Chapter 3: The economic insurance value of ecosystem resilience

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Abstract: Ecosystem resilience, i.e. an ecosystem's ability to maintain its basic functions and controls under disturbances, is often interpreted as insurance: by decreasing the probability of future drops in the provision of ecosystem services, resilience insures risk-averse ecosystem users against potential welfare losses. Using a general and stringent definition of "insurance" and a simple ecological-economic model, we derive the economic insurance value of ecosystem resilience and study how it depends on ecosystem properties, economic context, and the ecosystem user's risk preferences. We show that (i) the insurance value of resilience is negative (positive) for low (high) levels of resilience, (ii) it increases with the level of resilience, and (iii) it is one additive component of the total economic value of resilience.

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1 Introduction

Ecosystems that are used and managed by humans for the ecosystem services they provide may exhibit multiple stability domains ("basins of attraction") that differ in fundamental system structure and behavior. As a result of exogenous natural disturbances or human management, a system may flip from one stability domain into another one with different basic functions and controls (Holling 1973, Levin et al. 1998, Scheffer et al. 2001). As a consequence, also the level, composition and quality of ecosystem services may abruptly and irreversibly change. Examples encompass a diverse set of ecosystem types that are highly relevant for economic use, such as boreal forests, semi-arid rangelands, wetlands, shallow lakes, coral reefs, or high-seas fisheries (Gunderson and Pritchard 2002).

The term "resilience" has been used to denote an ecosystem's ability to maintain its basic functions and controls under disturbances (Holling 1973, Carpenter et al. 2001). The economic relevance of ecosystem resilience is obvious, as a system flip may entail huge welfare losses.[1] For example, a combination of drought, fire and ill-adapted livestock grazing management in sub-Saharan Africa, central Asia and Australia have lead to severe degradation and desertification of semi-arid rangelands, which provide subsistence livelihood for more than one billion people worldwide. Once degraded, these grassland ecosystems cannot be used as pasture anymore (Perrings and Walker 1995, Perrings and Stern 2000). In Africa alone, almost 75 % of semi-arid regions are threatened by degradation and desertification (UNO 2002). Worldwide, the income loss associated with desertification of agricultural land is estimated to some 42 billion US dollars per year (UNCCD 2005).

An ecosystem's resilience in a given stability domain can be measured by the probability that exogenous perturbations make the system flip into another stability domain. Therefore, enhancing the resilience of a particular (desired) domain reduces the likelihood of a flip into another (less desired) domain. It is for this reason that ecosystem resilience has been referred to as "insurance", e.g. in the following manner:

> "Resilience can be regarded as an insurance against flips of the system into different basins of stability." (Mäler 2008: 17)

[1] Accordingly, some have included a reference to the provision of desired ecosystem services right into the definition of ecosystem resilience, e.g. as the capacity of an ecosystem "to maintain desired ecosystem services in the face of a fluctuating environment and human use" (Brand and Jax 2007: 3, referring to Folke et al. 2002).

> “[R]esilience [...] provides us with a kind of insurance against reaching a non-desired state.” (Mäler et al. 2009: 48)

> “The link between biodiversity, ecosystem resilience and insurance should now be transparent. [...] It follows that the value of biodiversity conservation lies in the value of that protection: the insurance it offers against catastrophic change.” (Perrings 1995: 72)

> “The resilience of the ecological system provides ‘insurance’ within which managers can affordably fail and learn while applying policies and practices.” (Holling et al. 2002: 415)

So far in the resilience literature, the term “insurance” is employed in a rather metaphoric manner – as a metaphor for “keeping an ecosystem in a desirable domain”. It is used to convey the message that resilience is a desirable property of some ecosystem since it helps to prevent catastrophic and irreversible reductions in ecosystem service flows. While ecosystem resilience obviously and undoubtedly includes an insurance aspect, no explicit attempt has been made so far to use a clearly defined concept of “insurance” from the established literature on insurance and financial economics. As a result, it remains unclear what exactly constitutes the economic insurance value of ecosystem resilience, how it depends on ecosystem properties, economic context, and the ecosystem user’s risk preferences, and how it relates to the total economic value of ecosystem resilience.

