Critical natural capital revisited: Ecological resilience and sustainable development

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1. Introduction

Sustainable development represents one of the key challenges of the 21st century (Sachs, 2005; Clark, 2007). Even though there is a wide political consensus on the principal idea of sustainability, scientific agreement regarding the key question ‘what to sustain?’ (Dobson, 1996; NRC, 1999; Kates et al., 2005) is a far off horizon. It is still controversial what types of capital, i.e. natural capital (e.g. ecosystems, air, water), cultivated natural capital (e.g. salmon farms, wineries), man-made capital (e.g. infrastructure), social capital (e.g. political institutions), human capital (e.g. skills, education) and knowledge capital ought to be preserved in favour of current and future generations (Costanza et al., 2007).

At this conceptual level of sustainability science, basically two positions fight for validity (cf. Neumayer, 1999; Ott and Döring, 2004). Weak sustainability holds that utility (or well-being) ought to be maintained over intergenerational time scales. In this conception, natural capital and man-made capital are viewed as substitutes within specific production processes. Consequently, the stock of the natural capital can be depleted, unless the utility over time is declining (cf. Pezzey, 1992; Norton and Toman, 1997). In contrast, strong sustainability states that natural capital and man-made capital must be viewed as complementary. We are obliged to keep each type of capital intact over time. Thus, the whole stock of natural capital ought to be preserved for current and future generations in the long run (Daly, 1996).

The concept of critical natural capital emerged as a trade-off between these two extreme positions. It signifies the part of the natural capital that performs important and irreplaceable environmental functions, i.e. ecosystem services (de Groot et al., 2005) that cannot be substituted by other types of capital (de Groot et al., 2003; Dietz and Neumayer, 2007). Paradigmatic examples include
essential ecosystem services, such as freshwater resources, climate regulation and fertile soils (MEA, 2005). It is this importance for the quality of life and the survival of humans that makes critical natural capital such an important objective of sustainability. Critical natural capital represents thus the part of the natural environment that ought to be maintained in any circumstances in favour of present and future generations. In addition, the identification and management of critical natural capital is a promising tool for a sound approach to environmental policy (Ekins et al., 2003). The quest for a clear conceptualization of ‘critical natural capital’ is hence worth the effort.

Yet conceptual confusion is immense, as numerous scientific disciplines and societal groups bring their own perspective in valuing nature. It is indeed highly unclear what makes natural capital ‘important’, ‘irreplaceable’, and therefore ‘critical’. In other words, it is controversial which measure would be appropriate to reflect or mirror ‘criticality’. Is it the ecological importance we ascribe to certain habitats due to a high degree of species richness or “naturalness”? Is it the economic value that some ecosystem services bring for human society? Or is it the socio-cultural relevance of the “landscape”? The important point here is that we urgently need well-founded criteria to assess the specific criticality of natural capital stocks (MacDonald et al., 1999; Ekins et al., 2003).

This article examines the link between the concept of ecological resilience and critical natural capital. I state that the empirical estimation of ecological resilience can help a great deal in assessing the “ecological criticality” of specific parts of renewable natural capital. More specifically, I propose that the amount of resilience can be used to estimate the degree of threat certain ecosystems are prone to. The concept of ecological resilience therefore adds a further important criterion to build a comprehensive conception of criticality.

The article is organized as follows. The first section offers a short description of the concept of ecological resilience with an emphasis on questions regarding the conceptualization and measurement of ecological resilience. Subsequently, the second section revisits the concept of critical natural capital and formulates a comprehensive approach to criticality. Based on these conceptual reflections, the third section examines the link between the concept of ecological resilience and a conception of critical natural capital. Finally, the fourth section concludes with the findings of this paper.

