent upon the window through which the system is viewed (Levin 1992), but, in my opinion, this window is the purpose of the scientist to provide knowledge with respect to SD. The goal of investigation can be formulated as “creating and applying knowledge in support of decision making for sustainable development” (Clark & Dickson 2003, 8059). Note that the specification of larger spatial and temporal scales are normative rather than ecologically descriptive. It reflects, above all, the human desire to live in a secure and “stable” world (Gorke 1996).

In view of this insight, still ecosystem dynamics occur at every scale and their importance depends on the level of scale. Folke, Berkes & Colding suggest that

> “[e]nvironmental and renewable resource issues tend to be neither small scale or large scale but *cross-scale* in both space and time. It follows, therefore, that the problems have to be tackled simultaneously at several levels” (Folke, Berkes & Colding 1998, 432).

In this respect, Clark et al. (2000) propose that no inherently superior scale of *vulnerability analysis* has emerged and that most, useful *vulnerability* *assessments* [103] will need to address multiple stresses that interact across a variety of scales. Turner et al. (2003) suggest that vulnerability analysis may be undertaken at any spatial or temporal scale suitable for the problem in question.

From the systemic-ecological perspective, ecosystem resilience corresponds to the capacity of an ecosystem to provide services in the face of a fluctuating environment and human perturbations (cf. section 3.4.3). Slow variables and their interactions to fast ones seem to “drive” this capacity. Thus, in my view, the scale of an ecosystem resilience analysis that provides knowledge with respect to SD, must be at least at a level that takes into account the slow variables that drive ecosystem behaviour. Certainly, levels below and above have to be taken into consideration *if* they influence the behaviour of these slow variables.

To sum up, for an ecosystem resilience analysis aiming to provide knowledge with respect to SD objectives there is something like a most appropriate scale of investigation although knowing that dynamics can be relevant on each scale of the ecosystem hierarchy.

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[103] Cf. section 4.3.3.

4.3 Managing for Ecosystem Resilience

The previous sections have explored the abilities to estimate ecosystem resilience empirically as well as theoretically (cf. section 4.2). Subsequently, after showing the pathology of current resource management approaches (section 4.3.1), some measures to manage for ecosystem resilience are proposed (section 4.3.2). Finally, section 4.3.3 will examine some attempts to model entire social-ecological systems which is seen as essential for a Sustainable Development.

4.3.1 Pathology of Resource Management

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 1995, 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, Colding & Folke 2003). 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 & 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 “inefficient” natural system into one that produces products in a predictable and economically efficient way (Holling & Meffe 1996).

> “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).

The result is a gradual loss of ecosystem 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 ecosystem 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 ecosystem resilience in ecosystems. Gunderson et al. (2002b) 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. 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 functions such as productivity (Levin et al. 1998, Folke et al. 2002).

Holling et al. (2002b) list the collapse of some fisheries, the vulnerability to drought of semiarid rangelands, increased vulnerability through flood control measures and irrigation developments as examples of pathologies of regional resource management. Investigating the entire human impact on coastal ecosystems, Jackson et al. (2001) propose destruction and collapse of various ecosystem types including kelp forests, coral reefs, tropical and subtropical seagrass beds, estuaries and offshore benthic communities. The historical human impact on and recent exploitation of coastal ecosystems lead to a decrease in ecosystem resilience to human use. “Early changes increased the sensitivity of coastal marine ecosystems to subsequent disturbance and thus preconditioned the collapse we are witnessing” (Jackson et al. 2001, 635).

Holling & 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).

The pathology has the following features (Holling 2003): (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 perverse[104] subsidies. (4) The relevant ecosystems gradually lose ecosystem 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).

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). In the following I will describe the attempts to *manage for ecosystem resilience*, an approach dubbed adaptive management.

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[104] Perverse in the sense of that it threatens the functioning of the system in the long-term.

4.3.2 Adaptive Management

Systems of people and nature co-evolve in an “adaptive dance” (Walters 1986). Recently, remarkable advances in the understanding of ecosystems have occurred (Holling 1995). However, pathologies of resource management show that understanding is still partial.

In order to take into consideration the unexpected behaviours of complex systems, Holling (1978) introduced the concept of *adaptive environmental assessment and management*. Change in ecosystem dynamics is considered to be inevitable and inherently unpredictable. The model of complex adaptive system - the *Panarchy* - put forward by the *Resilience Alliance* (cf. section 3.2.2), includes the notion that these system are inherently unpredictable (Gunderson, Holling & Light 1995). More particular, referring to the adaptive cycle, there are phases (r and K) when the dynamics are rather predictable, and there are phases (α and Ω) when they are highly unpredictable (Gunderson & Holling 2002). Gunderson suggests that “[n]onlinear interactions among multiple variables, scale invariant processes, emergent properties from self-organization and other factors all contribute to unpredictability” (Gunderson 2003, 33).

Surprises[105] are *inevitable*, and thus policies must always be adaptive. Therefore, within adaptive management, the existence of uncertainty and surprise are an accepted part of development, and management actions evolve to cope with their effects by spreading risks through diversification of both resource use patterns and alternative activities.

Facing this unpredictability, *adaptive environmental management* tries to preserve or increase the ecosystem resilience of ecosystems since this is considered as a precondition for the capacity of the SES to respond to and shape change (Peterson 2002), i.e. to increase its *adaptive capacity* (Folke, Colding & Berkes 2003). The objective is to examine “ways of *building resilience* to enhance the capacity to deal with change and surprise” (Berkes, Colding & Folke 2003, 22), indeed, to “actively strengthen and enhance the capacity of the biosphere, from local to global levels, to support and sustain social and economic development” (Folke 2003, 379).

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[105] “Surprise denotes the condition when perceived reality departs *qualitatively* from expectation” (Berkes & Folke 1998, 6).

This includes increasing the ecosystem resilience of desired ecological regimes while decreasing the ecosystem resilience of unwanted regimes since ecosystem resilience is not something that is desirable as such (Carpenter et al. 2001, Carpenter & Cottingham 2002, Gunderson & Holling 2002, Walker et al. 2002).

There is another fundamental point here.

Management measures can be treated as “experiments” from which managers can learn. Organizations and institutions can “learn” as individuals do, and hence adaptive management is based on social and institutional learning emphasizing the importance of feedbacks from the environment in shaping policy. As Berkes & Folke point out:

> “Key factor in successful adaptation may be the presence of appropriate feedback mechanisms which enable consequences of earlier decisions to influence the next set of decisions which make adaptations possible” (Berkes & Folke 1998, 19).

Key bridges for the feasibility of an adaptive management approach are provided by views that include a systems perspective, interdisciplinary, nonlinearity, and cross-scale views (Gunderson, Holling & Light 1995) – hence more or less the view that takes into consideration the *background theory of the ecosystem resilience concept* (cf. section 3.2).

Berkes, Folke & Colding (1998) and Berkes, Colding & Folke (2003) identify three measures of an adaptive environmental management approach for building ecosystem resilience into SESs that will be explored in the following sections.

4.3.2.1 Ecological Knowledge

Sustainable use of the capacity of ecosystems to generate services is unlikely without improved understanding of ecosystem dynamics. However, only a fraction of ecosystems have been subject of careful examinations within the framework of conventional (Western) resource management and science.

However, groups of resource users, such as indigenous people who live off wildlife, fish, and forests, also create knowledge from their own observations and ecological understanding, based on the accumulation of generations of trial-and-error experience (Berkes & Folke 2002). In this respect, *traditional ecological knowledge* 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 (including humans) with one another and with their environment” (Berkes 1999, 8) and represents a fundamental basis of *social-ecological linkages* between the natural system and the human system (Folke, Berkes & Colding 1998).

Proponents of adaptive resource management do not claim that traditional ecological knowledge is to replace conventional Western science and management. Rather, it may be thought to be complementary to scientific knowledge. Scientific knowledge and traditional knowledge are seen to focus on different aspects of management problems highlighting the need for conceptual pluralism in resource and ecosystem management (Berkes & Folke 2002).

There is an important point here. Berkes & Folke suggest that

> “traditional practices have certain similarities and parallels to the theory of complex systems, with emphasis on nonlinear relationships, threshold effects, multiple equilibria, the existence of several stability domains, cross-scale linkages in time and space, disturbance, and surprise” (Berkes & Folke 2002, 124).

Moreover, these practices appear to focus on the back-loop, i.e. the release (Ω) and reorganization (α) phase of the adaptive cycle. This point is crucial. If traditional practices are similar to insights of modern ecological theory, this traditional knowledge can be used for a sound environmental management.

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. though 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.

Thus, within adaptive management, *local ecological knowledge* is seen as central (Folke, Berkes & Colding 1998, Folke, Colding & Berkes 2003). The following chapters will point out some examples of these traditional practices.

4.3.2.2 Focus on Biodiversity

There are various ecosystem resilience mechanisms that are closely connected to biodiversity (cf. section 3.3.3).

Ecosystem functioning is not related to species richness per se, rather the existence of functional groups and their representation is seen as critical. Various authors propose that *functional diversity* increases the performance of the community as a whole, while *response diversity* enables the community to keep performing in the same complementary way in the face of stresses and disturbances (e.g. Elmqvist et al. 2003). In this respect, Mooney et al. (1996) conclude that sensitive ecological systems *all* have low representation of key functional types, i.e. little functional redundancy within functional groups. The conservation focus shifts to those functional groups that have low representation of species, termed *keystone functional groups* and the corresponding species *keystone species* (Walker 1995, cf. section 3.3.3.1). For example, acidification of lakes eliminates a key group of bacteria, thereby blocking the cycle of nitrogen, one of the most important nutrients. Ecosystem resilience to acidification would be higher if several species were capable of performing this function (Carpenter & Cottingham 2002).

Response diversity is likely lead to an increased functional compensation within communities with respect to a given ecosystem function in the face of disturbance regimes. It might be difficult, however, to predict which species will account for functional compensation. To identify these redundant and compensating species would be of high value for an ecosystem management. For coastal ecosystems, Jackson et al. state that

> “[t]he importance of biodiversity in the form of ecological redundancy is clearly apparent for the delay in the collapse of kelp forests in southern California compared with Alaska after the extirpation of sea otters. Sheephead fish, spiny lobsters, and abalone in the more diverse Californian kelp forests kept sea urchin populations in check until these predators and competitors of sea urchins had also been effectively eliminated (…).

> Similarly, the sea urchin Diadema kept macroalgae in check long after the extreme overfishing of herbivorous fishes on Caribbean coral reefs” (Jackson et al. 2001, 636).

Additionally, the recovery rate of corals after disturbance in the low-diversity Caribbean has been slower than recovery in the high-diversity Indo-Pacific. MacClanahan et al. (2002) suggest that this might be weak evidence that species diversity increases the capacity of reefs to tolerate and recover from disturbance[106].

Another ecosystem resilience mechanism, namely *ecological memory* on all levels (genetic, species, landscape) consisting of biotic legacies (Franklin & MacMahon 2000), mobile links (Lundberg & Moberg 2003) and support areas (Elmqvist et al. 2003) buffers against large-scale disturbances and fosters reorganization processes. Folke et al. propose that “[b]iodiversity plays an important role in the reorganization and renewal process following disturbance (Folke, Colding & Berkes 2003, 362).

In Polynesian agriculture, for instance, the maintenance of a high degree of species diversity can be seen as part of a deliberate strategy deployed by local farmers to reduce their vulnerability, i.e. to increase the ecosystem resilience, of productive advanced polyculture, to tropical cyclones. In this case, a diverse set of crop species and cultivars reduces the risk of a total loss in food supply (Colding, Elmqvist & Olsson 2003).

From a different perspective, Bengtsson et al. (2003) point to the relevance of ecological memory, biodiversity and, thus, spatial resilience for modern intensively managed landscapes. Disappearance of natural mosaics through fragmentation of the landscape results in the loss of species that do not have the capacity for long-range dispersal between suitable patches. The lower diversity and more synchronized ecosystem dynamics in managed and fragmented landscapes are likely to lead to a lower capacity to recover naturally after disturbance, such as pest attacks or extreme climatic events. Reserves that are created for preservation of species or habitats will fail in its objective if the areas surrounding the reserve do not contain sufficient spatial resilience in the form of ecological memory. This *resilience gap*, in this case a decrease in landscape resilience leads to increased vulnerability and uncertainty concerning the delivery of essential ecosystem services. Therefore,

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[106] It is unclear and interesting whether this capacity corresponds to engineering resilience or to the extended-ecological meaning of ecosystem resilience. This leads to the question of the relation of these two stability properties (cf. section 3.4.2).

Bengtsson et al. (2003) argue for the relevance of dynamic *insurance reserves* that secure (ecosystem) resilience in managed landscapes and are both temporally and spatially flexible. *Ecological fallows* would be areas set aside for natural, or low-intensity managed succession immediately after a disturbance event. *Ephemeral reserves* would be aimed at preserving species in the early succession or exploitation phases that otherwise may be threatened by large-scale intensive management. *Mid-succession reserves*, in contrast to static reserves, can be allowed to cease or change in parts of the landscape, if there are other nearby areas providing persistence. Imperative to all these types of reserves is that they must be part of management at the landscape level, i.e. on a spatial scale of 10-100 km^2 or more. These reflections point to the relevance of landscape ecology and meta-population theory with respect to ecosystem resilience. In general, the relation of the ecosystem resilience concept and nature conservation (e.g. Plachter 1992) and landscape ecology (e.g. Leser 1997), respectively, would be of high interest but cannot be pointed out in detail here.

Considering these reflections, adaptive management ought to nurture biodiversity in order to ensure the performance of ecosystem functioning and to buffer against disturbances. It is considered to be a prerequisite for the ecosystem resilience of natural systems (Deutsch, Folke & Skanberg 2003).

4.3.2.3 Allowing Small-scale Disturbances

One of the key features of an adaptive management approach to strengthen ecosystem resilience is to consider small-scale, natural disturbances as *intrinsic* part in the internal dynamics of ecosystems (Holling 1986, Folke, Berkes & Colding 1998, Gunderson & Holling 2002, Bengtsson et al. 2003, Berkes & Folke 2003, Colding, Elmqvist & Olsson 2003) which I dubbed the *renewal aspect of ecosystem resilience* (cf. section 3.3.3.5). In order to ensure the ecosystem resilience of the structural and functional performance of ecosystems on a larger scale (e.g. landscape, 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 basins of attraction. The aim of adaptive management, therefore, is to prevent the build-up of large-scale crisis (Folke, Berkes & Colding 1998). This type of management is termed *backloop management* since it indirectly considers the release-reorganization phases of the adaptive cycle (Colding, Elmqvist & Olsson 2003).

> "In 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, Elmqvist & Olsson 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). Rural people of Bangladesh, including char-dwellers, consider annual floods as normal and have developed ecological and social strategies to live with this disturbance (Colding, Elmqvist & Olsson 2003).

In general, local resource users may actively create small-scale disturbances in the landscape. Many traditional societies nurture sources of ecosystem renewal by creating small-scale disturbances (Folke, Berkes & Colding 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 & Folke 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, Elmqvist & Olsson 2003)[107]. By mimicking fine-scale natural disturbances, these practices help avoid

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[107] Last year, I visited an acquaintance whose parents owned a farm in the "forest land" of Western Australia close to Bunbary. At dinner, I talked to her father who complained about the conservationists. They prohibited to use fire in order to preserve some forest species they valued in this region. The farmer held that the build-up of fuel in these forests made the region prone to large-scale crisis as it was likely to lead to immense fires in the long term. This example, in my view, illustrates that measures of a sound adaptive management of natural resources challenges not only traditional management practices but also conservationists' interests. The preservation of an "artificial" state of nature is seen as sceptical by the *Resilience Alliance*.

the accumulation of disturbance that move across scales further up in the *Panarchy* (Berkes & Folke 2002).

To illustrate the importance of small-scale disturbances for a sound management of natural resources, Berkes, Colding & Folke provide a short formula: “Resource and environmental management that suppresses disturbance and diversity will be unsustainable” (Berkes, Colding & Folke 2003, 23).

4.3.2.4 Adaptive Capacity

In reference to the notion of ecosystem dynamics *sensu Resilience Alliance*, the behaviour of ecological systems and SESs is inherently unpredictable. Uncertainty, unpredictability and surprise is seen as inevitable (Gunderson et al. 1995, Carpenter et al. 1999, Gunderson & Holling 2002, Gunderson 2003). As Carpenter, Brock & Ludwig (2002) point out:

> “[C]ollapse can occur even if the ecosystem dynamics are perfectly known and management has perfect control of the human actors. The economic optima (…) draw the system into a region where resilience is small relative to noise, so a disturbance eventually shifts the system to an undesirable domain. Such a disturbance will occur, the only question is when (…). One way to prevent such a disturbance would be to maintain the system in a region where resilience is large, but such a policy would be inconsistent with the economic optimization criterion” (Carpenter, Brock & Ludwig 2002, 187).

If one takes into account the inevitable features of uncertainty and surprise, the relevance of *learning* gets obvious. How people choose to deal with uncertainty and surprise appears to either increase or decrease the ecosystem resilience of an ecosystem (Gunderson 2003).