In an attempt to conceptually determine and to empirically capture the economic value of ecosystem resilience, Mäler et al. (2007) and Walker et al. (2007) have suggested to use the shadow price of resilience as a measure of its economic value. They calculate the present discounted value of future improvements in welfare from ecosystem services, where these future improvements accrue from reduced risks of a system flip due to a unit increase in the initial level of resilience. While this procedure establishes the total economic value of resilience, it does not explicitly relate it to any idea of “insurance”.

In this paper, we aim to close that gap and to provide some conceptual clarification. Any idea of “insurance” fundamentally refers to a combination of three elements: (i) the objective characteristics of risk in terms of different possible states of nature, (ii) people’s subjective risk preferences over these states, and (iii) a mechanism that allows mitigation of (i) in view of (ii). We believe that the ongoing discussion of resilience as an insurance could be clarified and fruitfully advanced if reference to these three elements was made explicitly and rigorously, and we propose an analytical framework for that purpose. We adopt a clear and generally accepted definition of "insurance" from the risk and finance literature, according to which *insurance* is an action or institution that mitigates the influence of uncertainty on a person's well-being (McCall 1987). Based on this definition, we conceptualize resilience's economic insurance value as the value of one very specific function of resilience: to reduce an ecosystem user's income risk from using ecosystem services under uncertainty. We also analyze how exactly the insurance value of ecosystem resilience depends on ecosystem properties, economic context, and on the ecosystem user's risk preferences.

Our analysis yields several interesting and important results. First, the insurance value of resilience is negative for low levels of resilience and positive for high levels of resilience. That is, ecosystem resilience actually functions as an economic insurance only at sufficiently high levels of resilience; it does *not* function as an economic insurance at low levels of resilience. Second, the (marginal) insurance value of resilience increases with the level of resilience – for some ecosystem types even monotonically. This is in contrast to normal economic goods, the (marginal) value of which *de*creases with their quantity. Third, the insurance value of resilience is one additive component of its total economic value. That is, the total economic value of resilience is larger than just its insurance value. While the latter may be negative, the total economic value of resilience turns out to be always positive.

The paper is organized as follows. In Section 2, we present a stylized model of an ecological-economic system that describes how different degrees of ecosystem resilience are related to different system outcomes, and how this contributes to an ecosystem user's well-being under uncertainty. In Section 3, we clarify what exactly we mean by "insurance" and "insurance value". On this basis, in Section 4, we present our results about the economic insurance value and the total value of ecosystem resilience, with all proofs and formal derivations contained in the Appendix. In Section 5, we discuss these findings and draw conclusions.

2 Model

To discuss the economic insurance value of ecosystem resilience, we propose the following simple and stylized model of an ecological-economic system. Consider an ecosystem that potentially exhibits two different stability domains with respective levels of ecosystem services-production. One domain is characterized by a high level of ecosystem service provision and corresponding net income $y_H \in Y$; the other domain is characterized by a low level of ecosystem service provision and corresponding net income $y_L \in Y$; with $Y \subseteq \mathbb{R}_+$ and $y_L < y_H$, so that

$$\Delta y := y_H - y_L > 0 \tag{1}$$

is the potential income loss when the system flips from the high-production into the low-production stability domain.

Initially, the ecosystem is in the high-production stability domain. In this domain, exogenous stochastic disturbances threaten to trigger a flip into the low-production stability domain. Such a flip may occur with probability p with $0 \le p \le 1$. Conversely, the ecosystem stays in the high-production domain with probability $1-p$.

In line with Holling's (1973) notion of resilience as the maximum amount of disturbance a system can absorb in a given stability domain while still remaining in that stability domain, we define and measure resilience as a continuous state variable $R \in [0,1]$ that determines the probability of the system flipping from the high-production into the low-production stability domain as follows:

$$p = p(R) \qquad \text{with} \qquad p'(R) \le 0 \text{ for all } R \;\text{ and }\; p'(R) < 0 \text{ for all } R \in (0,1) \tag{2}$$

$$\text{and} \qquad p(0) = 1,\; p(1) = 0\;. \tag{3}$$

In words, the higher the ecosystem's resilience in the high-production domain, the lower the probability that it flips into the low-production domain due to exogenous disturbance; with zero resilience, it flips for sure; and with maximum resilience it will certainly not flip. For future reference, we define $\underline{R}$ through $p(\underline{R}) = 1/2$ as the level of resilience at which the probability of a system flip exactly equals the probability of the system not flipping. This is the level of resilience at which the future state of nature is maximally uncertain.