2. The concept of ecological resilience

Among the scientific concepts currently used in sustainability science, ‘resilience’ is one of the most prevalent and topical (e.g. Kates et al., 2001; Foley et al., 2005). First of all, two distinct meanings of the term must be distinguished. The first one refers to dynamics close to equilibrium and is defined as the time required for a system to return to an equilibrium point following a disturbance event. It has been coined “engineering resilience” (Holling, 1996) and is largely identical to the stability property “elasticity” (Grimm and Wissel, 1997). The second meaning of resilience refers to dynamics far from any equilibrium steady state and is defined as the capacity to absorb shocks and still maintain “function” (cf. for the ambiguous term ‘function’ Jax, 2005). This meaning has been termed “ecological resilience” (e.g. Gunderson and Holling, 2002; Gunderson and Pritchard, 2002) and it is this second kind of resilience to which I refer in this text.

The concept of ecological resilience emerged in ecology in the 1960–1970s but has been adopted since then by numerous scientific disciplines, e.g. sociology, economy or political science (cf. Folke, 2006 for a recent review). For instance, Common and Perrings (1992) examined its relation to ecological economics and sustainability concepts. It is currently used either as a descriptive concept that is applied primarily to ecological systems, i.e. ecological resilience (e.g. Bellwood et al., 2003; Nyström, 2006), or as a boundary object, a term that facilitates communication across disciplinary borders, i.e. social–ecological resilience (cf. Brand and Jax, 2007). In the latter interpretation, the concept is viewed as an innovative perspective to analyze coupled social–ecological systems (Walker and Salt, 2006; Walker et al., 2006).

This article focuses on the former descriptive meaning of the term. Ecological resilience is defined as the capacity of an ecosystem to resist disturbance and still maintain a specified state. In this definition the concept gets close to the stability concept ‘resistance’, as identified by Grimm and Wissel (1997). This may underestimate other important characteristics of ecological resilience, such as the capacities for renewal, reorganization and development (Folke, 2006). Yet this definition is in my view useful and workable to be used for the measurement of ecological resilience in real-world ecosystems.

How to measure ecological resilience? Carpenter et al. (2005) recently noticed that ecological resilience cannot be measured directly. Rather it must be estimated by means of resilience surrogates, i.e. indirect proxies that are derived from theory used in indicating resilience (cf. for resilience theory Walker et al., 2006). Surrogates for ecological resilience refer to the concept of resilience mechanisms, e.g. ecological redundancy, response diversity or ecological memory (e.g. Bellwood et al., 2003; Bengtsson et al., 2003; Nyström, 2006), the concept of maintained system identity (Cumming et al., 2005), probabilistic resilience and percolation theory (Peterson, 2002) or to approaches using the concept of alternative stable states and ecological thresholds (e.g. Scheffer et al., 2001; Bennett et al., 2005). Before I will expand in some detail on the latter approach it must be noted that each estimation of ecological resilience is based on a comprehensive resilience analysis, which includes the identification of the specific disturbance regime and a societal choice of the desired ecosystem services. Confer for a detailed review of a resilience analysis Carpenter et al. (2001), Walker et al. (2002) and Resilience Alliance (2007).

The threshold (T-) approach to resilience surrogates is used widely in the relevant literature (e.g. Carpenter et al., 2001; Scheffer et al., 2001; Peterson et al., 2003; Bennett et al., 2005). Despite its actual prominence it is based on two controversial assumptions. The first assumption holds that ecosystems can shift non-linearly between alternative stable states that are separated by ecological thresholds. For example, coral reefs can show an algae-dominated or a coral-dominated state while savannahs may exhibit a grassy or woody state. Yet this is true for many ecosystem types (Folke et al., 2004; Walker and Meyers, 2004) but not all (Schröder et al., 2005). Indeed, the weight of empirical evidence shows that the relative frequency of the occurrence of alternative stable states across systems is higher for systems controlled by environmental adversity, e.g. deserts,
Scheffer and Carpenter (2003) and Brand and Jax (2007). (For the reader is referred to the web version of this article.)