Moreover, scenario planning represents a vital tool for developing more adaptive conservation policies (Peterson, Cumming & Carpenter 2003). In this context, *scenario* can be understood as a structural account of a plausible future. Scenario planning consists of using a few contrasting scenarios to explore the uncertainty surrounding the future consequences of a decision.

In this respect, the ecosystem resilience concept is often extended to social or social-ecological 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). *Social memory*, hereby, refers to the long-term communal understanding of the dynamics of environmental change and the transmission of the pertinent experience.

Similarly, Walker et al. (2004) suggest the term *adaptability* to describe the capacity of actors in a system to influence or manage ecosystem resilience, which is seen to be mainly a function of the social component of an SES. Corresponding to the four aspects of ecosystem resilience *sensu* Walker et al. 2004 (cf. section 3.2.4), measures of adaptability represents the ability to control the trajectory of the system (change precariousness), change the topology of the stability landscape (latitude and resistance), or change the processes in response to dynamics at other scales (panarchy response).

Building on the three fundamental measures for adaptive management, people can decide to increase ecosystem resilience by building up or allow ecosystem resilience mechanisms, respectively.

Moreover, the identification of slow variables is one of the possibilities to estimate ecosystem resilience and is seen as crucial for a sound environmental management approach (cf. section 4.2.3.3). Carpenter, Brock & Ludwig (2002) suggest that

> “[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, Brock & Ludwig 2002, 193).

In order to estimate the amount of ecosystem resilience some authors use simple models of two variables and bifurcation curves. However, according to Peterson (2002), ecological managers often do not have simple models of their systems and, thus, methods of estimating ecosystem resilience that do not depend on the construction of system models would greatly facilitate the application of ecosystem resilience-based management. Referring to this problem, Peterson (2002) suggests two methods termed *probabilistic resilience* and *cross-scale edge*.

The probabilistic approach, first, identifies a set of alternative stable regimes that could potentially exist across a given landscape. Underlying dynamics are treated as black box. Focus of interest is on transition among discrete regimes. The behaviour of a discrete regime can be assessed in terms of the probabilities of leaving that regime and remaining in that regime. The probability that a regime will persist [*sensu* Grimm & Wissel (1997)] is treated as a measure of its ecosystem resilience. Observations or simulated regime transitions are used to estimate ecosystem resilience. Hereby, *ecotones* can be thought of as the edge that separates regions that are dominated by two alternative stable regimes, and represent areas where small changes can cause a shift from one regime to the other. Then, using *percolation theory*, and *cross-scale edge* the two identified regimes of the given landscape are mapped for a particular spatial scale of concern. For estimation and comparison of the ecosystem resilience several management strategies can be modelled and compared, and the best management strategy can be identified. Peterson (2002) concludes that cross-scale edge represents a simple measure and tool for the prediction of ecosystem resilience. It represents, thus, another possibility to operationalize ecosystem resilience.

All these abilities are summarized in the concept of *adaptive capacity*, which is defined as the capacity of a SES to adapt to slower dynamics (Gunderson 2003) or, rather general, as the capacity of an SES to respond to and shape change (Peterson 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).

However, even if a resource management system is dynamic in its response to ecosystem change and surprise and builds social-ecological resilience, it may be fragile and vulnerable to external social and economic drivers (Folke, Colding & Berkes 2003).

Referring to this, Walker et al. suggest the term *transformability* for "the capacity to create a new stability landscape (…), to create untried beginnings from which to evolve a new way of living when existing ecological, economic, or social structures become untenable" (Walker et al. 2004).

This is what occurred in rangelands in south-eastern Zimbabwe, for instance. After many decades of cattle ranching, the rangeland ecosystem has changed undesirably for livestock and terms of trade have declined. A severe drought in the early 1980s triggered a transformation from many individual cattle ranches to a few wildlife “conservancies” with all livestock and fences removed and managed collectively for tourism and hunting (Walker et al. 2004). In this sense, slow and fast variables of the stability landscape also depend on the objective of the observer and land-user.

4.3.3 Models of Social-ecological Systems

Ecosystems change in response to the stress imposed by human use, and human societies adjust their behaviour affecting ecosystems in response to perceived changes in these systems. A thorough understanding of this feedback would be the ultimate scientific foundation for designing strategies to achieve sustainable society-nature interaction (Scheffer et al. 2002). One key technique to gain understanding is to abstract essential properties in a model that mimics behaviour over time for a variety of conditions. Note that no model – mental or mathematical – is “true”. Rather, degrees of credibility and usefulness for given objectives can be defined (Holling 1978, Leser 1997). The following sections explore some attempts of the *Resilience Alliance* and the *Forum of Science and Technology for Sustainability*, respectively, to model the behaviour of large-scale SESs[108].

According to Carpenter, Brock & Hanson (1999), the *minimal elements of models* that are sufficient representations of ecosystem management include ecosystem dynamics that involve nonlinear interactions of variables with distinctly different turnover rates (fast and slow variables), and a social arena in which agents assess the status and potential future state of the ecosystem, compare possible actions, and choose policies that subsequently affect the ecosystem and the scope of future choices.

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[108] There is a rich literature about the relation of ecosystem resilience and economics. Hereby, ecological economist attempt to analyse nonlinear dynamic systems from an economic perspective (Common & Perrings 1992, Perrings & Walker 1997, Perrings 1998, Brock, Mäler & Perrings 2002). It appears to be an interesting field having high relevance for a Theory of Sustainable Development. Unfortunately, it is beyond the scope of this thesis to point out their main ideas.

For lakes, Carpenter, Brock & Hanson (1999) distinguish (1) the market manager model, (2) the governing board model and (3) the land manager model. Only model (3) allows to manipulate the ecosystem resilience of the lake by managing the slowest variable, i.e. soil phosphorus. Thus, the game player has a mechanism for manipulating the "stability" of the ecosystem through the slow variable which is pictured in *Figure 10*.

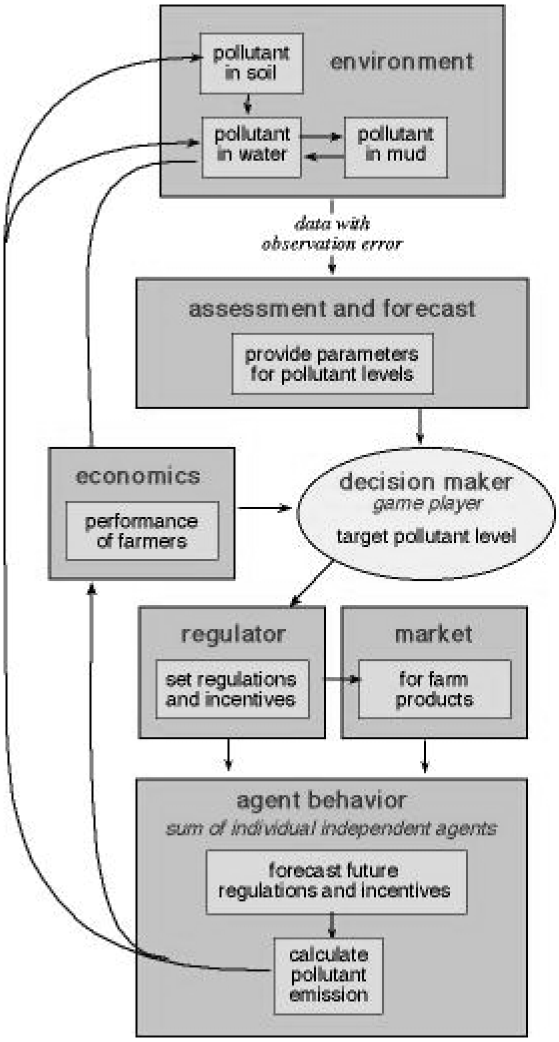

Figure 10: Land manager model for social-ecological systems
(from Carpenter, Brock & Hanson 1999)

As illustrated in *Figure 11*, the dynamics of the lake-SES can be conceived as an adaptive cycle showing changes in net economic yield (in millions of U.S.$) from farms plus ecosystem services, proportion of farms using phosphorus-intensive practices, and width of the attractor for the desirable regime of the lake.

Even a manager with perfect knowledge and perfect control of the social system cannot avoid outbreaks of pollution under certain circumstances. Some disturbances will knock the system out of the low-P attractor into the high-P attractor, creating an outbreak (Ω phase). An adaptive manager would move toward sustainability by shrinking the scope of the cycles and explore the stability domain of soil phosphorus.

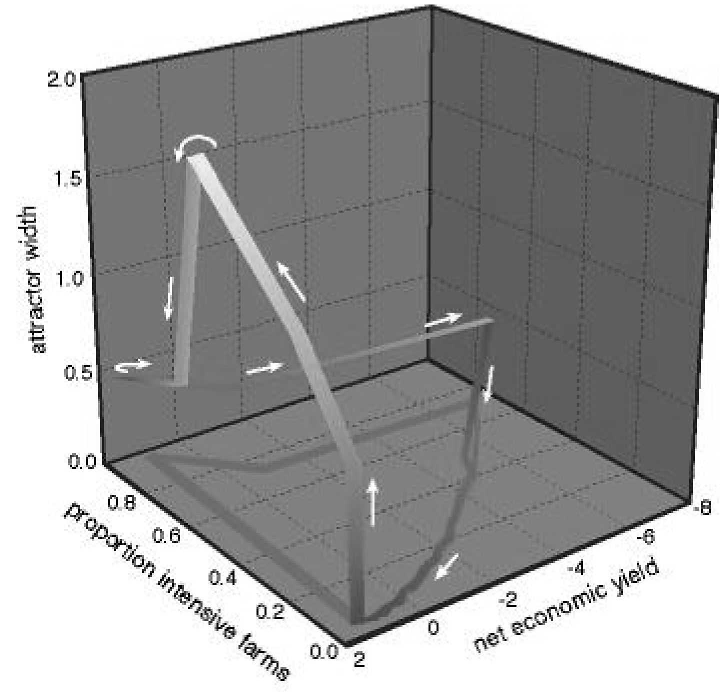

Figure 11: Adaptive cycle showing dynamics of a lake-SES with three characteristics: net economic yield, proportion of phosphorus-intensive farms, and attractor-width of the desirable regime of the lake
(from Carpenter, Brock & Hanson 1999)

> “A moderate proportion of phosphorus-intensive farms would be maintained and adjusted to bring soil phosphorus toward levels that reduce the risk of eutrophying the lake. Such policy experiments may be expensive, in the sense that they appear to be suboptimal economically (…), yet, they are “safe” in the sense that they expand the desirable attractor and enable the manager to learn how the attractor responds to policy choice. Information gained from these experiments would be used to adjust policies with the goal of sustaining both water quality and farming activity. Continual learning and continual

> adjustment become the norm. The result would be cycles of smaller amplitude, with general low levels of lake water phosphorus, variable proportions of phosphorus-intensive farms, and moderately large attractor width" (Carpenter, Brock & Hanson 1999, 19).

A crash, followed by several hundred years of exploratory, adaptive, sustainable management, is illustrated in *Figure 12*. According to Carpenter, Brock & Hanson (1999), it is important to recognize that *sustainability cannot be achieved by seeking a fixed stable point*. Frozen policy is a route to disaster.

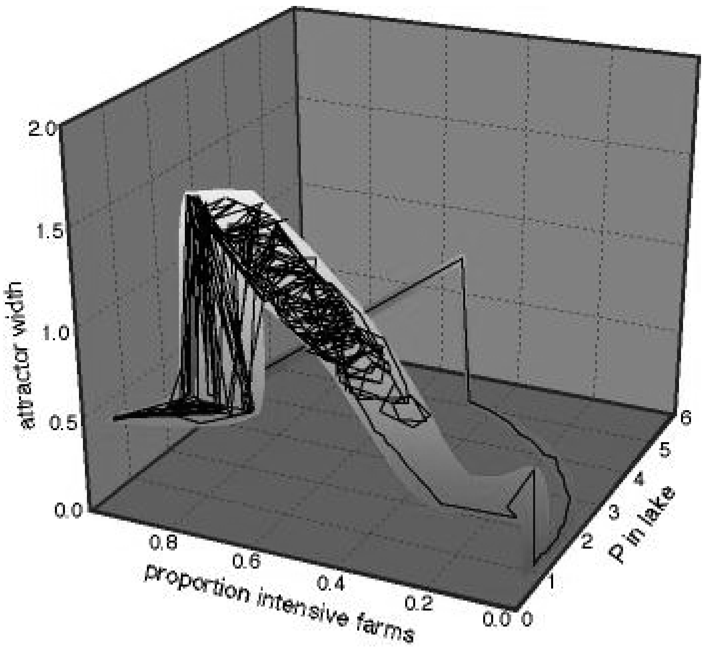

Figure 12: Crash of an lake-SES followed by several hundred years of sustainable management
(from Carpenter, Brock & Hanson 1999)

Similarly, the *minimal model of ecosystem management* from Peterson, Carpenter & Brock (2003) consists of a model of lake dynamics, a learning process, and a management decision-making process. The authors also conceive soil phosphorus as an indicator and the distance between the current level and the critical load as a measure of ecosystem resilience and suggest to launch safe, informative experiments to test models beyond the range of available data.

An even more comprehensive approach has been explored by the *Forum of Science and Technology for Sustainability* (Clark et al. 2001, Kates et al. 2001, Clark & Dickson 2003, Polsky et al. 2003, Turner et al. 2003, 2003b). As part of the emerging research paradigm *sustainability science* (Clark & Dickson 2003), a framework for *vulnerability analysis* has been proposed (Turner et al. 2003) as it is illustrated in *Figure 13*.

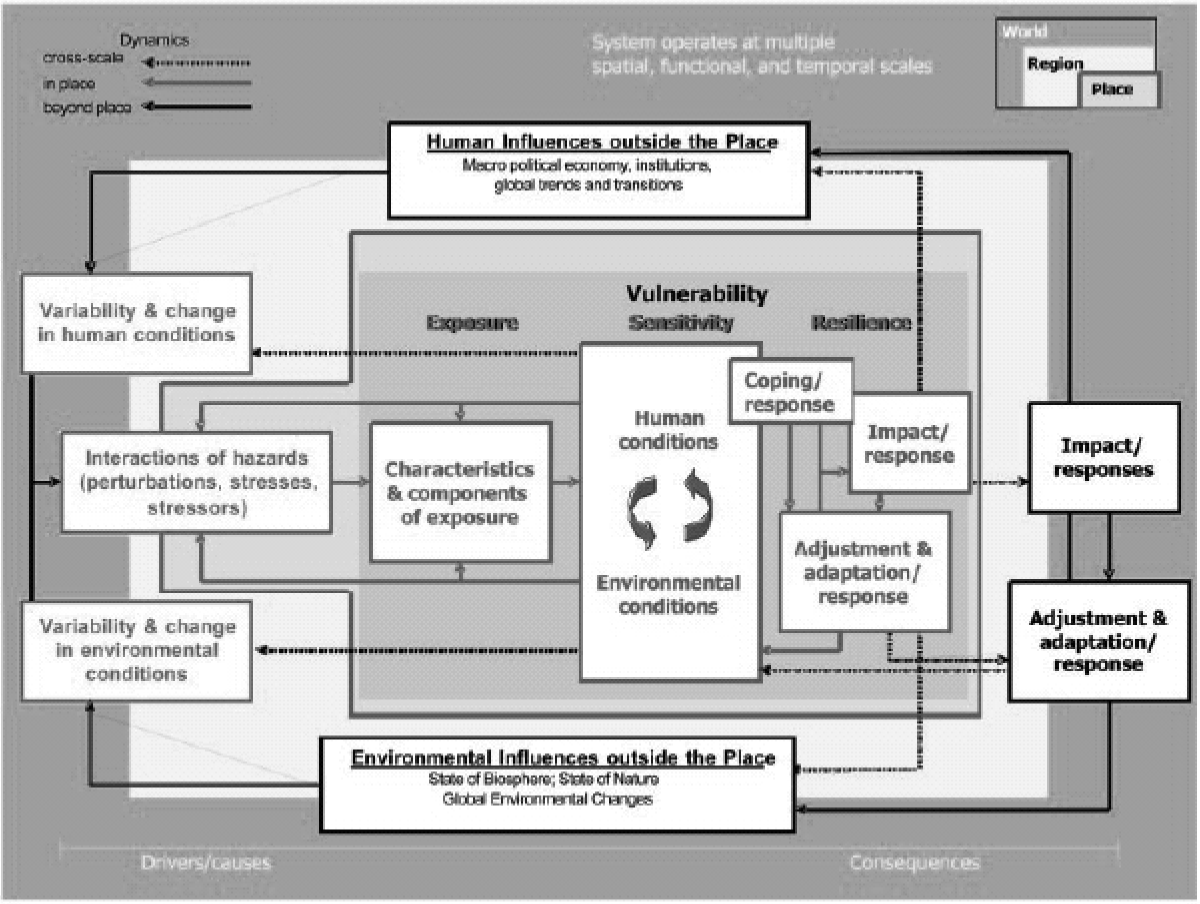

Figure 13: Comprehensive model for vulnerability assessments; vulnerability comprises exposure, sensitivity, and ecosystem as well as engineering resilience.
(from Turner et al. 2003)

Vulnerability is hereby defined as “the degree to which a system, subsystem, or system component is likely to experience harm due to exposure to a hazard, either a perturbation or stress/ stressors” (Turner et al. 2003, 8074). It represents a multidimensional concept that includes *exposure*, i.e. the degree to which a human group or ecosystem comes into contact with particular stresses, *sensitivity*, i.e. the degree to which an exposure unit is affected by exposure to any set of stresses, and ecosystem (and engineering) resilience, which is considered to be - seen from a slightly different perspective - the ability of the exposure unit to resist or recover from the damage associated with the convergence of multiple stresses (Clark et al. 2000).