In order to give more ecological structure to our ecosystem model (2)–(3), in some parts of our analysis we assume the following more specific model about the relationship between the level of resilience R and the flip probability p:

$$p(R) = 1 - R^{1-\sigma} \quad \text{with} \quad -\infty < \sigma < 1\;. \tag{4}$$

This model has the fundamental resilience-defining properties (2) and (3). In addition, it has the analytically handsome property that $p'(\cdot)$ is a constant-elasticity function of R, where the parameter σ is the (constant) elasticity of $p'(\cdot)$,[2] i.e. σ specifies by how much (in percent) the flip probability's slope increases when the level of resilience increases by 1 %. For short, we will refer to σ as "the ecosystem's elasticity". As σ may be positive or negative, one has[3]

$$p''(R) \left\{ \begin{matrix} > \\ = \\ < \end{matrix} \right\} 0 \text{ for all } R \in (0,1) \qquad \text{if and only if} \qquad \sigma \left\{ \begin{matrix} > \\ = \\ < \end{matrix} \right\} 0 \,.$$

Lacking ecological evidence or a plausible guess on the value of σ, we will study the full range of theoretically possible values of σ. Notwithstanding this generality, the case of $\sigma = 0$ has an epistemically outstanding importance. For, one may argue that one can meaningfully define and measure the system's state variable "resilience" only through, and not independently of, the observable variable "flip probability".[4] Such an epistemic equivalence between the state variable R and the observable p is exactly what is expressed by $\sigma = 0$. In this case, (4) reduces to a linear negative relationship, $p(R) = 1 - R$, so that resilience is measured directly in units of reduced flip-probability. As this case has an epistemically outstanding importance, we will treat the case of $\sigma = 0$ as the preeminent case, and discuss the cases of $\sigma < 0$ and $\sigma > 0$ against it.

Given the ecosystem model (2), (3) – or, more specifically, model (4) – the ecosystem user thus faces a binary income lottery $\{y_L, y_H; p(R), (1 - p(R))\}$. That is, given that the system is initially in the high-production stability domain and is characterized by a level R of resilience, the system will provide net income y_L with probability $p(R)$ and net income y_H with probability $1 - p(R)$. Obviously, with changing level of resilience R the statistical distribution of income will also change. As in our simple analytical framework only the level of resilience R may vary, R uniquely characterizes the income lottery. One may thus speak of "the income lottery R".

[2] Note that (4) implies $-p''(R)R/p'(R) = \sigma$.

[3] For $\sigma = 0$, $p''(R) = 0$ holds also for $R = 0$ and $R = 1$. Yet, for $\sigma < 0$, one has $p''(0) = 0$, and for $\sigma \to 1$, one has $p''(1) \to 0$.

[4] If the system's state space was one-dimensional, one could indeed meaningfully define and measure the system's resilience (sensu Holling 1973) independently of the system's flip probability, namely as the "distance" in state space – measured in units of the single state variable – between the current system state and the threshold between stability domains. However, if the system is characterized by more than one state variable, the "distance" in state space is not uniquely defined but requires some metric which is not naturally given. Then, the system's resilience in a given stability domain cannot be measured through some distance in state space, but only through the observable consequence in terms of flip probability.

We assume that the ecosystem user only cares about (uncertain) income, and not directly about the underlying states of nature in terms of resilience. The ecosystem user's preferences over income lotteries are represented by a von Neumann-Morgenstern expected utility function

$$U = \mathscr{E}_R[u(y)] \qquad \text{with} \qquad u'(y) > 0 \text{ and } u''(y) < 0 \text{ for all } y\ , \tag{5}$$

where $\mathscr{E}_R$ is the expectancy operator based on the probabilities of lottery R, y is net income,[5] and $u(y)$ is a continuous and differentiable Bernoulli utility function which is assumed to be increasing and strictly concave, i.e. the ecosystem user is non-satiated and risk averse.[6] In order to study in the most simple way how the insurance value of resilience depends on the ecosystem user's degree of risk aversion, we assume that the ecosystem user is characterized by constant absolute risk aversion in the sense of Arrow (1965) and Pratt (1964), i.e. $-u''(y)/u'(y) \equiv$ const., so that the Bernoulli utility $u(y)$ function is

$$u(y) = -e^{-\rho y} \qquad \text{with} \qquad \rho > 0\ , \tag{6}$$

where the parameter ρ measures the ecosystem user's risk aversion.