The second assumption states that ecosystem dynamics can be understood by analyzing a few key variables, which is termed the ‘rule of hand’. Key variables are subdivided into fast variables and slow variables according to the turnover rates in space and time (Rinaldi and Scheffer, 2000; Walker et al., 2006). The important (and controversial) assumption here is that the slow variables are viewed to ‘control’ the whole ecosystem in determining the system’s position within a stability landscape (Walker et al., 2004). The value of the slow variable, e.g. the abundance of woody plants in rangelands or the phosphate concentrations in a shallow lake, is thus regarded as the most relevant factor for the maintenance of ecological resilience (e.g. Gunderson and Walters, 2002).

However, the concept of ‘rule of hand’ is limited, as it is dependent on a certain educated guess regarding the variables and parameters that are to be included in the model (Schmitz 2000). A further shortcoming is that this approach must ignore the individual variability of organisms (Grimm, 1999). The ‘rule of hand’-approach is not false but rather partial and may be complemented by bottom-up approaches (Grimm, 1999), e.g. individual-based models (cf. Grimm and Railsback, 2005). From the perspective of individual-based modellers, it is highly controversial, for instance, whether the key variables and the controlling slow variables actually can be identified (V. Grimm, personal communication).

The threshold approach for the estimation of ecological resilience is thus restricted to ecological situations in which both assumptions hold: ecosystems must exhibit alternative stable states and it must be possible to identify the key controlling variables. When these necessary preconditions are met, the threshold approach tracks ecological resilience by means of a bifurcation diagram. As illustrated in Fig. 1, this diagram plots the equilibria of an ecosystem on axes of a fast variable and a slow variable. In the case of a shallow lake the fast variable is represented by the abundance of submerged plants while the slow controlling variable corresponds to the phosphate concentrations in the sediment of the lake. The plot then shows upper and lower sets of stable regions (the solid lines in Fig. 1) separated by two ecological thresholds (ET₁ and ET₂) and an unstable set of equilibria (the dashed line). The important point here is that the value of the slow variable CVsv is regarded to control an ecosystem’s position in state space, and thus, to be responsible for the maintenance of ecological resilience of the whole system. Therefore, the degree of the ecosystem’s ecological resilience is estimated as the distance from the current value of the slow variable (CVsv) to the value of the ecological threshold (ET₁) (Rinaldi and Scheffer, 2000; Peterson et al., 2003). In other words: in this approach, the slow variable performs as a resilience surrogate, as it is this variable that controls the position of the whole ecosystem within state space.

Note that this methodology of estimating ecological resilience focuses on the precariousness of the system, i.e. the current trajectory of the system and proximity to a limit or threshold. Clearly, there might be other important facets of ecological resilience, such as resistance or latitude (Walker et al., 2004).

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

Note that this early type of threshold approach referring to the ‘slow variable surrogate’ and bifurcation diagrams has recently been developed further by Bennett et al.’s (2005) alternative approach where they also take into account time (i.e. ‘how fast is the slow variable moving toward or away the ecological threshold’) and whether ecological thresholds are dynamic or static. A comprehensive discussion of Bennett et al.’s (2005) sound approach is beyond the realm of this article.

The message of this section is that ecological resilience represents a relatively clearly defined stability concept that is embedded in a rich resilience theory. It is principally possible to empirically estimate surrogates for ecological resilience. This will be important in the fourth section when I connect the

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Fig. 1 – Bifurcation diagram of a system described by a fast and a slow variable: the stable regimes are given by the solid lines and the unstable states by the dashed line. ET₁ and ET₂ represent ecological threshold points and CVsv signifies the current value of the slow variable. Ecological resilience (ER) is measured as the distance from CVsv to ET₁. Modified from Scheffer and Carpenter (2003) and Brand and Jax (2007). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)
concept of ecological resilience to the concept of critical natural capital. In the next section we will at first have a closer look at the specific conceptualization of the term ‘critical natural capital’.