Essential elements of an vulnerability analysis, particularly those aimed at sustainability, include (i) multiple interacting perturbations and stresses and the sequencing of them, (ii) *exposure* beyond the presence of a perturbation and stress, including the manner in which the coupled system experiences hazards, (iii) *sensitivity* of the coupled system to the exposure, (iv) the system's capacities to cope or respond (*ecosystem resilience*), (v) the system's restructuring after the responses taken, and (vi) nested scales and scalar dynamics of hazards, coupled systems, and their responses (Turner et al. 2003).

A vulnerability analysis may be undertaken at any spatial or temporal scale suitable for the problem in question. A comprehensive analysis ideally considers the totality of the system but, in fact, necessitates a "reduced" vulnerability assessment.

As illustrated in *Figure 13*, the *basic architecture* consists of (1) linkages to the broader human and biophysical (environmental) conditions and processes operating on the coupled system in question, (2) perturbations and stressors that emerge from these conditions and processes, and (3) the coupled human-environment system of concern in which vulnerability resides, including exposure and responses (Turner et al. 2003).

According to Polsky et al. (2003), there are *five minimal criteria* that vulnerability assessments should satisfy, i.e. (1) the knowledge base for analysis should include all relevant academic disciplines, (2) vulnerability research and assessments should be *place-based*, i.e. on a landscape scale, (3) the stresses examined should be recognized as multiple and interacting instead of unique or multiple independent, (4) the research should allow for differential adaptive capacity, i.e. the heterogeneity of the adaptive capacity of the people in a given place has to be recognized, and (5) the information should be prospective and historical.

Moreover, Polsky et al. (2003) propose eight steps as methodology for an vulnerability assessment, which include (1) to define study area in tandem with stakeholders, (2) to develop in-depth knowledge of the stakeholders, the ecosystem services they value and why, the important vulnerability drivers over which they may have control, (3) to hypothesize who is vulnerable to what, i.e. to identify which stresses and interactions among stresses pose the greatest risk of harm to people and the environmental services on which they depend, (4) to develop a causal model of vulnerability, (5) to find indicators for the components of vulnerability, (6) to weight and combine the indicators, (7) to project future vulnerability and (8) to communicate vulnerability creatively.

The *Forum on Science and Technology for Sustainability* heavily draws on the concept of ecosystem resilience proposed by the *Resilience Alliance*, and embeds it in a larger framework of vulnerability analysis or assessment. A full vulnerability assessment following the framework and basic architecture proposed by Turner et al. (2003) may lie well beyond the capacities of most research efforts (Turner et al. 2003b). Yet this general conceptual framework can be seen as providing a useful point of departure for examining vulnerability. For practical and theoretical reasons, such frameworks should be simplified to suit the specifics of a given application.

5 Summary & Synthesis

Within the *argumentative space of sustainability discourse*, various authors regard ecosystem resilience as one of the key research issues of the future (Perrings et al. 1995, Folke, Holling & Perrings 1996, Levin et al. 1998, Perrings 2002). A group of highly acknowledged ecologists and ecological economists regard ecosystem resilience as an useful index of environmental sustainability and conclude that economic activities are sustainable only if the life-support ecosystems upon which they depend are resilient (Arrow et al. 1995). The concept of ecological resilience is considered to be able to address the resource management issues of the day (Holling & Meffe 1996, Gunderson & Holling 2002). Some authors identify ecosystem resilience as a guideline for a *Theory of Sustainable Development* (e.g. Ott 2001). Within the German debate, both Ott (2001) and Kopfmüller et al. (2001) see clearly the theoretical relevance of the ecosystem resilience concept and are interested in the possibilities to operationalize it.

The research questions put forward by the ecosystem resilience concept lead to an immense dialogue dubbed the *Resilience Project*, a five-year collaboration among an international group of ecological economists, ecologists, social scientists, and mathematicians labelled *Resilience Alliance*.

For this thesis, I have reviewed relevant papers and books most of which are the scientific product of the *Resilience Alliance* since this scientific network presents the leading authority in the field of ecosystem resilience. The purpose of the thesis is two-fold: (1) to present the theoretical fundamentals of the concept of ecosystem resilience and those concepts that are related to ecosystem resilience which together, therefore, constitute its *background theory*, and (2) to examine the abilities to estimate ecosystem resilience empirically as well as theoretically and to unfold its relevance for a Theory of Sustainable Development.

Within sustainability discourse ecological resilience is being discussed with respect to some notions and concepts that are used frequently within the relevant debate.

Ecological resilience is seen to be connected to the notion of limits to growth and earth's carrying capacity (Folke et al. 1994, Arrow et al. 1995, Perrings et al. 1995b, Seidl & Tisdell 1999, Perrings 2002, Ekins, Folke & deGroot 2003). Outer limits refer to the carrying capacity of the environment to support human activities at various scales and this capacity is, in turn, dependent on the ecosystem resilience of ecosystem functioning (similar Folke et al. 1994).

Moreover, ecosystem resilience appears to be a tool in order to put the rather vague concept of critical natural capital that is used within Ecological Economics in concrete terms (Deutsch, Folke & Skanberg 2003, Ekins 2003, Ekins & Simon 2003). Hereby, the concept of ecosystem resilience provides a framework for analyzing the performance and maintenance of ecosystem functioning (Deutsch, Folke & Skanberg 2003).

The distinction of weak and strong sustainability reflects one of the main discussions within sustainability discourse (Daly 1996, Faucheux & O'Connor 1998, Ott 2001, 2003, Neumayer 2003). The conception of strong sustainability represents a prerequisite for the political and moral relevance of the concept of ecological resilience.

Within ecological science 70 distinct concepts and 163 definitions with respect to “stability” have emerged (Grimm, Schmidt & Wissel 1992). In order to avoid confusion, the meaning of distinct stability properties has to be delimited. In this respect, Grimm & Wissel (1997) and Hansson & Helgesson (2003), respectively, distinguish between three fundamental stability properties, namely constancy, resilience and persistence and constancy, resilience and robustness, respectively. Hereby, *constancy* corresponds to “staying essentially unchanged”, *resilience* is defined as “returning to a reference state (or dynamic) after a temporary disturbance” and *persistence* matches “persistence through time of an ecological system” (Grimm & Wissel 1997). “Stability” is, thus, a meta-concept for the three distinct stability properties.

Resilience in its *original-ecological sense* has been defined in two different ways in the ecological literature (Holling 1986, Holling et al. 1999, Gunderson & Holling 2002).

The first, more classical, definition concentrates on stability near an *equilibrium steady state*, where the rate and speed of return to pre-existing conditions after perturbation are used to measure the property (deAngelis 1980, Pimm 1984, Tilman & Downing 1994, WBGU 2000, Lugo et al. 2002). Focus of study are slow dynamics near a global equilibrium (Ludwig, Walker & Holling 1997, 2002).

The second definition emphasizes conditions far from any equilibrium steady state, where instabilities can shift a system to another *basin of attraction* which is controlled by a different set of variables and characterized by a different structure (Holling 1973, 2001, Gunderson 2001, Gunderson & Pritchard Jr. 2002, Holling & Gunderson 2002, Walker et al. 2002, 2004). Resilience, understood in this way, is the “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior” (Gunderson & Holling 2002, 4). Focus of study are slow dynamics in a region around an equilibrium that separates two alternative basins of attraction (Ludwig, Walker & Holling 1997, 2002).

These two different definitions of the resilience concept reflect *two different stability properties* of ecological systems: resilience and persistence (Grimm & Wissel 1997) or resilience and robustness (Hansson & Helgesson 2003), respectively. Both definitions are contrastive aspects of the meta-concept “stability”. The collaborative research group *Resilience Alliance* highlights this distinction and labels the property resilience *engineering resilience* and robustness or persistence, respectively, *ecosystem resilience*.

The ecosystem resilience concept draws on two major related concepts which constitute the *background theory of ecosystem resilience*.

First, the ecosystem resilience concept is related to a heuristic of complex adaptive systems. According to this view, ecosystem dynamics exhibit an episodic behaviour passing through (1) an *exploitation* and pioneer species (r) phase and (2) a *conservation* (K) phase, where biomass is slowly accumulated before suddenly the more and more tightly connected biomass is released in the *creative destruction* (Ω) phase and reorganized in the *alpha* (α) phase. If the omega phase represents the end, then it is immediately followed by the alpha phase, the beginning, to complete the (adaptive) 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 1986, Holling et al. 1999, Gunderson & Holling 2002).

Since dynamics occur at each scale the notion of a set of adaptive cycles on each level of an ecoystem’s hierarchy is summarized in the concept of a *Panarchy*. Several interactions between the levels are identified. Among them, *Revolt* is a process where fast and small levels that experience the release (Ω) phase trigger change in slower and larger levels currently existing in the conservation (K) phase. The interaction *Remember* occurs when slower and larger levels exhibiting the K phase influence faster and smaller levels of the hierarchy that exist in the (α) phase.

Within this notion of dynamics of complex adaptive systems, ecosystem resilience is conceived as a changing quantity that, in general, occurs on each level of an ecosystem's hierarchy. It is high in Ω and α phase, low in r phase and still lower in K phase. Since ecosystem resilience is low, the K phase represents a stage in ecosystem dynamics that is an "accident waiting to happen" until creative destruction (Ω phase) and reorganization (α phase) occurs (Holling 2001). This *back loop* (Ω and α phase) [as opposed to the *forward loop* (r and K phase)] of the ecosystem renewal cycle represents an engine of renewal and novelty for ecosystem dynamics.

Following the *extended keystone hypothesis* (Holling 1992), the hierarchical structure of ecosystems is conceived to be primarily regulated by a small set of plant, animal, and abiotic processes that can be identified as the essential or key variables. The complexity of adaptive systems can be traced to interactions among three to five sets of variables, each operating at a qualitatively distinct speed and scale. Slow variables operating at a large and slow scale and their interactions to fast variables are considered to control ecosystem dynamics (Holling et al. 1999, Holling et al. 2002).

The second feature of ecosystem resilience background theory is the concept of *alternative stable regimes*, which says that ecosystems can exhibit several stable regimes depending on the value of controlling slow variables. Recent studies have provided a strong case for the existence of alternative stable regimes in various important ecosystems such as in lakes (Carpenter 2001, Scheffer et al. 2001), marine systems (Done 1991) and oceans (Hare & Mantua 2000), in deserts (Foley et al. 2003) as well as rangelands (Perrings & Walker 1997, Janssen et al. 2004), in woodlands (Dublin, Sinclair & McGlade 1990) as well as forests (Holling 1986, Peterson 2002, Peterson et al. 2002).

These stable regimes correspond to alternative *basins of attraction*, specified as regions in *state space* (the variables considered and their relation) in which the system tends to remain (Walker et al. 2004). They are not static, rather they expand and contract, and disappear in response to changes in slow variables

(Holling 1986, Walker et al. 2004). If a disturbance regime is severe enough to overcome the stabilizing mechanisms of a basin of attraction, a regime shift occurs. Then, a critical level of a controlling variable of the system is passed, the nature and extent of internal feedbacks alters (Walker & Meyers 2004) and the ecosystem reorganizes around another set of controlling key variables and processes (Holling & Gunderson 2002). This breakpoint between two regimes of a system, i.e. the critical values of variables around which the system shifts from one stable regime to the other is termed *ecological threshold* (Muradian 2001, Walker & Meyers 2004).

In the heuristic of a *stability landscape*, basins of attraction are illustrated as valleys and ecological thresholds as boundaries (Walker et al. 2004). Slow variables determine in which basin of attraction the ecosystem will stay. The stability landscape reflects these slow variables, within which the faster variables move around, reacting to the changes in the “topography” (Jansson & Jansson 2002, Gunderson & Walters 2002).

With respect to stability landscapes, ecosystem resilience corresponds to the size of the basin of attraction (Holling 1973, Scheffer & Carpenter 2003) or the ability to stay in the same basin of attraction (Walker et al. 2004). Walker et al. (2004) distinguish four crucial *aspects of ecosystem resilience* three of which correspond to different characteristics of a basin of attraction. (1) *Latitude* is defined as the maximum amount the system can be changed before losing its ability to recover which corresponds to the width of the domain of attraction. (2) *Resistance* is defined as the ease or difficulty of changing the system which is related to the topology of the domain. (3) *Precariousness* matches 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). The fourth aspect of ecosystem 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. This notion of ecosystem resilience represents a *holistic* and *qualitative concept* which corresponds to Grimm & Wissel’s (1997) persistence. It is holistic since it refers to the entire ecosystem and qualitative because the focus is no longer on dynamics with its quantitative details, but on the qualitative question of whether the whole set of variables of interest used to characterize the system remains within certain boundaries (cf. Grimm et al. 1999 and Walker 2004).

Several mechanisms have been proposed that are seen as responsible for the *emergent property* ecosystem resilience and constitute distinct components of *biophysical ecosystem resilience* (as opposed to *societal* or *social-ecological ecosystem resilience*). The following discussion draws heavily on decade-long debates within ecological science about the relation of biodiversity and ecosystem functioning and stability properties, respectively (Loreau et al. 2001, 2002, Naeem 2002). Hereby, the importance of a comprehensive notion of biodiversity on all levels of biological organization (genes, population, species, landscape) (e.g. Lincoln, Boxshall & Clark 1998) and a functional rather than a mere species richness approach to biodiversity gets important.

The first *ecosystem resilience mechanisms* is termed ecological redundancy. There is *ecological redundancy* if more than one species exist within a precisely separated guild that performs the same ecological function – a functional group (Walker 1992, 1995, Gitay et al. 1996). Functional groups that are represented by a single or only a few species are dubbed *keystone functional groups*. Conservation focus shifts to the species in functional groups with low redundancy since the loss of these species would result in loss of overall ecosystem function (Walker 1992, 1995, Mooney et al. 1996, WBGU 2000).

If a functional group comprises more than one species, responses of these species to various disturbances are likely to be different. The variability in responses is termed *response diversity* and defined as the diversity of responses to environmental change among species that contribute to the same ecosystem function (Elmquist et al. 2003). If diverse species within the same functional group respond to disturbance regimes un-identical the possibility increases that this functional group is able to perform its function despite various disturbance events.

This is due to a mechanism termed *functional compensation*, i.e. structural change within communities that stabilizes ecosystem process rates (Frost et al. 1995, Tilman 1996, Carpenter & Cottingham 2002). Response diversity and compensation within functional groups is seen to be critical for ecosystem resilience as buffering effect to various disturbances (Holling et al. 1995, Naeem 1998, Yachi and Loreau 1999, Elmquist et al. 2003). In this sense, each species can be critical and contributes to ecosystem resilience (Gunderson 2000). This claim is dubbed the *insurance hypothesis* (Yachi & Loreau 1999).

This compensating overlap of ecological function *at the same scale* (e.g. different plant taxa) performing as insurance capital for ecosystem functioning results in persistence of the whole ecosystem regime, a pattern which is termed *within-scale resilience* (Peterson, Allen & Holling 1998, Holling et al. 2002, Gunderson et al. 2002). Ecosystems, however, are structured hierarchically in a lumpy manner across various scales (Holling 2001, Holling & Gunderson 2002). Ecosystem processes, such as seed dispersal, can be replicated at different scales (Holling et al. 2002, Gunderson et al. 2002). The property that arises of these mechanisms is dubbed *(a)cross-scale resilience* (Peterson, Allen & Holling 1998, Holling et al. 2002, Gunderson et al. 2002).

Within-scale resilience, thus, complements across-scale resilience which is summarized in the concept of *imbricated resilience* (Holling et al. 2002). Thus, ecosystem resilience derives from overlapping function within scales and reinforcement of function across scales as a product of functional diversity and ecological redundancy (Peterson, Allen & Holling 1998).

Another key component of ecosystem resilience is represented by the concept of *ecological memory*, which is defined as “the 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).

Internal memory comprises *biological legacies* which include 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) (Franklin & MacMahon 2000). Internal memory is limited by the assembly rules (e.g. facilitation, competition, trophic interactions) that determine which species proliferate despite or after disturbance.