3 Conceptual clarification: insurance and insurance value

Before we derive results about the economic insurance value of ecosystem resilience in the next section, in this section we provide exact definitions of the terms "insurance", "insurance value" and "total economic value". Adopting a very general and widely accepted definition, insurance may be defined in the following way (cf. McCall 1987).

Definition 1
Insurance is an action or institution that mitigates the influence of uncertainty on a person's well-being or on a firm's profitability.

In the concrete setting described in the previous section, the term "insurance" takes on a more concrete meaning. As a person's (here: the ecosystem user's) *well-being* is determined

[5]For notational simplicity, y denotes both the random variable income and income in a particular state of the world.

[6]While risk aversion is a natural and standard assumption for farm *households* (Besley 1995, Dasgupta 1993: Chapter 8), it appears as an induced property in the behavior of (farm) *companies* which are fundamentally risk neutral but act as if they were risk averse when facing e.g. external financing constraints or bankruptcy costs (Caillaud et al. 2000, Mayers and Smith 1990).

by a preference relation over income lotteries, insurance is about the mitigation of income uncertainty, and the person's risk preferences specify what changes in the income lottery actually constitute a "mitigation". Thereby, *uncertainty* exists due the existence of many potential future states of the world (here: high and low ecosystem-service production), in each of which the state-specific income is known (y_H and y_L) and the probability of which is also known ($1-p(R)$ and $p(R)$). That is, uncertainty comes in the form of *risk* in the sense of Knight (1921).

In this more concrete understanding of the term, insurance may come in many forms. One example is the classic insurance contract that an insuree signs with an insurance company under private law, and which specifies that the insuree pays an insurance premium to the insurance company in all states of the world and in exchange obtains from the insurance company an indemnification payment if and only if one particular unfavorable state of the world should occur. Another example is so-called "self-protection" (Ehrlich and Becker 1972), which means that a person undertakes some real action that reduces the probability by which an unfavorable – in terms of net income – state of the world occurs. In this terminology, an increase in the ecosystem's resilience by the manager may be interpreted as insurance because it is a real action that may provide self-protection in terms of net income obtained from the ecosystem.

In order to precisely define and measure the economic insurance value of some act of self-protection (here: an increase in the ecosystem's resilience), we follow Baumgärtner (2007: 103–104). One standard method of how to value the riskiness of an income lottery to a decision maker in monetary terms is to calculate the *risk premium* Π of the lottery, which is defined by (e.g. Kreps 1990, Varian 1992: 181)[7]

$$u\left(\mathscr{E}_R[y]-\Pi\right)=\mathscr{E}_R\left[u(y)\right] \ . \qquad (7)$$

In words, the risk premium Π is the amount of money that leaves a decision maker equally well-off, in terms of utility, between the two situations of (i) receiving for sure the expected pay-off from the income lottery R, $\mathscr{E}_R[y]$, minus the risk premium Π, and (ii) playing the risky income lottery R with random pay-off y.[8] In the model employed here, the risk premium as

[7]By Equation (7), $\mathscr{E}_R[y]-\Pi$ is the *certainty equivalent* of lottery R, as it yields exactly the same expected utility as playing the risky lottery, $\mathscr{E}_R\left[u(y)\right]$.

[8]The risk premium is, thus, the maximum amount of money that a decision maker would be willing to pay for getting the expected pay-off from the income lottery, $\mathscr{E}[y]$, for sure instead of playing the risky income lottery

defined by Equation (7) uniquely exists because, by assumption (cf. Section 2), $y \in Y$ with Y as an interval of $\mathbb{R}$, and u is continuous and strictly increasing (Kreps 1990: 84). In general, if the Bernoulli utility function u characterizes a risk averse decision maker, i.e. if $\rho > 0$ in Equation (6), the risk premium Π is strictly positive.

The economic insurance value of resilience can now be defined based on the risk premium of the income lottery R as follows.

Definition 2

The *insurance value I* of resilience is given by the change of the risk premium Π of the income lottery R due to a marginal change in the level of resilience R:

$$I(R) := -\frac{d\Pi(R)}{dR} \ . \tag{8}$$

Thus, the economic insurance value of ecosystem resilience is the marginal value of its function to reduce the risk premium of the ecosystem user's income risk from using ecosystem services under uncertainty. Being a marginal value, it depends on the existing level of resilience R. The minus sign in the defining Equation (8) serves to express a *reduction* of the risk premium as a *positive* value.