3. Critical natural capital: some conceptual remarks

The concept of critical natural capital has been developed by Turner (1993) following capital theory in economics (cf. for the term ‘capital’ Ekins et al., 2003). It gained some attention with the recent EU-project CRITINC (cf. Ekins, 2003; Ekins et al., 2003) and has been applied to several EU-countries, such as France (Douguet and O’Connor, 2003) and the UK (Ekins and Simon, 2003). The following delineations in this section must be viewed as complementary to the sound work of the CRITINC project.

The concept of critical natural capital is obviously based on the concept of natural capital, often understood as any stock of natural resources or environmental assets that provides a flow of useful goods or services, now and in the future (Pearce and Turner, 1990; MacDonald et al., 1999; de Groot et al., 2003). The term ‘natural capital’ has been criticized for its reductionistic and utilitarian connotations (cf. Chiesura and de Groot, 2003), praised for its terminological strengths (Dobson, 1996) and continues to stimulate a debate about its accurate conceptual intension and extension (Ott and Döring, 2004; De Groot, 2006). It is used widely to signify a myriad of components (e.g. resources, biodiversity, fertile soil, ozone layer), properties (e.g. ecological resilience, ecosystem health, integrity) and dispositions (e.g. regulative or assimilative capacities). Natural capital is thus a multidimensional meta-concept for a plurality of interrelated and heterogeneous stocks that perform various functions and services for human society (Chiesura and de Groot, 2003; Ott and Döring, 2004; Aronson et al., 2006a).

Regarding the multidimensional character of natural capital, it is not surprising to find conceptual confusion about ‘critical natural capital’ (Turner, 1993, MacDonald et al., 1999). This is due to the existence of different domains under which natural capital can be considered as critical, as different disciplines bring different conceptual frameworks to value ecosystems (Chiesura and de Groot, 2003). The decisive question is: ‘critical for what and for whom?’ (de Groot et al., 2003). Considering all the relevant literature at least six domains may be distinguished under which natural capital is evaluated as critical:

1. socio-cultural: natural capital becomes important, crucial or vital for a particular social group, as it provides the socio-cultural context for human society in terms of non-materialistic needs, e.g. health, recreation, scientific and educational information, cultural identity, source of spiritual experience or aesthetic enjoyment (Chiesura and de Groot, 2003, cf. also Kazal et al., 2006).
2. ecological: natural capital is ecologically valued for its importance in terms of naturalness, biodiversity, irreversibility or uniqueness (de Groot et al., 2003; de Groot, 2006), for instance.
3. sustainability: this domain refers to the debate of weak vs. strong sustainability described in the introduction of this paper. Natural capital is viewed as critical as regards to human well-being if it is non-substitutable with other types of capital (Turner, 1993; Neumayer, 1999; Dietz and Neumayer, 2007). Good examples are life-securing ecosystem services, such as the provision of food, raw materials or drinking water.
4. ethical: a loss of natural capital can be morally disadvantageous in that moral values are being violated (Dietz and Neumayer, 2007). For example, from the standpoint of sentientism the preservation of higher developed animals, e.g. bears, beavers or casuaries, would be prima facie regarded as critical (cf. Haider and Jax, 2007).
5. economic: the loss of natural capital can also bring about very high economic costs. These costs can be validated by the full spectrum of monetary valuation (cf. de Groot, 2006).
6. human survival: natural capital becomes obviously critical when without it human life would not be possible (Dobson, 1998). Examples are climate regulation, flood regulation or fertile soils.