External memory is provided by support areas of colonizing species and is restricted by the permeability of the matrix between the colonized patches (Bengtsson et al. 2003). This buffering capacity depending on areas in the vicinity of the patch affected by disturbance is termed *spatial resilience* (Nyström & Folke 2001, Bengtsson et al. 2003). Walker (2002) suggests that in semi-arid rangelands decreasing spatial heterogeneity indicates declining ecosystem resilience. Similarly, van de Koppel & Rietkerk (2004) propose that spatial interactions on coarser scales increase both engineering and ecosystem resilience of vegetation cover in arid regions facing variable rainfall levels. For coral reefs, Nyström & Folke (2001) identify three basic and interacting parts of ecological memory that ensure spatial resilience: biotic legacies, support areas of potentially colonizing species, and mobile link organisms. *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).

Despite historic debates about the relevance of the two components of ecological memory (internal and external), Bengtsson et al. (2003) find no conflict between the two components and state that, depending on disturbance intensity and landscape composition, their relative importance may vary.

There is another mechanism that is responsible for ecosystem resilience which is related to the notion of the dynamics of complex adaptive systems explored above. The *back loop of the adaptive cycle* (Ω and α phases) and the *Revolt-interaction* across scales as part of the panarchy reflects the importance of renewal and variability in relatively small-scale patches or, more general, at each level of the ecosystem hierarchy. Walker & Abel (2002) suggest that repetitions of the cycle lead to a combination of species that is better adapted and, therefore, more ecosystem resilient to environmental fluctuations.

This implies the recognition of the relevance of small-scale disturbances as *intrinsic parts* of ecosystem dynamics (Holling 1986, Gunderson & Holling 2002, Bengtsson et al. 2003), i.e. disturbance is endogenous to the cyclic process of ecosystem dynamics (Folke, Berkes & Colding 1998, Berkes & Folke 2003). Some loss of ecosystem resilience, at some scales, is an inevitable feature of the cross-scale dynamics in complex adaptive systems. However, a system that loses ecosystem resilience at small, more manageable scales of organization will be more ecosystem resilient than one where these losses occur at larger scales (Walker et al. 2004). In this sense, ecosystem resilience within a system is generated by destroying and renewing systems at smaller, faster scales (Gunderson 2000). Small-scale disturbance is seen to contribute to an increase in species richness, heterogeneity and function. Small-scale disturbance increases species diversity producing redundancy in functions, and, in turn, functional redundancy ensures the persistence of functions in response to disturbance (Jentsch, Beierkuhnlein & White 2002). This phenomena can be dubbed the *renewal aspect of ecosystem resilience* (similar Bengtsson et al. 2003).

This view on the persistence of ecosystem dynamics and structure depends on the notion that disturbances on smaller scales produce equilibrium or persistence at larger scales (Jentsch, Beierkuhnlein & White 2002). In this respect, Turner et al. (1993) predict both the presence and absence of equilibrium and variance in ecosystem states as a function of two ratios: the ratio of the disturbed area to the landscape area and the ratio of the disturbance frequency to the time needed for successional recovery. The smaller the patch relative to the recovery time, the greater the chance for dynamic equilibrium in all patches.

To sum up, ecosystem resilience mechanisms can be found on at least five levels of an ecosystem's hierarchy (Lugo et al. 2002). (1) In individuals as part of their responses to their environment. Examples are high reproduction, high mobility, phenotypic plasticity, flexible feeding behaviour and physiological tolerance, which represent mechanisms Grimm et al. (1999) identified for the Wadden Sea, Germany. (2) In the cumulative effect of how organisms of different species react to their respective environments, e.g. functional diversity and response diversity as cumulative mechanisms to absorb disturbances. (3) In the effect of legacies after an event, e.g. biotic legacies. (4) As the consequence of inputs from, or effects of, processes from other levels in the hierarchy, e.g. internal and external ecological memory. (5) As inherent characteristics of ecological systems, such as negative feedback function of storages (Lugo et al. 2002).

The influence of biotic conditions on ecosystem structure and function is often strong. In many ecosystems, these abiotic variables represent the slow and controlling variables that drive ecosystem function. The loss of several ecosystem resilience mechanisms such as ecological redundancy, and ecological memory that is related to biotic variables of the ecosystem results in systems that are more vulnerable to change. But, finally, it is then the changes in the slow variables that define the loss of ecosystem resilience (similar Gunderson & Walters 2002, 177) which corresponds to a decrease of the size of the basin of attraction in a stability landscape. In many cases, these slow variables tend to be abiotic variables.

Considering the previous discussions at least six levels of meaning can be identified with respect to the term ecosystem resilience. In 1973, Holling originally defines (ecosystem) 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). 29 years later, Holling and Gunderson define ecosystem resilience similarly as the “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour” (Holling & Gunderson 2002, 4). This *original-ecological meaning* more or less corresponds to persistence *sensu* Grimm & Wissel (1997) and to robustness *sensu* Hansson & Helgeson (2003).

Due to the progress in ecological debates about related concepts (e.g. complex adaptive systems, alternative stable regimes, thresholds, ecosystem resilience mechanisms) the ecosystem resilience concept has been extended in its meaning which is reflected in its definition. Referring to these debates, Walker et al. (2002) define ecosystem resilience as “the potential of a system to remain in a particular configuration and to maintain its feedbacks and functions, (…) [which] involves the ability of the system to reorganize following disturbance-driven change” (Walker et al. 2002). Similarly, Walker et al. (2004) identify ecosystem resilience as “the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks” (Walker et al. 2004, 2). In this *extended-ecological meaning*, ecosystem resilience comprises three defining characteristics: (1) the amount of disturbance a system can withstand and still retain the same controls of function and structure, (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). In my view, the last two characteristics extend the meaning of the original definition from Holling (1973), as ecosystem resilience is seen to comprise not only the ability to absorb disturbances and to stay in the same basin of attraction but also to include the capabilities of self- and re-organization and renewal.

In the ongoing debate on theoretical aspects of the ecosystem resilience concept and its relevance within a Theory of Sustainable Development, some authors recognized the importance of a clear and measurable definition. Carpenter et al. (2001), for instance, suggested an *operational meaning*, thus, a measurable concept of ecosystem resilience by clearly defining ecosystem resilience *of* what *to* what. In order to analyse ecosystem resilience appropriately one has to specify the time scale as well as the spatial scale, the reference regime and the disturbances of interest (Carpenter et al. 2001, Walker et al. 2002).

When proposing ecosystem resilience statements, one has to refer to underlying, controlling slow variables. In this *ecological-systemic meaning* ecosystem resilience corresponds to the capacity of ecosystems (i.e. the underlying mechanisms and variables) to provide services in the face of a fluctuating environment and human perturbations.

The focus of study within sustainability discourse is not on ecosystems and the environment only but rather on several other dimensions, such as the economic system and the social system. Therefore, some authors refer to social systems and identify *social resilience* as “the ability of human communities to withstand external shocks to their social infrastructure, such as environmental variability or social, economic, and political upheaval” (Adger 2000, 347). Clearly, this *sociological meaning* resilience corresponds to the original-ecological meaning of ecosystem resilience.

A crucial point in the work of the *Resilience Alliance* on ecosystem resilience is their claim that a sound ecosystem resilience analysis has to take into account not only the natural system, but rather whole social-ecological systems (Holling 1999, Folke et al. 2002, Gunderson & Holling 2002, Berkes, Elmqvist & Olsson 2003). The separation of social systems and natural systems is seen to be more of a recent mental artefact than an observation of the real world. Therefore, some authors identify *social-ecological resilience* as “the capacity of ecosystems to sustain societal development and progress with essential ecosystem services” (Folke, Colding & Berkes 2003, 354) which constitutes the *social-ecological meaning* of ecosystem resilience.

All these meanings of the term ecosystem resilience exist in the literature relevant to the subject. What is apparent is that even though there are three fundamental stability properties (constancy, resilience and robustness or persistence) the majority of the authors focus on ecosystem resilience (i.e. robustness or persistence) as opposed to engineering resilience (i.e. resilience).

To put emphasis on one of the two different stability properties has very different consequences for evaluating, understanding, and managing ecological systems. The first definition focuses on efficiency, control, constancy, and predictability, the second definition focuses on robustness, adaptiveness, variability and unpredictability. These are fundamentally different views of science (Berkes & Folke 1998, Gunderson & Holling 2002).

Some authors claim that sustainable relationships between people and nature require an emphasis on ecosystem resilience (Peterson, Allen & Holling 1998, Gunderson & Holling 2002). Engineering resilience focuses on small portions of a system's stability landscape whereas ecosystem resilience focuses upon its contours. It does not take into account the existence of alternative stable regimes and basins of attraction, respectively (Peterson, Allen & Holling 1998). Moreover, analysis based on engineering resilience lead to scientific examinations and results that often do not meet the appropriate landscape scale in order to be useful for objectives for Sustainable Development and environmental management. In order to address environmental management issues the analysis has to be based on the concept of ecosystem resilience (Holling & Meffe 1996).

However, some voices in the debate suggest that the emphasis on ecosystem resilience has to be conceived more as a paradigm rather than a reflection of reality in nature (Gunderson & Holling 2002, Gunderson & Pritchard 2002), i.e. they do not doubt that other stability properties can be relevant for ecosystem analysis.

In this respect, Grimm et al. (1999) show that the relevance of different stability properties might depend on the temporal and spatial scale of concern. In the case of the Wadden Sea in Germany, the concept of ecosystem resilience gets more important on larger (landscape) scales. From a different perspective, Carpenter & Cottingham (1997) state that although engineering resilience may miss important features of a system's "stability", empirical estimates of return rate may be useful for comparing responses of different systems to a given perturbation.

Models of ecosystem resilience illustrate that ecosystem resilience focuses on dynamics in a region that separates two alternative basins of attraction, i.e. a *separatrix*, whereas engineering resilience concentrates on conditions near a *global equilibrium*. Hereby, long return times may be a diagnostic tool for both low ecosystem resilience and engineering resilience or for disturbances that are large enough to take the system near an unstable equilibrium (Ludwig, Walker & Holling 1997, 2002, similar Wissel 1984).

Additionally, the *extended-ecological meaning* of ecosystem resilience includes "a measure of the maximum amount the system can be changed before losing its ability to recover" (Walker et al. 2004), which corresponds to the width of the domain of attraction and can be defined in terms of engineering resilience

(Grimm & Wissel 1997). It is arguable whether the ecosystem resilience concept sensu Walker et al. (2004) includes the concept of engineering resilience or not and whether this leads either to a renewed confusion about stability terms or to a more comprehensive notion of “stability”.

Therefore, the emphasis on ecosystem resilience seems to be useful in order to recognize the importance of alternative basins of attraction, non-linear behaviour, the relevance of slow variables and ecosystem resilience as a changing quantity. It can foster to break away from an equilibrium-centred view on “stability”. The emphasis on ecosystem resilience is *not* useful *if* it is to reflect the reality of nature as if ecosystem resilience was the only observable stability property that ecosystems perform. Other stability properties can be highly relevant for the analysis of ecological systems.

The concept of ecosystem resilience has not been of theoretical interest only, rather some sustainability scholars consider it to be useful to address the resource management issues of the day (Holling & Meffe 1996). In this respect, both the *possibility to operationalize ecosystem resilience*, i.e. the possibility to identify inter-subjectively for a concrete situation whether a term applies to empirical reality (Jax 2002) or to *measure ecosystem resilience*, and the possibility to *implement ecosystem resilience* are seen as vague while being of high interest (Kopfmüller et al. 2001, Ott 2001).

Facing these tasks, Carpenter et al. (2001) propose a clear and measurable concept of ecosystem resilience *of* what *to* what which represents the basis for an appropriate *ecosystem resilience analysis* (Walker et al. 2002). The goals of an ecosystem resilience analysis are (1) to prevent a socio-ecological system from moving into undesirable basins of attraction in the face of external stresses and (2) to nurture and preserve the elements that enable the system to renew and reorganize itself following a massive change (Walker et al. 2002).

An important point here is that ecosystem resilience - in its descriptive meanings - is not something that is desirable as such (Carpenter et al. 2001, Carpenter & Cottingham 2002, Gunderson & Holling 2002, Walker et al. 2002). Ecosystem resilience can be desirable or undesirable which depends on the regime of the system of concern[109]. For example, a degraded savannah or a polluted lake can be highly ecosystem resilient.

An appropriate ecosystem resilience analysis includes five essential steps.

First, the concept of ecosystem resilience as such has to be examined. As explored above, Walker et al. (2004) identify four distinct but related aspects of ecosystem resilience, i.e. *latitude*, *resistance*, *precariousness* and *panarchy*. The scientist has to specify which aspect of ecosystem resilience she is going to estimate in order to avoid confusion.

The second step is termed the *to-what part* of an ecosystem resilience analysis and examines to what disturbance regime a given ecosystem is to be ecosystem resilient. Hereby, the response of the ecosystem depends largely on the type, severity and scale of the disturbance event (Pickett and White 1985, Frelich & Reich 1998, Romme et al. 1998, Turner & Dale 1998). This corresponds to the claim that the notion of a general ecosystem resilience in the face of any and all disturbances is not achievable (Grimm & Wissel 1997, Walker & Abel 2002).

Step three is the specification of the temporal and spatial scale of investigation. It constitutes the first feature of the *of-what part* of ecosystem resilience analysis. In general, *scale* can be defined as “a range of spatial and temporal frequencies” (Peterson, Allen & Holling 1998, 11). The specification of scale is important since ecosystem resilience of a SES in one time period can be gained at the expense of a succeeding period of time or the ecosystem resilience at one spatial extent can be subsidized from a broader scale (Carpenter et al. 2001, Folke et al. 2002). Clearly, a chosen scale of examination is affected by scales below and above.

Step four represents the identification of desired ecosystem services. Daily (1997) defines *ecosystem services* as the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life. As the ecosystem resilience of different ecosystem services can be in conflict (e.g. agricultural and aquatic ecosystem services) it is essential to decide which ecosystem services are of primary concern within a given area, i.e. what exactly *should* remain ecosystem resilient with respect to a given disturbance regime. The ecosystem resilience concept does not relieve us to reflect on what nature we want

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[109] These considerations apply to the descriptive meanings of ecosystem resilience (original-ecological, extended-ecological, systemic-ecological, operational) only. The normative meanings of ecosystem resilience already include evaluative judgements in favour of certain ecosystem states.

to have. This task is taken into account, first, by ethical reflections about our moral attitude and behaviour towards natural entities (*environmental ethics*) and, second, by *environmental assessment* procedures or *wildlife conservation evaluation* (Eser & Potthast 1997) and evaluative judgements about *environmental quality objectives* (Wiegleb 1997).

The sixth step for an ecosystem resilience analysis consists in the identification of slow, controlling variables. The continued performance of ecosystem services (functions-for humans) requires underlying dynamics of ecosystem functioning which is termed the functions-of natural capital (Ekins 2003). These functions-of are constituted by the basic processes and cycles – the slow variables - in the internal functioning of natural systems, which are responsible for sustaining and maintaining the ecosystem resilience of natural systems (similar Holling et al. 1995). Carpenter et al. (2001) suggest that we can conceive of the slow variables as defining the underlying structure of the system, while the fast variables reveal the dynamics of this underlying structure.

Seen from this perspective ecosystem resilience is a measure of the capacity of an ecosystem to maintain services in the face of a fluctuating environment and human perturbations (the ecological-systemic meaning) which points to underlying slow variables of ecosystems that ensure the provision of ecosystem services and foster the ability to stay in the same basin of attraction.

There is a principal problem to *measure* ecosystem resilience since this would imply the alteration of ecosystem structure and functioning and this is what most environmental management is attempting to avoid (Peterson 2002).

In this respect, some authors suggest to *estimate* ecosystem resilience by plotting the equilibria of the system on axes of the rapidly changing variable and a more slowly changing variable. The plot then shows upper and lower sets of stable states separated by an unstable set of equilibria – which corresponds to a *bifurcation diagram* (Carpenter et al. 2001, Peterson, Carpenter & Brock 2003). In the model, ecosystem resilience is tracked as the size of the attractor of the desired regime or the distance between the current state of the slow variable and the critical state, which has to be estimated empirically (Peterson, Carpenter & Brock 2003) or theoretically (Wissel 1984, Ludwig, Walker & Holling 1997, 2002). In practice, ecosystem resilience could be tracked by monitoring the slowly changing ecological variables that control the desired regime (Carpenter et al. 2001). To my view, it remains unclear, however, how the values of attractor size or precariousness (the distance to the ecological threshold) should be estimated empirically.

It would be of high value to predict the location of an ecological threshold. In this respect, Walker & Meyers (2004) provide a developing database including five classes and eleven categories of ecological thresholds. Data from areas that have already undergone a regime shift can be compared with data from similar environments that have not. Whether an ecological threshold can be identified before it has been crossed depends on data extrapolation from similar closely related systems. Additionally, Wissel's (1984) and Ludwig, Walker & Holling's (2002) theoretical analyses may help to predict the position of ecological thresholds with respect to environmental variables.

However, even if the position of an ecological threshold cannot be predicted at the current knowledge (Muradian 2001) the existence of alternative basins of attraction can be and has been predicted (Foley et al. 2003). By examining the underlying state of environmental systems and their degree of nonlinearity, scientists may be able to predict which regions of the world are susceptible to regime shifts and which are not (Scheffer & Carpenter 2003).