As it is apparent already from Definition 2 (and as it will become more explicit in the following section), the economic insurance value of ecosystem resilience has, in general, an objective and a subjective dimension. The objective dimension is captured by the ecosystem's sensitivity of the flip probability $p(R)$ to changes in the level of resilience, σ; the subjective dimension is captured by the ecosystem user's degree of risk aversion, ρ. If the flip probability would not vary with the level of resilience (i.e. $p'(R) \equiv 0$), or if the ecosystem user was risk-neutral (i.e. $\rho = 0$), the risk premium Π of income lottery R would not vary with R, thus yielding a vanishing insurance value of resilience.

The insurance value of resilience is only a fraction of resilience's total economic value, namely the value of its function to reduce the risk premium of the ecosystem user's income risk from using ecosystem services under uncertainty. Beyond its insurance value, resilience also has economic value in its function to increase the ecosystem user's expected income from ecosystem services. In order to characterize the insurance value of resilience as a fraction of its total economic value, we adopt the following general and widely accepted definition of total economic value under uncertainty (e.g. Freeman 2003: Chap. 8).

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with random pay-off y.

Definition 3

The *total economic value* V of resilience is given by the maximum willingness to pay WTP per unit for a marginal increase of ΔR in the level of resilience R:

$$V(R) := \lim_{\Delta R \to 0} \frac{WTP(\Delta R)}{\Delta R} \,, \tag{9}$$

where WTP is defined through

$$\mathscr{E}_R \left[u(y) \right] = \mathscr{E}_{R+\Delta R} \left[u(y - WTP(\Delta R)) \right] \,. \tag{10}$$

In words, we measure the total economic value of a change ΔR in resilience as the maximum willingness to pay (WTP) for that change, more exactly as the WTP per marginal unit of resilience. The maximum willingness to pay for the increase ΔR in resilience is the amount of money that leaves an individual indifferent, in terms of expected utility, between the two situations of (i) resting in the original position with resilience R and (ii) paying the amount WTP and getting into a situation with resilience $R + \Delta R$.[9] As value is typically expressed as a per-unit quantity characterizing a marginal change, we divide WTP by ΔR and let $\Delta R \to 0$ to obtain the marginal value of resilience. Being a marginal value, it depends on the existing level of resilience R.

In the simple model studied here, with no other constraints or alternative options for action in place, the total economic value of resilience as defined by Definition 3, evaluated at the socially optimal level of resilience, is exactly equivalent to its shadow price (as measured e.g. by Mäler et al. 2007, Walker et al. 2007).

4 Results

Using the concepts defined in Section 3, we can make statements about the model described in Section 2, and, thus, about the economic insurance value of ecosystem resilience. To start

[9]In the language of welfare measurement, WTP is the Hicksian compensating surplus for a finite change of ΔR in the level of resilience (Hicks 1943, Freeman 2003: Chap. 3). Alternatively, one could also use the Hicksian equivalent surplus to measure the monetary value of the welfare change associated with a finite change of ΔR in the level of resilience, that is, the minimum amount of monetary compensation to the individual ("willingness to accept", WTA) that leaves the individual indifferent between the two situations of (i) resting in the original position with resilience R and receiving a monetary payment of WTA and (ii) getting into a situation with resilience $R + \Delta R$. In general, WTP and WTA will differ for finite changes of ΔR. However, for the marginal changes studied here, i.e. for $\Delta R \to 0$, WTP and WTA coincide, so that the value of $V(R)$ does not depend upon whether WTP or WTA is used in the defining Equation (9).

with, we discuss the risk premium associated with different levels of resilience.

Lemma 1

The risk premium $\Pi(R)$ of the income lottery R is given by

$$\Pi(R) = -p(R)\Delta y + \frac{1}{\rho}\ln\left[1 + p(R)\left(e^{\rho\Delta y} - 1\right)\right] , \tag{11}$$

which has the following properties:

(i)

$$\Pi(0) = \Pi(1) = 0 \quad \text{and} \quad \Pi(R) > 0 \text{ for all } R \in (0,1) . \tag{12}$$