Consequently, definitions of critical natural capital are manifold, as specific domains of criticality are being stressed. Some definitions refer to one domain of criticality only. For instance, Turner’s (1993, 11) definition “[t]he constraint [of critical natural capital] will be required to maintain populations/resource stocks within bounds thought to be consistent with ecosystem stability and resilience” is ecological whereas the definition put forward by Douguet and O’Connor’s (2003, 237): “natural capital which is responsible for important environmental functions and which cannot be substituted in the provision of these functions by manufactured capital” stresses the sustainability domain. Alternative definitions try to include a higher amount of criticality domains, as for instance, the definition proposed by Dietz and Neumayer (2007, 619): “we may ‘ring-fence’ as critical any natural capital that is strictly non-substitutable (also by other forms of natural capital), the loss of which would be irreversible, would entail very large costs due to its vital role for human welfare or would be unethical”.

Each of these definitions refers to specific domains of criticality only and can thus be criticized as partial and incomplete. There is the need for a more comprehensive approach to criticality. In this article I consider natural capital to be critical if it applies to at least one of the six domains of criticality, i.e. the socio-cultural, ecological, sustainability, ethical, economic or the human survival domain. In this conception criticality comes in degrees. Criticality is dependent on (a) the amount of significance within one domain of criticality (e.g. the socio-cultural importance, the ethical value, the economic costs), (b) the amount of domains under which the natural capital is valued (i.e. the more domains the more critical) and (c) the weighing of the different domains. Hence, different parts of natural capital can have various degrees of criticality. It is also important to note that criticality is to some degree context-specific (de Groot et al., 2003), as it is related to certain standards of living and human values that may change over time.

Altogether it has become clear that the concept of critical natural capital is by no means rooted solely in the natural sciences but also in the full array of social sciences and the humanities. In the subsequent section I will have a closer look
4. Ecological resilience and critical natural capital

What is the relation between ecological resilience and critical natural capital? Previous work determined several links of ecological resilience to criticality. For instance, Serrão et al. (1996) point to environmental criticality, i.e. a state of nature in which the extent of environmental degradation passes a threshold beyond which current levels of social welfare may not be supported. Even though no clear measure is being proposed, Serrão et al. (1996) suggest that the estimation of environmental criticality requires information about ecological resilience.

Alternatively, Deutsch et al. (2003) acknowledge ecosystem performance as a criterion for criticality. By ‘ecosystem performance’ they mean ‘the dynamic, often non-linear interrelations between populations and communities of plants, animals and microorganisms and their energetic, hydrological and biogeochemical environment’. This ecosystem property is perceived as an underlying pre-requisite for human well-being because it generates and sustains the flow of ecosystem services. This notion is closely related to several other terms used in the literature on natural capital, e.g. functions-of nature (De Groot et al., 2003), regulation functions (De Groot et al., 2002), life-support functions (Ekins et al., 2003) or regulating services (MEA, 2005). The important point here is that Deutsch et al. (2003) link ecosystem performance to resilience theory, and in particular to the concepts of pulse disturbance, alternative stable states, slow variables and biodiversity. Hence, the relation of ecological resilience and critical natural capital has been the subject of scientific debate since several years.

In the subsequent delineations I will follow De Groot et al.’s (2003) approach to criticality. In order to stress the ecological aspect within a conception of critical natural capital, its multi-dimensionality can be boiled down to two criteria: importance and degree of threat (cf. Fig. 2). As De Groot et al. (2003) argue, it is at first appropriate to conceptualize criticality with reference to the importance society ascribes to natural capital. Yet human activities can bring changes in the natural capital and these changes, in turn, can affect its ecological, socio-cultural or economic importance, for instance. Therefore, another criterion for determining criticality is found: ‘degree of threat’. De Groot et al. (2003) argue that the degree of threat ecosystems are exposed to is assessed based on changes in quantity and quality of the remaining natural capital. Ecosystem quantity simply refers to the percentage a region is covered by a certain ecosystem type and can be determined by means of land cover databases. In contrast, ecosystem quality is being related to the concepts of integrity and vulnerability and estimated by changes in species richness or pressure on ecosystems.