The identification of *indicators* represents a vital option for environmental management. With respect to slow controlling variables, indicators clearly should reflect the internal dynamics of ecosystems and their ecosystem resilience, respectively.

For example, Jansson & Jansson (2002) identify phosphorus storages and the nitrogen/phosphorus ratio in the sediments, as well as the pool of organic matter as slow variables for the Baltic. These variables can be estimated and then be used as indicators for a given ecosystem regime that exhibits desired or undesired ecosystem services.

The identification and specification of slow variables and indicators that represent them is the sixth and last step for an appropriate ecosystem resilience analysis. It constitutes one of the most important tasks within further investigations of ecosystem resilience. Such an estimation would apparently provide useful data for an adaptive, environmental management approach. It might be difficult, however, to find the controlling underlying variables for each ecosystem type. It seems to be relatively easy for aquatic types since nutrient storages play a key role in these ecosystems. According to Volker Grimm working at the UFZ

('*Umweltforschungszentrum*') Leipzig-Halle, this leads to the debate about modelling within theoretical ecology. Grimm (1999) argues for the relevance of bottom-up or *individual-based models* but insists on using top-down or *state variable models* in a complementary way in order to achieve an appropriate integrated view of ecosystem dynamics.

For an ecosystem resilience analysis there might be no single "correct" scale since different processes are likely to be important on different scales (Levin 1992). Moreoever, environmental issues tend to be neither small-scale or large-scale but cross-scale in both space and time (Folke, Berkes & Colding 1998). Thus, an ecological investigation is, in general, contingent upon the window through which the system is viewed (Levin 1992).

On the other hand, an ecosystem resilience analysis ought to be a tool in order to address problems within environmental management. Therefore, the window through which the system is viewed (Levin 1992) is the purpose of the scientist to provide knowledge with respect to SD. The goal of investigation is to create and apply knowledge in support of decision making for sustainable development (Clark & Dickson 2003). Typically, these issues require a relatively large scale, i.e. a landscape scale (e.g. Carpenter & Cottingham 2002, Gunderson & Walters 2002, Jansson & Jansson 2002, Lugo et al. 2002, McClanahan et al. 2002, Walker 2002, Allison & Hobbs 2004).

We have explored the importance of slow, controlling variables for an ecosystem resilience analysis. Hence, being aware that dynamics occur at every scale and cross-scale, respectively, and different processes are likely to be important on different scales, we can conclude that the scale of an ecosystem resilience analysis that provides knowledge with respect to SD, must be at least at a level that takes into account the slow variables that drive ecosystem behaviour. Levels below and above have to be taken into consideration if they influence the behaviour of these slow variables. Therefore, there is something like a most appropriate scale for an ecosystem resilience analysis.

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 1995, 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, Colding & Folke 2003). 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) is reduced and, thus, ecosystem resilience mechanisms are lost.

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 ecosystem 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 ecosystem resilience in ecosystems. Gunderson et al. (2002b) 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. 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 functions such as productivity (Levin et al. 1998, Folke et al. 2002).

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).

The pathology has the following features (Holling 2003): (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 perverse[110] subsidies. (4) The relevant ecosystems gradually lose ecosystem resilience to become fragile and vulnerable and more

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[110] Perverse in the sense of that it threatens the functioning of the system in the long-term.

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.

In order to avoid this pathology decision-makers that are concerned with environmental issues can decide to manage for ecosystem resilience. The model of complex adaptive system - the *Panarchy* - includes the notion that these systems are inherently *unpredictable* (Gunderson, Holling & Light 1995). More particular, referring to the adaptive cycle, there are phases (r and K) when the dynamics are rather predictable, and there are phases (α and Ω) when they are highly unpredictable (Gunderson & Holling 2002). Gunderson (2003) suggests that nonlinear interactions among multiple variables, scale invariant processes, emergent properties from self-organization and other factors all contribute to unpredictability. Surprises are *inevitable*, and thus policies must always be adaptive. Therefore, within adaptive management, the existence of uncertainty and surprise are an accepted part of development, and management actions evolve to cope with their effects by spreading risks through diversification of both resource use patterns and alternative activities.

Adaptive environmental management tries to preserve or increase the ecosystem resilience of ecosystems since this is considered as a precondition for the capacity of the social-ecological system (SES) to respond to and shape change (Peterson 2002), i.e. to increase its *adaptive capacity* (Folke, Colding & Berkes 2003). The objective is to examine “ways of *building resilience* to enhance the capacity to deal with change and surprise” (Berkes, Colding & Folke 2003, 22), indeed, to “actively strengthen and enhance the capacity of the biosphere, from local to global levels, to support and sustain social and economic development” (Folke, Colding & Berkes 2003, 379).

In this respect, Berkes, Folke & Colding (1998) and Berkes, Colding & Folke (2003) identify three measures of an adaptive environmental management approach for building ecosystem resilience into SESs.

First, ecological knowledge is of high value. Along with traditional Western resource science *traditional ecological knowledge*, 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 (including humans) with one another and with their environment” (Berkes 1999, 8), is considered to be useful for adaptive environmental management. Traditional practices appear to have certain similarities and parallels to the theory of complex systems with their emphasis on non-linear relationships, threshold effects, multiple equlibria, the existence of several stability basins, disturbance and their focus on the back-loop, i.e. the release (Ω) and reorganization (α) phases of the adaptive cycle. This point is crucial. If traditional practices are similar to insights of modern ecological theory, this traditional knowledge can be used for a sound environmental management.

Second, a management approach for ecosystem resilience takes into account the ecosystem resilience mechanisms that are seen to be responsible for this emergent stability property on the ecosystem level.

For instance, various authors propose that *functional diversity* increases the performance of the community as a whole, while *response diversity* enables the community to keep performing in the same complementary way in the face of stresses and disturbances (Mooney et al. 1996, Elmqvist et al. 2003). *Response diversity* is likely to lead to an increased *functional compensation* within communities with respect to a given ecosystem function in the face of disturbance regimes. It might be difficult, however, to predict which species will account for functional compensation. To identify these redundant and compensating species would be of high value for an ecosystem management. Ecological redundancy might be responsible for the delay in the collapse of kelp forests in Southern California compared with Alaska after extirpation of Sea Otters (Jackson et al. 2001). Similarly, the recovery rate of corals after disturbance in the low-diversity Caribbean has been slower than recovery in the high-diversity Indo-Pacific. This might be weak evidence that species diversity increases the capacity of reefs to tolerate and recover from disturbance (MacClahanan et al. 2002).

Another ecosystem resilience mechanism, namely *ecological memory* on all levels (genetic, species, landscape) consisting of biotic legacies (Franklin & MacMahon 2000), mobile links (Lundberg & Moberg 2003) and support areas (Elmqvist et al. 2003) buffers against large-scale disturbances and fosters reorganization processes. This aspect of biodiversity may play an important role in the reorganization and renewal process following disturbance (Folke, Colding & Berkes 2003).

Bengtsson et al. (2003) point to the relevance of ecological memory, biodiversity and, thus, spatial resilience for modern intensively managed landscapes. Dynamic *insurance reserves* that secure (ecosystem) resilience in managed landscapes are both temporally and spatially flexible, *ecological fallows*, i.e. areas set aside for natural, or low-intensity managed succession immediately after a disturbance event, *ephemeral reserves* aimed at preserving species in the early succession or exploitation phases that otherwise may be threatened by large-scale intensive management and *mid-succession reserves* are considered to be decisive for the reorganization and renewal of whole landscapes.

Considering these reflections adaptive management nurtures biodiversity in order to ensure the performance of ecosystem functioning and to buffer against disturbances. Biodiversity is considered to be a prerequisite for the ecosystem resilience of natural systems (Deutsch, Folke & Skanberg 2003).

The third key feature of an adaptive management approach to strengthen ecosystem resilience is to consider small-scale, natural disturbances as *intrinsic* part in the internal dynamics of ecosystems (Holling 1986, Folke, Berkes & Colding 1998, Gunderson & Holling 2002, Bengtsson et al. 2003, Berkes & Folke 2003, Colding, Elmqvist & Olsson 2003). In order to ensure the ecosystem resilience of the structural and functional performance of ecosystems on a larger scale (e.g. landscape, 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 basins of attraction. The aim of adaptive management, therefore, is to prevent the build-up of large-scale crisis (Folke, Berkes & Colding 1998). This type of management is termed *backloop management* since it indirectly considers the release-reorganization phases of the adaptive cycle (Colding, Elmqvist & Olsson 2003). A management approach that suppresses disturbance is regarded as unsustainable (Berkes, Colding & Folke 2003).

Local resource users may actively create small-scale disturbances in the landscape. Many traditional societies nurture sources of ecosystem renewal by creating small-scale disturbances (Folke, Berkes & Colding 1998). By mimicking fine-scale natural disturbances, these practices help avoid the accumulation of disturbance that move across scales further up in the *Panarchy* (Berkes & Folke 2002).

Building on these three fundamental measures for adaptive environmental management, people can decide to increase ecosystem resilience by building up or allow ecosystem resilience mechanisms.

Moreover, the identification and monitoring of slow variables is one of the possibilities to estimate ecosystem resilience and is seen as crucial for a sound environmental management approach (Carpenter et al. 2001, Folke et al. 2002).

As we have seen, some authors try to estimate the amount of ecosystem resilience by simple models of two variables and bifurcation curves. However, ecological managers often do not have simple models of their systems and, thus, methods of estimating ecosystem resilience that do not depend on the construction of system models would greatly facilitate the application of ecosystem resilience-based management. Referring to this problem, Peterson (2002) suggests methods termed *probabilistic resilience* and *cross-scale edge* which represent a simple measure and tool for prediction of ecosystem resilience. These methods provide another possibility to operationalize ecosystem resilience.

All these abilities are summarized in the concept of *adaptive capacity* or *adaptability*, which is defined as the capacity of a SES to adapt to slower dynamics (Gunderson 2003) or, rather general, as the capacity of an SES to respond to and shape change (Peterson 2002).

Let us dare to attempt a synthesis. Ecological resilience appears to be a concept that is theoretically well-founded. As one of the three fundamental stability properties it represents a holistic and qualitative concept. Its presuppositions (a model of complex adaptive systems, alternative stable regimes) reflect insights of modern ecological science and numerous case studies from a broad range of ecosystem types. During its scientific history the concept has extended in its meaning which is reflected by the corresponding definitions.

The concept of ecological resilience refers to slow, underlying variables that determine the location of an ecosystem's regime within a heuristic stability landscape. Slow variables are finally responsible for a regime's shift to another basin of attraction. If one accepts this notion of ecosystem behaviour, the identification of slow variables represents, among others, an important possibility to estimate and model different aspects of ecosystem resilience (latitude, resistance, precariousness, panarchy). The estimation of slow controlling variables and their connection to fast variables for a broad range of ecosystem types is, in my view, one of the most important areas for further research within the debate about ecosystem resilience. It could help to identify the position of ecological thresholds and predict shifts of regimes to undesired basins of attraction. Hereby, indicators for slow variables of specific regimes can play an important role.

Along with slow variables, the identification of an appropriate temporal and spatial scale, desired ecosystem services, and the disturbance regime of concern are essential parts of a detailed ecosystem resilience analysis.

Several mechanisms on multiple scales are considered to be responsible for the emergence of ecological resilience on a larger scale. Many of them are related to a comprehensive notion of biodiversity, such as ecological redundancy, response diversity and ecological memory. Small-scale disturbances are seen to be an intrinsic part of ecosystem dynamics and sources for renewal and recovery which represents a fundamental characteristic of ecosystem resilience. These mechanisms can be used within an adaptive approach to environmental management that nurtures biodiversity, allows small-scale disturbances, and reacts flexible to unpredictable changes occuring within ecosystem dynamics.

Therefore, one can conclude that, in general, the estimation and operationalization of ecosystem resilience is possible even though it might be problematic if not impossible to identify the precise position within a stability landscape. Due to the additional possibility to implement mechanisms that are responsible for the stability property, the concept of ecological resilience is highly relevant for environmental management and nature conservation issues on a larger landscape scale. In order to take into account normative claims of a Theory of Sustainable Development, it can be used as a tool for keeping (critical) natural capital intact.

Appendix

a) Deutsche Zusammenfassung

Innerhalb des ‚argumentativen Raumes' des Nachhaltigkeitsdiskurses, wird ökologische Resilienz von einer Vielzahl von Autoren als eines der wichtigsten Themen für zukünftige Forschungsbemühungen angesehen (Perrings et al. 1995, Folke, Holling & Perrings 1996, Levin et al. 1998, Perrings 2002). Anerkannte Ökologen und Ökonomen betrachten ökologische Resilienz als einen nützlichen ‚Indikator' für ökologische Nachhaltigkeit und folgern, dass ökonomische Entwicklung nur dann als nachhaltig bezeichnet werden kann, wenn die ökologischen Systeme, von denen sie abhängig ist, ökologisch resilient sind (Arrow et al. 1995). Ferner wird das Konzept der ökologischen Resilienz als ein nützliches Werkzeug angesehen, um Probleme im Natur- und Ressourcenmanagement zu lösen (Holling & Meffe 1996, Gunderson & Holling 2002). Einige Autoren benutzen ökologische Resilienz als Leitlinie für eine Theorie Nachhaltiger Entwicklung (zB Ott 2001). Innerhalb der deutschen Debatte ist die theoretische Relevanz der ökologischen Resilienz unumstritten, die Möglichkeit der Operationalisierung wird jedoch in Frage gestellt (Ott 2001, Kopfmüller et al. 2001).

Im Zuge der Untersuchungen zur ökologischen Resilienz kam es zu einem regen wissenschaftlichen Diskurs und zur Gründung der *Resilience Alliance*, einer Forschungsgruppe aus Ökonomen, Ökologen, Sozialwissenschaftlern und Mathematikern, die sich in einem 5-Jahres Projekt - dem *Resilience Project* - mit Fragen der ökologischen Resilienz auseinander setzten.

Diese Diplomarbeit besteht aus einem Überblick der relevanten Artikel und Bücher der Resilienz-Debatte. Das Ziel der Arbeit ist es, (1) die theoretischen Grundlagen des Konzeptes der ökologischen Resilienz darzustellen, was deren Hintergrundtheorie umfasst, sowie (2) die Möglichkeiten zu untersuchen, wie ökologische Resilienz operationalisiert werden kann, und, im allgemeinen, deren Relevanz innerhalb einer Theorie Nachhaltiger Entwicklung zu erhellen.

Innerhalb der Nachhaltigkeitsdebatte wird die ökologische Resilienz im Zusammenhang mit einigen häufig genannten Konzepten diskutiert.

Ökologische Resilienz wird mit der Vorstellung von Grenzen des Wachstums und dem Konzept der Tragekapazität in Verbindung gebracht (Folke et al. 1994, Arrow et al. 1995, Perrings et al. 1995b, Seidl & Tisdell 1999, Perrings 2002, Ekins, Folke & deGroot 2003). Grenzen des Wachstums beziehen sich auf Tragekapazitäten von Ökosystemen, bestimmtes ökonomisches Wachstum zu ermöglichen. Die Tragekapazität ist wiederum abhängig von der ökologischen Resilienz der Funktion ökologischer Systeme (ähnlich Folke et al. 1994).

Darüberhinaus kann das Konzept der ökologischen Resilienz als Werkzeug angesehen werden, das eher unpräzise Konzept des kritischen Naturkapitals, welches innerhalb der ökologischen Ökonomie eine Rolle spielt, zu konkretisieren (Deutsch, Folke & Skanberg 2003, Ekins 2003, Ekins & Simon 2003). Dabei stellt die ökologische Resilienz eine Methodik dar, mit der die Aufrechterhaltung von essentiellen Ökosystemfunktionen untersucht werden kann (Deutsch, Folke & Skanberg 2003).

Die Unterscheidung von schwacher und starker Nachhaltigkeit spiegelt eine der Hauptkontroversen innerhalb der Nachhaltigkeitsdebatte wider (Daly 1996, Faucheux & O'Connor 1998, Ott 2001, 2003, Neumayer 2003). Die Konzeption starker Nachhaltigkeit stellt die Voraussetzung für die politische und moralische Relevanz der ökologischen Resilienz dar.

In der wissenschaftlich-ökologischen Diskussion über Stabilitätsaussagen wurden mehr als 70 Stabilitätskonzepte und 163 Definitionen vorgebracht (Grimm, Schmidt & Wissel 1992). Gemäß begrifflicher Klarheit, muss die Bedeutung dieser Stabilitätseigenschaften unterschieden werden. Untersuchungen der einschlägigen Literatur zufolge ergeben sich nur drei fundamental unterschiedliche Stabilitätseigenschaften, und zwar *constancy, resilience*, und *persistence* (Grimm & Wissel 1997) bzw. *constancy, resilience*, und *robustness* (Hansson & Helgesson 2003). Constancy wird definiert als "staying essentially unchanged", resilience als "returning to a reference state (or dynamic) after a temporary disturbance" und persistence als "persistence through time of an ecological system" (Grimm & Wissel 1997). Persistence ist dabei eine holistische und qualitative Stabilitätskategorie, da sie sich per definitionem auf ganze Ökosysteme bezieht und auf die Frage ob das gesamte Set an *variables of interest* in einem bestimmten Zustand bleibt. „Stabilität" ist damit eine Meta-Kategorie für die drei fundamentalen Stabilitätseigenschaften (Loreau et al. 2002).