(ii) For all $R \in (0,1)$[10]

$$\Pi'(R) \left\{ \begin{array}{c} > \\ = \\ < \end{array} \right\} 0 \qquad \text{for} \qquad R \left\{ \begin{array}{c} < \\ = \\ > \end{array} \right\} \tilde{R} , \tag{13}$$

$$\text{where} \quad \tilde{R} := p^{-1}\left(\frac{1}{\rho\Delta y} - \frac{1}{e^{\rho\Delta y} - 1}\right) , \tag{14}$$

$$\text{so that} \quad \underline{R} < \tilde{R} < 1 \quad \text{and} \quad \frac{d\tilde{R}}{d\rho}, \frac{d\tilde{R}}{d\Delta y} > 0, \frac{d\tilde{R}}{d\sigma} < 0 \tag{15}$$

(iii) There exists $\bar{\sigma}$ with $0 < \bar{\sigma} \leq 1$ and

$$\frac{d\bar{\sigma}}{d(\rho\Delta y)} > 0 , \quad \lim_{\rho\Delta y \to +\infty} \bar{\sigma} = 1 , \quad \lim_{\rho\Delta y \to 0} \bar{\sigma} = 0 , \tag{16}$$

so that

$$\Pi''(R) < 0 \quad \begin{cases} \text{for } R > \tilde{\tilde{R}} & \text{if} \quad \sigma < 0 \\ \text{for all } R \in (0,1) & \text{if and only if} \quad 0 \leq \sigma \leq \bar{\sigma} \\ \text{for } R < \tilde{\tilde{R}} & \text{if} \quad \sigma > \bar{\sigma} \end{cases} , \tag{17}$$

where $\tilde{\tilde{R}}$ is defined through $\Pi''(\tilde{\tilde{R}}) = 0$ for $\sigma < 0$, and through $\tilde{\tilde{R}} = \min\{R \mid \Pi''(R) = 0\}$ for $\sigma > \bar{\sigma}$, so that $\tilde{\tilde{R}} \gtrless \tilde{R}$ for $\sigma \gtrless 0$.

(iv) For all $R \in (0,1)$

$$\frac{d\Pi(R)}{d\rho} > 0 \quad \text{and} \quad \lim_{\rho \to 0} \Pi(R) = 0 , \tag{18}$$

$$\frac{d\Pi(R)}{d\Delta y} > 0 \quad \text{and} \quad \lim_{\Delta y \to 0} \Pi(R) = 0 , \tag{19}$$

[10] For $\sigma = 0$, the statement about the sign of $\Pi'(R)$ holds also for $R = 0$ and $R = 1$. Yet, for $\sigma < 0$, one has $\Pi'(0) = 0$; for $\sigma \to 1$, one has $\Pi'(1) \to 0$.

$$\frac{d\Pi(R)}{d\sigma} \left\{ \begin{array}{c} > \\ \equiv \\ < \end{array} \right\} 0 \quad \text{for} \quad R \left\{ \begin{array}{c} < \\ \equiv \\ > \end{array} \right\} \tilde{R} \, , \tag{20}$$

$$\text{and} \quad \lim_{\sigma \to 1} \Pi(R) = \lim_{\sigma \to -\infty} \Pi(R) = 0 \, . \tag{21}$$

Proof. See Appendix A.1. □

Result (12) states that the risk premium of income lottery R is strictly positive at all levels of resilience in between 0 and the maximum level of 1, and is zero at the extreme levels of 0 and 1. That is, income is risky at all levels of resilience in between 0 and 1; and only at the extreme levels of 0 and 1 does the income risk vanish, as in the case $R = 0$ the system will flip into the low-productivity domain with income y_L for certain, and at $R = 1$ the system will remain in the high-productivity domain with income y_L for certain.

As a consequence of Result (12), the risk premium varies with the level of resilience in a non-monotonic way (Figures 1 and 2, orange line). Result (13) states that there uniquely exists a level $\tilde{R}$ of the domain's resilience at which the risk premium is maximal, that is, the income lottery is most risky ($\tilde{R} = 0.647$ in Figure 1, $\tilde{R} = 0.794$ in Figure 2 left, $\tilde{R} = 0.004$ in Figure 2 right). For $R > \tilde{R}$ a marginal increase in resilience reduces the risk premium, and for $R < \tilde{R}$ a marginal increase in resilience raises the risk premium. This maximum-income-risk level of resilience, $\tilde{R}$ (Equation 14), is strictly in between $\underline{R}$ and 1, where $\underline{R} > 0$ denotes the level of resilience at which the probability of a system flip exactly equals the probability of the system not flipping (Result 15a).[11] So, the maximum-income-risk level of resilience $\tilde{R}$ is always strictly larger than the level of resilience $\underline{R}$, at which the future state of nature is maximally uncertain. Also, the range $(0, \tilde{R}]$ of low levels of resilience, for which the risk premium is strictly increasing with resilience, is non-empty.