This article follows the approach to criticality suggested by De Groot et al. (2003) but proposes an alternative method to assess the degree of threat based on the concept of ecological resilience. This section thus develops an approach to ‘ecological criticality’ and puts an emphasis on the ecological domain of criticality. It is certainly not the suggestion to neglect neither the relevance of other domains of criticality, e.g. the socio-cultural, ethical, economic importance nor the criterion of ecological importance, which includes e.g. uniqueness, naturalness or biodiversity. What I propose is rather that the concept of ecological resilience provides a useful means for estimating the degree of threat ecosystems may face.

My argument goes as follows. Ecological resilience is defined in its ecosystem services-related meaning (Brand and Jax, 2007) as the underlying capacity of an ecosystem to maintain desirable ecosystem services in the face of human use and a fluctuating environment (e.g. Carpenter et al., 2001; Folke et al., 2002). The loss of ecological resilience thus indicates whether ecosystems are prone to shifts to undesirable ecosystem states that cease to deliver the ecosystem services people value in a specific case (Folke et al., 2004). I infer from this first that an ecosystem’s amount of ecological resilience is directly linked to the degree of threat this ecosystem may face. Second, I propose that in order to quantify the degree of threat an ecosystem is exposed to, it is necessary to estimate the ecological resilience of an ecosystem in a specific case.

Consider again the bifurcation diagram in Fig. 3. By using the threshold approach, a surrogate for ecological resilience can be estimated as the distance from the current value of the slow variable ($C_{sv}$) to the (predicted) value of the ecological threshold ($ET_1$). The important point here is that ecological resilience (ER) is directly linked to the degree of threat (DT) the ecosystem is exposed to. Strictly speaking, the amount of ER is inversely related to DT, that means $ER = \frac{1}{DT}$. If the amount of ER is low (i.e. the system is close to the ecological threshold $ET_1$), the DT would be high, and if the amount of ER is high (i.e. the system is far from the ecological threshold $ET_1$), the DT would be low. Clearly, the inverse relationship of ecological resilience and the degree of threat applies to each surrogate of ecological resilience. That means, it is also possible to estimate the degree of threat by means of other resilience surrogates, such as resilience mechanisms (e.g. Nyström, 2006), maintained system identity (Cumming et al., 2005) or percolation theory (Peterson, 2002).

Consider a simple model of a shallow lake ecosystem as an example. Shallow lakes can exhibit two stable regimes with respect to nutrient load, i.e. a clear-water regime with aquatic plants and a turbid regime without vegetation. If the lake is in the clear-water regime, an increase of the nutrient level will lead to a gradual and moderate rise in turbidity until the critical

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Fig. 2 – A conception of critical natural capital. The ‘criticality’ results from the two criteria ‘importance’ and ‘degree of threat’. Modified from De Groot et al. (2003).
turbidity for plant survival is reached. At this point, vegetation collapses and the lake shifts to the turbid regime. Following the threshold approach, ecological resilience corresponds to the distance of the current value of nutrient concentrations to the critical level (Rinaldi and Scheffer, 2000; Scheffer et al., 2001; Peterson et al., 2003). Thus, if the specific shallow lake had a very low nutrient level, the ecological resilience would be high. As a consequence, its degree of threat would be low. This example is certainly oversimplifying (cf. Scheffer and van Nes, 2007). Yet I want to illustrate here that ecological resilience theory is principally rich enough to spell out the degree of threat ecosystems are prone to.

It is important to note that the amount of ecological resilience can only be used as a criterion of criticality for specific dimensions of natural capital, as natural capital is a multi-facet concept that includes divergent environmental media, such as air, water or habitats. According to the classifications proposed by de Groot et al. (2002) and Ekins et al. (2003), ecological resilience is thus applied exclusively to natural capital that (a) is renewable, (b) refers to the basic type ‘habitat’ (i.e. ecosystems, flora and fauna) and (c) refers to life-support functions, i.e. the capacity to sustain “ecosystem health” and “function”. Paradigmatic examples are represented by any ecosystem type, e.g. shallow lake, savannah or boreal forest.