In der originär-ökologischen Bedeutung wurde *resilience* auf zwei unterschiedliche Arten definiert (Holling 1986, Holling et al. 1999, Gunderson & Holling 2002).

Die erste „klassische“ Definition bezieht sich auf Stabilitätsphänomene in der Nähe eines *steady states*, wobei die Rate und Geschwindigkeit der Rückkehr zu den Ausgangsbedingungen dieses regimes als Maß für Stabilität bestimmt werden (deAngelis 1980, Pimm 1984, Tilman & Downing 1994, WBGU 2000, Lugo et al. 2002). Die langsame Dynamik in der Nähe eines *global equilibrium* steht im Mittelpunkt des Interesses (Ludwig, Walker & Holling 1997, 2002).

Die zweite Definition von Resilienz untersucht die Stabilitätsbedingungen in der Region eines *local equilibrium* das zwei alternative *basins of attraction* voneinander trennt (Ludwig, Walker & Holling 1997, 2002), wobei bestimmte Störungen ein System in eine andere basin of attraction verschieben können, die durch ein anderes Set an Zustandsvariablen und eine andere Ökosystem-Struktur gekennzeichnet ist (Holling 1973, 2001, Gunderson 2000, Gunderson & Pritchard Jr. 2002, Holling & Gunderson 2002, Walker et al. 2002, 2004).

Diese beiden Definitionen spiegeln die Betonung zweier unterschiedlicher Stabilitätseigenschaften ökologischer Systeme wider, und zwar *resilience* oder *persistence* (Grimm & Wissel 1997) bzw. *resilience* oder *robustness* (Hansson & Helgesson 2003). Mit den Begriffen *engineering resilience* für resilience und *ecosystem* (oder: *ecological*) *resilience* für persistence hebt das Forschungs-Netzwerk *Resilience Alliance* diese Differenz hervor.

Das Stabilitätskonzept der *ecosystem resilience* (ökologische Resilienz), dem im folgenden das Hauptinteresse gilt, bezieht sich auf zwei grundlegende ökologische Konzepte, welche die *Hintergrundtheorie der ökologischen Resilienz* ausmachen.

Erstens hängt die ecosystem resilience mit einem heuristisches Konzept von komplexen, adaptiven Systemen zusammen, welches auf die *Resilience Alliance* zurückgeht. Demnach zeigt die Sukzession und Dynamik produktiver Ökosysteme auf dem *patch scale* ein episodisches Verhalten, welches durch die zwei klassischen Phasen exploitation (r) und conservation (K) mit Pionierarten und langsam akkumulierender Biomasse eingeleitet wird. Diese werden dann abgelöst von der Phase der *creative destruction* (Ω), in der die stark vernetzte Biomasse freigesetzt wird und der *reorganization* (α), in der sich neue Ökosystemstrukturen aufbauen.

Stellt die Ω Phase das Ende des Zyklus dar, repräsentiert die darauf folgende α Phase den Anfang des *adaptive cycle*. Die Phasen haben unterschiedliche Dauer, wobei der *adaptive cycle* langsam von der r zur K Phase übergeht, während jeweils die Übergänge von der K zur Ω zur α zur r Phase schnell vonstatten gehen (Holling 1986, Holling et al. 1999, Gunderson & Holling 2002).

Da Prozesse und Veränderungen in Ökosystemen auf allen Hierarchieebenen vorkommen, entwirft die *Resilience Alliance* ein Modell für Ökosysteme – die *Panarchy* - das davon ausgeht, dass ein Set von *adaptive cycles* auf den unterschiedlichen Ebenen existiert, zwischen denen charakteristische Interaktionen stattfinden. Für ein Verständnis der Abläufe am wichtigsten sind die Prozesse *Revolt* und *Remember* (Holling 2001, Holling et al. 2002, Gunderson & Pritchard 2002). *Revolt* bezeichnet einen Prozess, bei dem adaptive cycles auf kleinen, schnellen Hierarchieebenen, die sich in der Ω Phase befinden, eine Veränderung in adaptive cycles auf großen, langsamen Hierarchieebenen auslösen, die gerade die K Phase durchlaufen. Zum Beispiel können Waldbrände auf der kleinen Ebene des Blattes beginnen, um sich dann auf Bäume und ganze Waldstücke auszuweiten. Die Interaktion *Remember* kennzeichnet einen Vorgang, bei dem größere und langsamere Ebenen in der K Phase kleinere, schnellere Ebenen in der α Phase beeinflussen. Ein Beispiel ist die Beeinflussung der Sukzession von Waldlücken nach Windwürfen durch nebenstehende Bäume und deren Samen.

Vor dem Hintergrund dieser Vorstellung von Ökosystem-Dynamiken erscheint *ecosystem resilience* als eine quantitativ veränderbare Größe, die auf jeder Ebene der Hierarchie des Ökosystems vorkommt. Die ecosystem resilience ist hoch in der Ω und α Phase, klein in der r Phase und sehr klein in der K Phase. Die K-Phase stellt einen *accident-waiting-to-happen* dar und ist sehr anfällig für Störungen bis das Ökosystem wieder in die creative destruction- und reorganization-Phasen übergeht (Holling 2001). Dieser *back loop* (Ω und α Phase) - unterschieden von dem *front loop* (r und K Phase) - wird innerhalb der Ökosystemdynamik als Motor für Erneuerung angesehen.

Der *extended keystone hypothesis* zufolge (aufgestellt von Holling 1992) wird die hierarchische Struktur von Ökosystemen in erster Linie von einem kleinen Set von planzen- und tier-physiologischen sowie abiotischen Prozessen reguliert, die als essentielle Variablen oder *key variables* bezeichnet werden. Die Komplexität adaptiver Systeme kann zurückgeführt werden auf Wechselwirkungen zwischen drei bis fünf Variablen-Sets, die jeweils auf einem qualitativ unterschiedlichen zeitlich und räumlichen Maßstab operieren. Dabei wird die Ökosystemdynamik von langsamen Variablen auf einem großen und langsamen Maßstab kontrolliert (Holling et al. 1999, Holling et al. 2002).

Das zweite Konzept der Hintergrundtheorie des ecosystem resilience-Konzeptes ist das Konzept der *alternative stable regimes*, welches besagt, dass ein Ökosystem - abhängig von dem Zustand der langsamen Variablen - in mehreren stabilen *regimes* vorkommen kann. Zahlreiche Studien bekräftigen die Hypothese einer Existenz von *alternative stable regimes* im Falle von Seen (Scheffer et al. 2001), marinen Systemen (Done 1991) und Ozeanen (Hare & Mantua 2000), Wüsten (Foley et al. 2003) sowie rangelands (Perrings and Walker 1997, Janssen et al. 2004), woodlands (Dublin, Sinclair & McGlade 1990) und Wäldern (Peterson 2002, Peterson et al. 2002).

Die jeweiligen stabilen *regimes* liegen innerhalb alternativer *basins of attraction*, welche Regionen im *state space* (d.h. die Zustandsvariablen und ihre Wechselwirkungen) darstellen, in denen das System danach strebt, in ähnlicher Struktur und Funktion fortzubestehen (Walker et al. 2004). Diese Regionen sind dabei nicht statisch, sondern kontrahieren und extrahieren als Reaktion auf Veränderungen in den langsamen Variablen (Holling 1986, Walker et al. 2004). Ist ein Störungsregime stark genug, die stabilisierenden Mechanismen eines *basins of attraction* zu überwinden, kommt es zu einem *regime shift*. In diesem Fall werden bestimmte kritische Zustandswerte kontrollierender Variablen überschritten, der Zustand und das Ausmaß interner *feedbacks* ändert sich (Walker & Meyers 2004), worauf sich nun das Ökosystem in einer anderen Konfiguration reorganisiert, die durch andere kontrollierende langsame Variablen und Prozesse bestimmt ist (Holling & Gunderson 2002). Dieser Übergangspunkt zwischen zwei alternativen basins of attraction eines Systems (die kritischen Werte der Zustandsvariablen) wird als ökologischer Schwellenwert (*ecological threshold*) bezeichnet (Muradian 2001, Walker & Meyers 2004).

Die alternativen basins of attraction und die Schwellenwerte werden bildlich und heuristisch als Senken und Schwellen in einer Stabilitätslandschaft (*stability landscape*) dargestellt (Walker et al. 2004), wobei langsame Variablen dafür verantwortlich sind, in welcher *basin of attraction* sich das Ökosystem befindet. Die Stabilitätslandschaft gibt somit die langsamen Variablen topographisch wieder, die

die Basis für das Verhalten der schnellen Variablen darstellen (Jansson & Jansson 2002, Gunderson & Walters 2002).

Im Bild der Stabilitätslandschaft wird die ökologische Resilienz als die Größe des *basin of attraction* (der Senken) (Holling 1973, Scheffer & Carpenter 2003), oder als die Fähigkeit, in einer *basin of attraction* zu verbleiben (Walker et al. 2004), aufgefasst. Walker et al. (2004) unterscheiden zwischen vier grundlegenden Aspekten der ökologischen Resilienz. (1) *Latitude* wird definiert als „the maximum amount the system can be changed before losing its ability to recover", was der Länge des basin of attraction entspricht. (2) *Resistance* wird spezifiziert als "the ease or difficulty of changing the system", was sich auf die Tiefe des basin bezieht. (3) *Precariousness* stellt den momentanen Zustand des Systems dar und wie nah dieser einem ökologischen Schwellenwert ist. Der vierte Aspekt (4) *panarchy* drückt aus, inwiefern die drei vorangehenden Aspekte durch Ökosystemhierarchieebenen auf Maßstäben ober- und unterhalb beeinflusst werden. Mittels dieser bildhaften Vorstellung wird ersichtlich, dass das Konzept der ökologischen Resilienz eine holistische und qualitative Stabilitätskategorie darstellt (Grimm et al. 1999), was der *persistence sensu* Grimm & Wissel (1997) entspricht.

Für die Emergenz der ökologischen Resilienz werden mehrere Mechanismen auf mindestens fünf Maßstäben (Lugo et al. 2002) als verantwortlich angesehen, die unterschiedlichen biophysikalischen Komponenten der ökologischen Resilienz entsprechen. Hierbei wird eine umfassende Definition von Biodiversität (Gene, Populationen, Spezies, Landschaft) (e.g. Lincoln, Boxshall & Clark 1998) sowie eine funktionelle Ökosystemvorstellung zugrunde gelegt.

Der erste Resilienz-Mechanismus wird als ökologische Redundanz (*ecological redundancy*) bezeichnet. Ökologische Redundanz liegt vor, wenn mehr als eine Art in einer genau definierten funktionellen Gruppe (*functional group*) vorliegt (Walker 1992, 1995, Gitay et al. 1996). Funktionelle Gruppen mit nur einer oder wenigen Arten werden als *keystone functional group* bezeichnet. Dabei verlagert sich der Schwerpunkt der Naturschutzbemühungen auf Arten in funktionellen Gruppen mit niedriger ökologischer Redundanz, da der Verlust dieser Arten zu einem Verlust ganzer Ökosystemfunktionen führen würde (Walker 1992, 1995, Mooney et al. 1996, WBGU 2000).

Umfasst eine funktionelle Gruppe mehr als eine Art, so können die Reaktionen dieser Arten auf Störungen unterschiedlich sein. Diese Variabilität der

Reaktionsfähigkeit wird als *response diversity* bezeichnet und definiert als „the diversity of responses to environmental change among species that contribute to the same ecosystem function” (Elmquist et al. 2003). Reagieren mehrere Arten derselben funktionellen Gruppe unterschiedlich auf Störungen, steigt die Wahrscheinlichkeit, dass diese Gruppe ihre Funktion trotz Störungen aufrecht erhalten kann. Durch strukturelle Änderung der Lebensgemeinschaften, welche die Ökosystemprozessraten stabilisiert, findet eine funktionelle Kompensation statt (Frost et al. 1995, Tilman 1996, Carpenter & Cottingham 2002). Die response diversity und deren Kompensationsleistungen auf Ökosystemebene können eine entscheidende Komponente der ökologischen Resilienz sein – als Pufferkapazität gegen Störungen (Naeem 1998, Holling et al. 1995, Yachi & Loreau 1999, Elmquist et al. 2003). In diesem Sinne, kann jede Spezies entscheidend sein und zur ökologischen Resilienz beitragen (Gunderson 2000), was als *insurance hypothesis* bezeichnet wird (Yachi & Loreau 1999).

Die ökologische Redundanz führt zu einer kompensierenden Überlappung bezüglich ökologischer Funktion auf demselben Maßstab (zB unterschiedliche Pflanzenarten) die zu einer erhöhten persistence des Ökosystems gegenüber Störungen führt. Dieser Prozess wird als *within-scale resilience* benannt (Peterson, Allen & Holling 1998, Holling et al. 2002, Gunderson et al. 2002). Ökosysteme sind jedoch hierarchisch und in einer „klumpigen“ Weise über mehrere Ebenen hinweg strukturiert (Holling 1992, Holling 2001, Holling & Gunderson 2002). Ökologische Prozesse, wie zB Samenverbreitung, können auf mehreren Ebenen stattfinden (Holling et al. 2002, Gunderson et al. 2002). Die response diversity, die über mehrere Ebenen hinweg besteht, führt zu einer erhöhten persistence auf Ökosystemebene, was als *(a)cross-scale resilience* beizeichnet wird (Peterson, Allen & Holling 1998, Holling et al. 2002, Gunderson et al. 2002). Somit ergänzt *within-scale resilience across-scale resilience*, was in dem Konzept der *imbricated resilience* zusammengefasst wird (Holling et al. 2002). In diesem Sinne entsteht ökologische Resilienz aufgrund von überlappender ökologischer Funktion auf derselben Ebene und Verstärkung dieser Funktion über mehrere Ökosystemebenen hinweg als ein Produkt funktioneller Diversität und ökologischer Redundanz (Peterson, Allen & Holling 1998).

Die nächste grundlegende Komponente der ökologischen Resilienz stellt das *ecological memory* dar, welches definiert wird als “the 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).

Dabei umfasst *internal memory biological legacies*, welche wiederum Organismen einschließen, die Störungen überleben, sowie biologische Strukturen, die als Ausgangsherde für Regeneration dienen und neuen Arten erlauben einzuwandern (Franklin & MacMahon 2000). Internal memory wird begrenzt durch *assembly rules* (zB. Fazilitation, Konkurrenz, trophische Wechselwirkungen), die bestimmen welche Arten trotz Störungen proliferieren können.

External memory setzt sich aus Quellen-Habitaten von kolonisierenden Arten zusammen und wird durch die Permeabilität der Matrix zwischen den kolonisierten patches beschränkt (Bengtsson et al. 2003). Diese Pufferkapazität ist abhängig von Regionen in der Nachbarschaft des kolonisierten patches und wird somit *spatial resilience* genannt (Nyström & Folke 2001, Bengtsson et al. 2003). Wenn diese Quellenhabitate heterogen über die Landschaft verteilt sind und ein Austausch zwischen ihnen stattfindet, besitzt diese Region eine erhöhte ökologische Resilienz gegenüber bestimmten Störungen (Walker 2002, van de Koppel & Rietkerk 2004). Es wird somit zwischen drei grundlegenden und miteinander interagierenden Komponenten von ecological memory unterschieden, die zu einer erhöhten ökologischen Resilienz führen können (Nyström & Folke 2001): biotic legacies, Quellenhabitate, und *mobile link species*. Letztere sind definiert als “organisms, which support essential functions by connecting areas to one another and contribute to ecosystem resilience” (Lundberg & Moberg 2003, 87).

Die relative Bedeutung von *internal memory* auf der einen Seite und *external memory* auf der anderen, war in der Ökologie lange umstritten. Die beiden Komponenten schließen sich jedoch grundsätzlich nicht aus, und ihre Bedeutung hängt von der Art der Störung und der Struktur der Landschaft ab (Bengtsson et al. 2003).

Ein weiterer grundlegender Mechanismus, der zu einer erhöhten ökologischen Resilienz führt, wird ersichtlich im Lichte unseres Modells zur Ökosystemdynamik. Der *back loop* des *adaptive cycles* (Ω and α Phasen) und die *Revolt-interaction*, die mindestens zwei Ebenen der Ökosystemhierarchie umgreift, spiegeln die Bedeutung von Erneuerung und Variabilität auf jeder Ebene der Hierarchie wider. Die sich wiederholenden Zyklen des *adaptive cycles* und deren

Interaktionen über mehrere Ebenen hinweg, führen zu einer neuen Artenkombination, die besser an die herrschenden Umweltbedingungen angepasst, und deshalb ökologisch resilienter gegenüber einer fluktuierenden Umwelt ist. Innerhalb dieses Ökosystemmodells werden Störungen einer niedrigen Ebene auf einem kleinen und schnellen Maßstab als integraler Bestandteil der Ökosystemdynamik betrachtet (Holling 1986, Gunderson & Holling 2002, Bengtsson et al. 2003).

Der Verlust von ökologischer Resilienz auf einem kleinen und schnellen Maßstab ist damit ein unvermeidlicher Teil der Ökosystemdynamik. Ein System, welches ökologische Resilienz jedoch auf solchen kleinen Maßstäben verliert, die für ein Ökosystemmanagement handhabbarer sind, erweist sich als „stabiler“ als ein System, welches ökologische Resilienz auf größeren Maßstäben verliert (Walker et al. 2004). In diesem Sinne wird ökologische Resilienz durch die Zerstörung und Erneuerung von Systemen auf kleinem Maßstab generiert (Gunderson 2000). Dabei können Störungen zu einer erhöhten Artenvielfalt und landschaftlichen Heterogenität führen. Artenvielfalt führt wiederum zu ökologischer Redundanz bezüglich Ökosystemfunktionen und diese stellt eine erhöhte ökologische Resilienz gegenüber Störungen (auf einem größeren Maßstab) sicher (Jentsch, Beierkuhnlein & White 2002). Dieser Vorgang kann als *Erneuerungsaspekt der ökologischen Resilienz* bezeichnet werden (ähnlich Bengtsson et al. 2003).

Diese Sichtweise setzt voraus, dass Störungen auf kleinen Maßstäben zu *persistence* auf größeren Maßstäben führt (Jentsch, Beierkuhnlein & White 2002). Die *persistence* ist dabei abhängig von dem Verhältnis der gestörten Fläche zur Landschaftsfläche und dem Verhältnis der Störungshäufigkeit zur Regenerationszeit (Turner et al. 1993). Je kleiner die gestörte Fläche im Verhältnis zur Regenerationszeit, desto größer ist die Möglichkeit von *persistence* in allen *patches*.

Im allgemeinen kommen Resilienzmechanismen auf mindestens fünf Ebenen der Ökosystemhierarchie vor (Lugo et al. 2002). Erstens auf Individuen-Ebene als Teil der physiologischen Reaktion auf die Umwelt, wie zB hohe Reproduktionsraten, hohe Mobilität, phenotypische Plastizität, flexibles Fressverhalten und physiologische Toleranz (Grimm et al 1999). Zweitens auf der Ebene der Lebensgemeinschaften und deren Reaktion auf Umweltbedingungen, wie zB funktionelle Diversität und response diversity. Drittens spielt der Effekt von biotic legacies eine Rolle. Viertens können andere Hierarchieebenen einen verstärkenden

Effekt haben, wie zB external memory. Und fünftens können interne feedback-Mechanismen auf Ökosystemebene zu einer erhöhten ökologischen Resilienz führen, wie zB negative feedbacks von Nährstoffspeichern.

Der Einfluss abiotischer Faktoren auf die Struktur und Funktion von Ökosystemen ist oft sehr groß. In vielen Ökosystemen stellen abiotische Variablen die langsamen Variablen dar, welche die Ökosystemfunktionen kontrollieren. Der Verlust von Resilienz-Mechanismen, wie zB ökologischer Redundanz oder ecological memory resultiert in dem Verlust von ökologischer Resilienz und erzeugt Ökosysteme, die empfindlicher gegenüber Störungen sind. Doch letztendlich, ist es dann die Veränderung der langsamen Variablen, die den Verlust von ökologischer Resilienz anzeigen (ähnlich in Gunderson & Walters 2002, 177). Diese langsamen Variablen sind dabei oft abiotische Faktoren.

Die Diskussion der Hintergrundtheorie, sowie der Resilienz-Mechanismen macht deutlich, dass die ökologische Resilienz ein Konzept ist, welches mehrere Bedeutungsebenen umfasst. Man kann mindestens zwischen sechs Ebenen unterscheiden.

Schon 1973 definierte Holling (ökologische) Resilienz als “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). 29 Jahre später definierten Holling & Gunderson ökologische Resilienz sehr ähnlich als “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour” (Holling & Gunderson 2002, 4). Diese *originär-ökologischen Bedeutungen* stimmen mehr oder weniger mit *persistence sensu* Grimm & Wissel (1997) sowie *robustness sensu* Hansson & Helgesson (2003) überein.

Mit dem Fortschritt der ökologischen Wissenschaft und tieferen Einsichten in Konzepte, die auf die ökologische Resilienz bezogen sind, wurde das Konzept der ökologischen Resilienz in seiner Bedeutung verändert, wenn nicht erweitert. Dies schlägt sich auch in den Definitionen nieder. Walker et al. definieren ökologische Resilienz als “the potential of a system to remain in a particular configuration and to maintain its feedbacks and functions, (…) [which] involves the ability of the system to reorganize following disturbance-driven change” (Walker et al. 2002) und zwei Jahre später als “the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks" (Walker et al. 2004, 2). In dieser *erweitert-ökologischen Bedeutung*, umfasst das Konzept der ökologischen Resilienz drei kennzeichnende Aspekte: (1) the amount of disturbance a system can withstand and still retain the same controls of function and structure, (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). In meinen Augen erweitern die letzten beiden Aspekte die Bedeutung der originären Definition von Holling (1973), da ökologische Resilienz hier nicht mehr nur die Fähigkeit umfasst, Störungen zu absorbieren und in der selben *basin of attraction* zu verbleiben, sondern zudem die Fähigkeiten der Selbstorganisation und Erneuerung.

Im Zuge der Debatten haben einige Autoren den Bedarf einer empirisch klar bestimmbaren Definition erkannt. Ökologische Resilienz wird erhoben als Resilienz von was (?) und gegenüber was (?) (Carpenter et al. 2001). Diese *operationale Bedeutung* ist die Grundlage einer umfassenden Analyse der ökologischen Resilienz (Walker et al. 2002).

Wie wir bei der Diskussion von langsamen Variablen gesehen haben, müssen sich Aussagen über die ökologische Resilienz auf zugrunde liegende, die Ökosystemdynamik kontrollierende, langsame Variablen beziehen. In dieser *ökologisch-systemischen Bedeutung* ist ökologische Resilienz die Fähigkeit (die dafür verantwortlichen Mechanismen und Variablen) eines Ökosystems, trotz einer flukturierenden Umwelt, natürliche Dienstleistungen aufrecht zu erhalten (Folke et al. 2002).

Innerhalb eines Umweltmanagements und des Nachhaltigkeitsdiskurses werden nicht nur die natürlichen Teilsysteme von Ökosystemen untersucht, sondern vollständige Ökosysteme, welche das anthropogene System (zB Landnutzung) einschließen. Deshalb beziehen sich einige Autoren auf soziale Systeme und benutzen eine *soziologische Bedeutung* der ökologischen Resilienz. *Social resilience* wird dann definiert als "the ability of human communities to withstand external shocks to their social infrastructure, such as environmental variability or social, economic, and political upheaval" (Adger 2000, 347).

Außerdem wird das Konzept auch auf vollständige sozio-ökologische Systeme angewandt und zwar als "the capacity of ecosystems to sustain societal development and progress with essential ecosystem services" (Folke, Colding & Berkes 2003, 354), was als *sozio-ökologische Bedeutung* bezeichnet werden kann.

Alle sechs dieser Bedeutungen der ökologischen Resilienz kommen in der einschlägigen Literatur vor. Auffallend dabei ist dass, obwohl es drei fundamentale Stabilitätseigenschaften (constancy, resilience and persistence/ robustness) gibt, die meisten Autoren ihren Schwerpunkt auf die ökologische Resilienz (persistence/ robustness) legen.

Diese Schwerpunktsetzung hat sehr große Auswirkungen auf die Bewertung, das Verständnis und das Management sozio-ökologischer Systeme. *Engineering resilience* (*resilience*) betont Effizienz, Kontrolle, *constancy* und Vorhersagbarkeit. Ökologische Resilienz hingegen berücksichtigt eher *robustness*, Adaptivität, Variabilität und Unvorhersagbarkeit. Diese beiden Ansätze stellen grundlegend verschiedene Ansätze in der Wissenschaft dar (Berkes & Folke 1998, Gunderson & Holling 2002).

Manche Autoren behaupten, dass nur ein Schwerpunkt auf ökologischer Resilienz (persistence/ robustness) einer stark nachhaltigen Beziehung zwischen Menschen und Natur förderlich ist (Peterson, Allen & Holling 1998, Gunderson & Holling 2002). Untersuchungen zur *engineering resilience* konzentrieren sich auf kleinere Teile der Stabilitätslandschaft, solche zur ökologischen Resilienz auf ihre gesamten Konturen. Somit sind erstere nicht hinreichend für Fragestellungen, die für eine Nachhaltige Entwicklung oder ein Ökosystemmanagement relevant sind (Holling & Meffe 1996). Außerdem wird die Existenz von alternativen basins of attraction im Konzept der engineering resilience nicht berücksichtigt (Peterson, Allen & Holling 1998).

Die meisten Autoren betrachten diese Betonung der ökologischen Resilienz jedoch mehr als ein Paradigma als eine Wiederspiegelung der realen Tatsachen der Natur (Gunderson & Holling 2002, Gunderson & Pritchard 2002), d.h. andere Stabilitätseigenschaften (*constancy*, *resilience*) können für die Analyse von Ökosystemen sehr nützlich und relevant sein.

Man kann zB zeigen, dass die Bedeutung von verschiedenen Stabilitätseigenschaften vom räumlichen und zeitlichen Maßstab der Untersuchung abhängig sind (Grimm et al. 1999). Im Falle des Wattenmeeres in Deutschland, steigt die Bedeutung des Konzeptes der ökologischen Resilienz mit größeren und zeitlich längeren Maßstäben. Obwohl Untersuchungen zur *engineering resilience* viele Aspekte von Stabilitätslandschaften nicht berücksichtigen, können empirische Bestimmungen der Rückkehrzeit (und das ist das Maß für *engineering resilience*) nützlich sein, um Reaktionen verschiedener Systeme zu vergleichen (Carpenter & Cottingham 1997). Des weiteren verdeutlichen Modelle, dass für beide Stabilitätseigenschaften (ökologische Resilienz und *engineering resilience*) lange Rückkehrzeiten eine reduzierte Resilienz anzeigen (Ludwig, Waker & Holling 1997, 2002, similar Wissel 1984).

Außerdem beinhaltet die *erweitert-ökologische Bedeutung* von ökologischer Resilienz "a measure of the maximum amount the system can be changed before losing its ability to recover" (Walker et al. 2004), was als Maß für *engineering resilience* aufgefasst werden kann (Grimm & Wissel 1997). Somit kann man behaupten, dass das erweiterte Konzept der ökologischen Resilienz *sensu* Walker et al. (2004) das Konzept der engineering resilience in irgendeiner Form enthält. Dies könnte je nach Sichtweise entweder zu einer erneuten Verwirrung oder zu einem umfassenderen Stabilitätskonzept führen und ist dann schlecht zu heißen oder zu begrüßen.

Aus diesen Gründen kann man den Schwerpunkt auf ökologischer Resilienz als nützlich erachten, um die Bedeutung von alternativen *basins of attraction*, nicht-linearen Ökosystemprozessen und langsamen Variablen zu berücksichtigen, was die potentielle Veränderlichkeit der „Stabilität“ eines Systems anzeigt. Es kann helfen, eine ökologisch-wissenschaftliche Sichtweise, die sich zu sehr an „Stabilität“ und „Gleichgewicht“ hält, zu hinterfragen.

Die Betonung des Konzeptes der ökologischen Resilienz ist auf der anderen Seite nicht richtig, wenn man damit annimmt, dass dies das einzige Stabilitätskonzept ist, welches für die Untersuchung von ökologischen Vorgängen wichtig sein kann. Andere Stabilitätskonzepte können für bestimmte Ökosysteme und Maßstäbe hochrelevant sein.

Das Konzept der ökologischen Resilienz ist nicht nur von theoretischem Interesse. Die meisten Autoren verbinden mit dem Konzept die Hoffnung, Problemen gerecht zu werden, die sich im Bereich Naturschutz und Ökosystem- und Ressourcenmanagement stellen (Holling & Meffe 1996, Ott 2001, Kopfmüller et al. 2001). In diesem Zusammenhang ist zum einen fraglich ob es möglich ist, die ökologische Resilienz zu operationalisieren, d.h., ob die Möglichkeit besteht, für eine konkrete Situation intersubjektiv festzustellen, ob ein Begriff auf die empirische Realität zutrifft (*Operationalisierbarkeit*) (Jax 2002). Dabei wäre es wünschenswert, das Ausmaß der ökologischen Resilienz zu messen, oder wenn das nicht möglich ist, annähernd zu bestimmen. Zum anderen ist zu bedenken, ob und in welcher Weise Maßnahmen zur Aufrechterhaltung der ökologischen Resilienz für ein Ökosystemmanagement implementierbar sind.

Um der Forderung der Operationalisierbarkeit gerecht zu werden, entwickelten Carpenter et al. (2001) ein klares und messbares Konzept der ökologischen Resilienz (die *operationale Bedeutung*), welches die Grundlage einer umfassenden Resilienz-Analyse darstellt (Walker et al. 2002). Die Ziele einer solchen Analyse sind, (1) zu verhindern, dass ein Ökosystem aufgrund von Störungen in ein unerwünschtes basin of attraction übergeht und (2) die Komponenten zu erhalten und zu fördern, welche das System befähigen, sich nach einer Störung zu erneuern und zu reorganisieren (Walker et al. 2002).

Dabei ist zu berücksichtigen, dass, in der deskriptiven Bedeutung (originär-ökologische, erweitert-ökologische, ökologisch-systemische, operationale Bedeutungen) ökologische Resilienz nicht *per se* wünschenswert ist. Ökologische Resilienz kann wünschenswert sein oder nicht, das hängt von dem betreffenden *regime* ab. Eine degradierte Savanne, zum Beispiel, kann hochresilient, dabei aber auch unbrauchbar für Weidewirtschaft und deswegen unerwünscht sein (Carpenter et al. 2001, Carpenter & Cottingham 2002, Gunderson & Holling 2002, Walker et al. 2002).

Die Resilienz-Analyse umfasst fünf Schritte.

Zunächst muss das Konzept der ökologischen Resilienz genauer spezifiziert werden. Wie wir gesehen haben, kann man vier Aspekte der ökologischen Resilienz unterscheiden, nämlich *latitude*, *resistance*, *precariousness* und *panarchy* (Walker et al. 2004). Die Wissenschaftlerin muss als ersten Schritt angeben, welchen Aspekt der ökologischen Resilienz sie untersuchen möchte.

Den zweiten Schritt der Analyse bezeichnet man als *to-what part*. Dieser bezieht sich auf das Störungsregime, gegenüber dem das betreffende Ökosystem resilient sein soll. Die Reaktion des Ökosystems auf Störungen hängt zB stark von dem Typ, der Stärke und dem Maßstab der Störung ab (Pickett & White 1985, Frelich & Reich 1998, Romme et al. 1998, Turner & Dale 1998). Der to-what part spiegelt die prinzipielle Einsicht wider, dass eine umfassende ökologische Resilienz gegenüber allen vorkommenden Störungsregimes nicht erreichbar ist (Grimm & Wissel 1997, Walker & Abel 2002).

Im dritten Schritt gibt man den zeitlichen und räumlichen Maßstab an, auf dem die Untersuchung stattfindet, was den ersten Teil des *of-what parts* der Resilienz-Analyse darstellt. Im allgemeinen kann man Maßstab definieren als "a range of spatial and temporal frequencies" (Peterson, Allen & Holling 1998, 11). Diese Festsetzung des Maßstabs ist wichtig, weil ökologische Resilienz eines sozio-ökologischen Systems eines Zeitabschnitts oder eines bestimmten Gebietes auf Kosten eines anderen Zeitabschnitts oder eines anderen Gebietes erzielt werden kann (Carpenter et al. 2001, Folke et al. 2002). Dabei wird die gewählte Maßstabsebene von Ebenen darunter und darüber beeinflusst.

Der vierte Schritt ist die Bestimmung von gewünschten Ökosystem-Dienstleistungen. Daily (1997) definiert *ecosystem services* als "conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life". Verschiedene Dienstleistungen desselben Gebiets können konfligieren, wie zum Beispiel die landwirtschaftliche Produktion und die Erholungsfunktion von Seen innerhalb einer bestimmten Landschaft. Deshalb ist es vonnöten zu entscheiden, welche Dienstleistungen prioritär sind, d.h. was genau resilient gegenüber Störungen sein *soll*. Das Konzept der ökologischen Resilienz enthebt uns nicht der Reflektion darüber, welche Natur wir haben wollen oder brauchen (Honnefelder 1993). Dafür bedarf es einer ethischen Auseinandersetzung über unser moralisches Verhältnis gegenüber Naturentitäten (*Umweltethik*) (Krebs 1997, Ott & Gorke 2000) sowie naturschutzfachlicher Bewertungskonzeptionen, die Übereinkommen über gesellschaftliche Leitbilder einschließen (Eser & Potthast 1997, Wiegleb 1997).