Using the concept of ecological resilience can thus help to specify the ‘ecological criticality’ of specific ecosystem types. This approach to ecological criticality is based on the ideas proposed by Serrão et al. (1996), Deutsch et al. (2003) as well as De Groot et al. (2003) and is not suggesting to replace these ideas in any way. Rather, the resilience-approach should be used complementary to the concept of vulnerability (and integrity or ecosystem health) in order to estimate the degree of threat specific ecosystems face. At this point it becomes again apparent that ecological resilience and vulnerability can be used as complementary measures of “stability” (cf. Gallopin, 2006). The concept of ecological resilience therefore adds a further criterion to a comprehensive conception of critical natural capital.

5. Conclusion

This article concludes that the concept of ecological resilience can help a great deal to specify the ‘ecological criticality’ of specific parts of natural capital (cf. also Deutsch et al., 2003). The empirical estimation of ecological resilience represents a measure to estimate the degree of threat an ecosystem is exposed to. More specifically, I propose that an ecosystem’s degree of ecological resilience is inversely related to its degree of threat. Clearly, the quantification of the degree of threat requires the empirical estimation of ecological resilience. This is possible by means of several methods, such as the threshold approach (e.g. Carpenter et al., 2001; Peterson et al., 2003), the concept of resilience mechanisms (e.g. Bengtsson et al., 2003; Bellwood et al., 2003; Nyström, 2006), probabilistic resilience and percolation theory (Peterson, 2002) or the concept of maintained system identity (Cumming et al., 2005). The concept of ecological resilience may be used complementary to other approaches that mirror the degree of threat ecosystems face, such as integrity or vulnerability (cf. De Groot et al., 2003). Thus, the degree of ecological resilience can be used as a further criterion for the criticality of natural capital. This measure does not replace other criteria for criticality, such as the socio-cultural relevance, the economic value or the ecological importance, but rather complements them in order to build a comprehensive conception of critical natural capital.

This article suggests that the estimation of ecological criticality, and thus ecological resilience, is important for the maintenance of valuable ecosystem services and the sustainable use of renewable natural capital (cf. also MEA, 2005). In a like vein, Mäler (2008) states that the ecological resilience of a system should be regarded as an important capital stock for achieving sustainability, while Aronson et al. (2006a,b) assert that dwindling natural capital effectively limits economic growth globally and argue for ecological restoration measures as a counter-strategy. Yet the importance of natural capital for achieving sustainability is not self-evident. The relevance we ascribe to natural capital in general and a system’s ecological resilience in particular depends on the specific conception of ‘sustainable development’ we advocate. Apparently, a conception of strong sustainability would set a higher value on the management of critical natural capital and to concepts such as ecological resilience than a conception of weak sustainability would do (Brand, 2005: 22). The resolution of the persisting controversy of weak vs. strong sustainability (Neumayer, 1999) is a far off horizon, yet would bring immense theoretical progress to sustainability science and, in my view, ought to be one of the fundamental theoretical foundations of ecological economics. There is no doubt that the ‘discourse-rational choice about the right sustainability conception’ must be based on a variety of criteria (cf. Ott and Döring, 2004, 2007; Schlüns et al., in press) and that research on ecological resilience can only provide some tentative arguments for the importance of natural capital, and thus in favour of a conception of strong sustainability. Yet in this
way, resilience research (cf. Folke, 2006; Walker et al., 2006) may help to challenge one of the central controversies in sustainability science and to build an important theoretical foundation for ecological economics.

Five years after the CRITINC-research project (cf. Ekins, 2003) a clear conceptualization of critical natural capital is hence worth the effort. Such a concept may inform environmental policy and management to identify the natural capital that ought to be preserved in any circumstances in favour of current and future generations. It is thus an important step in our quest for sustainable development.

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