Da sich das Konzept der ökologischen Resilienz immer auf zugrunde liegende langsame Variablen bezieht, besteht der sechste Schritt der Resilienz-Analyse in der Bestimmung dieser das Ökosystem kontrollierenden Variablen. Um die Bereitstellung von Ökosystemdienstleistungen (*functions-for humans*) längerfristig sicherzustellen, bedarf es der Aufrechterhaltung der langsamen Variablen, die dafür sorgen, dass ein regime in einer wünschenswerten basin of attraction bleibt (*functions-of natural capital*) und somit letztendlich ein Maß dafür sind, die Fähigkeit des Ökosystems anzuzeigen, Störungen zu absorbieren, zu erneuern und zu reorganisieren (Holling et al. 1995, Carpenter et al. 2001, Ekins 2003). Von diesem systemischen Blickwinkel ist ökologische Resilienz die Fähigkeit eines Ökosystems trotz Störungen, ökologische Dienstleistungen bereit zu stellen (ökologisch-systemische Bedeutung), was auf die Mechanismen verweist, die für diese Fähigkeit verantwortlich sind.

Will die Wissenschaftlerin ökologische Resilienz messen, begegnet sie einer prinzipiellen Schwierigkeit, da die Messung die großmaßstäbige Störung eines Systems bedeuten würde, was wiederum eigentlich verhindert werden soll (Peterson 2002). Deswegen schlagen manche Autoren vor, die ökologische Resilienz zu bestimmen, indem man die *basins of attraction* in einem Koordiantensystem aufträgt, das eine schnelle und langsame, kontrollierende Variable wiedergibt. Das Diagramm zeigt dann ein oberes und unteres stabiles Equilibrium, die durch ein unstabiles Equilibrium voneinander getrennt sind – das sogen. *bifurcation diagram* (Carpenter et al. 2001, Peterson et al. 2003). Im Modell, wird die ökologische Resilienz als die Größe des *basins of attraction* oder als der Unterschied des momentanen und des kritischen Wertes der langsamen Variable dargestellt, welche empirisch oder theoretisch bestimmt werden (Wissel 1984, Ludwig, Walker & Holling 1997, 2002). In der Praxis könnten die Werte der langsamen Variable erhoben werden. Es bleibt aber unklar, ob eine solche Bestimmung zB des kritischen Wertes wirklich durchführbar ist.

Die Vorhersagbarkeit eines solchen ökologischen Schwellenwertes wäre von großem Wert für ein adaptives Umweltmanagment. Ein geeignetes Mittel wäre es, Untersuchungsergebnisse eines Gebietes, das einen *regime shift* durchlaufen hat, mit Daten von ähnlichen Gebieten zu vergleichen, die sich noch in der ursprünglichen *basin of attraction* befinden (Walker and Meyers 2004). Auch wenn dies nicht möglich ist, kann die *Existenz* einer alternativen *basin of attraction* und damit die Gefahr eines *regime shifts* vorausgesagt werden (Foley et al. 2003). Indem die zugrunde liegenden, kontrollierenden Variablen und deren nicht-lineares Verhalten untersucht werden, könnten Ökologen in der Lage sein, zu prognostizieren, welche Regionen der Welt einem *regime shift* ausgesetzt sind und welche nicht (Scheffer & Carpenter 2003).

Die Festlegung von aussagekräftigen Indikatoren ist eine Möglichkeit, langsame Variablen zu untersuchen und deren Trends festzustellen. Bezüglich der langsamen Variablen sollten Indikatoren die interne Dynamik von Ökosystemen und deren ökologische Resilienz angemessen widerspiegeln.

Für die Ostsee, Deutschland, werden der Phosphatgehalt und das Verhältnis von Stickstoff zu Phosphat im Sediment sowie die organische Substanz als langsame, das Ökosystem kontrollierende Variablen angesehen (Jansson & Jansson 2002). Diese Variablen können erhoben und als Indikatoren für das Ökosystemregime verwendet werden.

Die Bestimmung der langsamen Variablen und deren Indikatoren ist der sechste und letzte Schritt der Resilienz-Analyse. Meiner Meinung nach ist dieser Schritt eine der drängendsten Aufgaben für weitere ökologische Untersuchungen im Rahmen des Konzeptes der ökologischen Resilienz. Die Bestimmung der, das Ökosystem kontrollierenden, Variablen für jeden in der Natur vorkommenden Ökosystemtyp könnte problematisch sein. Für aquatische Ökosysteme dagegen, scheint es relativ leicht zu sein, da Primärnährstoffe in diesen Ökosystemen eine große Rolle spielen. Dem Ökologen und Ökosystemmodellierer Volker Grimm zufolge führt dies zu der Debatte über verschiedenen Modellierungsansätze, die für Ökosysteme herangezogen werden. Grimm (1999) erachtet *bottom-up Modelle* für wichtig, sie sollten jedoch durch *state variable Modelle* ergänzt werden, um ein integratives Modell von Ökosystemen zu erhalten.

Manche Autoren behaupten, dass es in der Ökologie generell, und für die Resilienz-Analyse im besonderen, keinen „einzig richtigen Maßstab“ (*most appropriate scale*) geben kann, da verschiedene grundlegende Prozesse auf unterschiedlichen Maßstäben maßgebend sein können (Levin 1992). Außerdem scheinen Umweltprobleme nicht nur entweder auf einem kleinen oder großen Maßstab relevant zu sein, sondern über mehrere Maßstäbe hinweg (*cross-scale*) sowohl zeitlich als auch räumlich (Folke, Berkes & Colding 1998). Somit erscheint eine ökologische Untersuchung prinzipiell von der Perspektive des Beobachters und Wissenschaftlers abhängig zu sein (Levin 1992).

Eine Resilienz-Analyse soll ein Werkzeug sein, um Umweltprobleme besser lösen zu können. Deshalb enspricht die Perspektive, von der aus das Ökosystem betrachtet wird, der Absicht des Wissenschaftlers, Wissen bereit zu stellen, welches für das Ökosystemmanagement und eine Theorie Nachhaltiger Entwicklung relevant ist (Clark & Dickson 2003). Meist wird solches Wissen auf relativ großem Maßstab gebraucht (e.g. Carpenter & Cottingham 2002, Gunderson & Walters 2002, Jansson & Jansson 2002, Lugo et al. 2002, McClanahan et al. 2002, Walker 2002, Allison & Hobbs 2004).

Für eine Resilienz-Analyse ist die Bestimmung der langsamen, kontrollierenden Variablen ein wichtiger Schritt. Ist man sich im klaren, dass grundlegende Prozesse auf unterschiedlichen Ebenen wichtig sein und Ökosystemdynamiken auf jeder Ebene und über mehrere Ebenen hinweg vorkommen können, kann man feststellen, dass der Maßstab für eine Resilienz-Analyse, welche Wissen erzeugt, das für Ökosystemmanagement relevant ist, mindestens so groß sein muss, sodass die langsamen Variablen berücksichtigt werden, die die Ökosystemdynamik kontrollieren. Ebenen unter und über diesem Maßstab können relevant sein, sofern sie das Verhalten dieser langsamen Variablen beeinflussen. In diesem Sinne gibt es für eine Resilienz-Analyse einen „einzig richtigen Maßstab".

Die meisten Ansätze im Ökosystem- und Ressourcenmanagement haben versucht, die Natur zu kontrollieren, um ihre Produkte abzuernten, die Bedrohungen, die von ihr ausgehen, zu reduzieren, und vorhersagbare Ergebnisse für den kurzfristigen Nutzen der Menschheit zu liefern (Holling 1995, Holling & Meffe 1996). Der Ressourcenmanager versucht eine Zielressource (zB Holz- oder Fischbestand) zu kontrollieren, wobei die r und K Phasen des adaptive cycles betont werden, während die Ω und die α Phasen ignoriert werden (Holling & Gunderson 2002, Berkes, Colding & Folke 2003). Diese *command-and-control*-Ansätze gehen einher mit einer Reduzierung der strukturellen und funktionellen Variabilität sowohl zeitlich (zB Störungen auf kleinem Maßstab) als auch räumlich (zB ökologische Redundanz, ecological memory, mobile links, landschaftliche Heterogenität) und führen zu dem Verlust von Resilienz-Mechanismen.

In der Analogie der Stabilitätslandschaft gesprochen, schrumpft die *basin of attraction* und die Gefahr eines *regime shifts* ist somit erhöht. Eine Störung, die vorher von dem entsprechenden *regime* absorbiert werden konnte, wird zum Auslöser eines nicht-linearen *regime shifts*, der oft den Verlust von essentiellen Ökosystemfunktionen mit sich bringt (Levin et al. 1998, Folke et al. 2002). Diese, dann überraschenden, *regime shifts* sind die unausweichliche Konsequenz eines *command-and-control*-Managements, was als *pathology of natural resource management* bezeichnet wird (Holling 1995, Holling & Meffe 1996).

Um diese *regime shifts* zu vermeiden, können sich Entscheidungsträger, die sich mit Umweltfragen auseinandersetzen, entschließen, ihr Ökosystemmanagement mit dem Ziel einer Aufrechterhaltung oder sogar Erhöhung der ökologischen Resilienz auszuüben. Unser vorher besprochenes Modell der Ökosystemdynamik – die *Panarchy* – beinhaltet dabei die Vorstellung, dass diese Systeme inhärent unvorhersehbar sind, da der *adaptive cycle* Phasen (α and Ω) durchläuft, die unvorhersehbar sind (Gunderson, Holling & Light 1995, Gunderson & Holling 2002). Deswegen muss Ökosystemmanagement immer adaptiv sein, um auf Veränderungen reagieren zu können.

Adaptive environmental management versucht die ökologische Resilienz eines *regimes* zu erhalten oder zu erhöhen, da dies als Voraussetzung für die Fähigkeit eines sozio-ökologischen Systems angesehen wird, auf Veränderungen und Störungen zu reagieren, d.h. es soll die *adaptive capacity* des gesamten sozio-ökologischen Systems beibehalten oder erhöht werden (Peterson 2002, Folke, Colding & Berkes 2003). Hier werden drei Maßnahmen als grundlegend angesehen (Berkes, Folke & Colding 1998, Berkes, Colding & Folke 2003).

Von großem Wert ist erstens ökologisches Wissen. Neben der ökologischen Wissenschaft wird *traditional ecological knowledge*, definiert als “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 (including humans) with one another and with their environment” (Berkes 1999, 8), als nützlich angesehen für ein adaptives Umweltmanagement. Traditionelle Praktiken scheinen gewisse Ähnlichkeiten und Parallelen zu unserem Modell komplexer Systeme zu haben was in der Betonung von non-linearen Prozessen, Schwellenwerten, alternativen basins of attraction, der Relevanz von kleinflächigen Störungen und der Betonung des *back loops* des *adaptive cycles* deutlich wird.

Zweitens benutzt ein adaptives Management alle bekannten Resilienz-Mechanismen. So werden funktionelle Diversität und *response diversity* als wichtig angesehen, um essentielle Ökosystemfunktionen trotz Störungen aufrecht zu erhalten (Mooney et al. 1996, Elmqvist et al. 2003). Eine Bestimmung der redundanten Arten könnte sehr nützlich sein für ein Ökosystemmanagement, es ist jedoch schwierig, vorauszusagen, welche Arten jeweils für eine funktionelle Kompensierung nach einer bestimmten Störung sorgen werden. Des weiteren hilft *ecological memory*, Störungen zu absorbieren und erhöht die Fähigkeit zur Reorganisation (Elmqvist et al. 2003, Folke et al. 2003). Eine Voraussetzung für die Erhaltung der ökologischen Resilienz ist somit die Erhaltung und Förderung von Biodiversität auf allen Ebenen (Deutsch, Folke & Skanberg 2003).

Die dritte Maßnahme zur Aufrechterhaltung der ökologischen Resilienz ist es, kleinflächige Störungen zuzulassen und sie als integralen Bestandteil der

Ökosystemdynamik aufzufassen (Holling 1986, Folke, Berkes & Colding 1998, Gunderson & Holling 2002, Bengtsson et al. 2003, Berkes & Folke 2003, Colding, Elmqvist & Olsson 2003). Andernfalls können Störungen akkumulieren und ganze Landschaften oder Regionen zu einem *regime shift* veranlassen. Das Ziel eines *adaptive management* ist es somit solche großflächigen Krisen zu vermeiden (Folke, Berkes & Colding 1998), was als *backloop*-Management bezeichnet wird, weil dieser sich auf die Ω und α Phasen, und somit kleinflächige Störungen, bezieht (Colding, Elmqvist & Olsson 2003). Ein Umweltmanagement, das kleinflächige Störungen unterdrückt, ist somit nicht nachhaltig (Berkes, Colding & Folke 2003).

Aufbauend auf diesen drei grundlegenden Maßnahmen eines adaptive Managements können Entscheidungsträger ökologische Resilienz erhalten oder sogar erzeugen. Außerdem erlaubt es die Bestimmung der langsamen Variablen, zugrunde liegende Mechanismen zu berücksichtigen, die letztendlich dafür verantwortlich sind, ob ein regime in einer bestimmten basin of attraction verbleibt (Carpenter et al. 2001, Folke et al. 2002).

Diese Werkzeuge für einen neuen Ansatz im Umweltmanagment sind Teil der *adaptive capcity*, der Fähigkeit des sozio-ökologischen Systems, auf Störungen und Veränderungen zu reagieren (Peterson 2002).

Wagen wir eine Synthese. Das Konzept der ökologischen Resilienz kann als theoretisch fundiert bezeichnet werden. Die Voraussetzungen des Konzeptes (eine bestimmte Vorstellung von Ökosystemdynamik, die Existenz alternativer basins of attraction) spiegeln Einsichten der modernen Ökologie und zahlreicher Feldstudien in unterschiedlichen Ökosystemtypen wider. Dabei stellt die ökologische Resilienz eine von drei grundlegenden Stabilitätseigenschaften ökologischer Systeme dar. Im Laufe der Geschichte des Begriffs kam es zu einer Erweiterung der Bedeutung, was in den ensprechenden Definitionen deutlich wird.

Das Konzept der ökologischen Resilienz bezieht sich auf langsame, zu Grunde liegende Variablen. Diese bestimmen die Lage des regimes eines Ökosystems innerhalb einer heuristischen Stabilitätslandschaft. Langsame Variablen sind letzten Endes für den Übergang eines regimes in eine andere basin of attraction verantwortlich. Akzeptiert man diese Vorstellung von Ökosystemdynamik, stellt die Kennzeichnung dieser langsamen Variablen, neben anderen, eine wichtige Möglichkeit dar, verschiedene Aspekte der ökologischen Resilienz (latitude, resistance, precariousness, panarchy) zu bestimmen. Die Identifizierung langsamer

Variablen und deren Verhalten gegenüber schnellen Variablen für die verschiedenen Typen von Ökosystemen ist, meiner Meinung nach, eines der wichtigsten Themen für weitere Forschungsbemühungen innerhalb der Debatte über Stabilitätseigenschaften. Dies könnte dazu beitragen, die Position von ökologischen Schwellenwerten zu bestimmen, sowie den Zeitpunkt von regime-Übergängen in ungewollte basins of attraction vorherzusagen. Dabei spielen Indikatoren der langsamen Variablen eine große Rolle.

Neben der Identifizierung langsamer Variablen, stellen die Festsetzung eines zeitlich und räumlichen Maßstabs, die Auswahl erwünschter Ökosystem-Services, und die Bestimmung des Störungsregimes, wichtige Bestandteile einer umfassenden Analyse der ökologischen Resilienz dar.

Mehrere Mechanismen auf unterschiedlichen Ebenen der Ökosystemhierarchie sind für die Emergenz der ökologischen Resilienz auf einem größeren Maßstab verantwortlich. Die meisten von ihnen beziehen sich auf eine umfassende Definition von Biodiversität, wie zB die Konzepte der ökologischen Redundanz, response diversity und ecological memory. Störungen auf einem kleinen Maßstab werden als integraler Bestandteil der Ökosystemdynamik und als ein Prozess angesehen, der Erneuerung und Neuheit in einem Ökosystem etabliert. Diese Mechanismen können für einen adaptiven Umweltmanagementansatz verwendet werden, der Biodiversität erhält bzw. steigert, Störungen auf einem kleinen Maßstab zulässt sowie flexibel auf unvorhersehbare Veränderungen innerhalb der Ökosystemdynamik reagiert.

Somit ist festzustellen, dass die Operationalisierung von ökologischer Resilienz möglich ist. Es ist allerdings schwierig, wenn nicht unmöglich, die genaue Position eines regimes innerhalb einer Stabilitätslandschaft zu bestimmen. Nimmt man die Möglichkeit hinzu, Resilienz-Mechanismen in ein Umweltmanagement zu implementieren, kann das Konzept der ökologischen Resilienz für ein Umweltmanagment und den Naturschutz von hoher Bedeutung sein. Um den normativen Ansprüchen einer Theorie Nachhaltiger Entwicklung gerecht zu werden, stellt es ein Werkzeug dar, (kritisches) Naturkapital konstant zu halten.

Essential literature arranged by topic

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