Furthermore, the maximum-income-risk level of resilience $\tilde{R}$ increases with the degree of risk aversion ρ and the potential income loss Δy, it decreases with the ecosystem's elasticity σ (Result 15b).

The statement about the second derivative of the risk premium (Result 17) is rather technical, and will be needed for the proof of an important property of the insurance value in Proposition 1 below. Essentially, it states that there exists a domain of (positive) values of ecosystem elasticity, $0 \leq \sigma \leq \bar{\sigma}$, including the preeminent case of $\sigma = 0$, for which the risk premium is strictly concave over the entire range of resilience (Figure 1, orange line). This

[11] Note that $\underline{R}$, which is defined through $p(\underline{R}) = 1/2$, will be greater than (equal to, smaller than) $1/2$ for $\sigma < 0$ ($= 0, > 0$).

domain of ecosystem elasticities is bounded from below by zero, and from above by some maximal value $\bar{\sigma}$, which has the properties stated in Result (16): it increases with the risk-aversion-weighted potential income loss, $\rho\Delta y$, and for $\rho\Delta y$ going to infinity (zero) approaches the maximal ecosystem elasticity of one (zero).

The more risk-averse the ecosystem user is, the larger the perceived riskiness of the income lottery R and the larger the associated risk premium (Result 18). For a risk-neutral individual, on the other hand, the risk premium would be 0 for all R. Similarly, for the potential income loss Δy (Result 19): the risk premium raises with an increasing potential income loss Δy. For equal income levels in both stability domains, which means no income loss in case of a system flip ($\Delta y = 0$), the risk premium would be zero over the whole range of R.

Result (20) states that the risk premium increases (decreases) with the ecosystem's elasticity for levels of resilience below (above) the maximum-income-risk level of resilience, $\tilde{R}$. That is, in the range of resilience where the riskiness of income increases (decreases) with resilience, i.e. for $R < (>)\tilde{R}$ (cf. Result 13), an increase in ecosystem elasticity increases (decreases) the riskiness of income. This can be seen from comparing the orange lines in Figures 2 (left), 1 and 2 (right), as σ increases in this order. Ecosystem elasticity thus has the very same ambivalent role as ecosystem resilience for the riskiness of income. Result (21) states that the risk premium vanishes as the ecosystem's elasticity approaches either its maximum or its minimum value. The reason is that in either limiting case, according to model (4), the flip probability $p(R)$ does not depend on the level of resilience any more, except for the extreme levels of $R = 0$ (for $\sigma \to 1$) or $R = 1$ (for $\sigma \to -\infty$) where it jumps from one to zero or from zero to one, respectively. As a result, the risk premium is non-vanishing only at $R = 0$ (for $\sigma \to 1$) or $R = 1$ (for $\sigma \to -\infty$), but vanishes for all other levels of resilience.[12]

Having explored the effect of the ecosystem user's risk preferences and ecosystem properties on the risk premium of income lottery R, we can now discuss the insurance value of resilience as introduced in Definition 2.

Proposition 1

The insurance value of resilience, $I(R)$, is given by

$$I(R) = p'(R)\left\{\Delta y - \frac{1}{\rho}\frac{e^{\rho\Delta y}-1}{1+p(R)\left(e^{\rho\Delta y}-1\right)}\right\}, \tag{22}$$

[12]Note that an overall vanshing risk premium, except for either $R = 0$ (for $\sigma \to 1$) or $R = 1$ (for $\sigma \to -\infty$) is compatible with Result (20)'s statement that the risk premium increases with σ for $R < \tilde{R}$, because $\tilde{R}$ decreases with σ (Result 15).

which has the following properties:

(i) For all $R \in (0,1)$[13]

$$I(R) \left\{ \begin{matrix} \leq \\ \equiv \\ > \end{matrix} \right\} 0 \quad \text{for} \quad R \left\{ \begin{matrix} \leq \\ \equiv \\ > \end{matrix} \right\} \tilde{R} \text{, where } \tilde{R} \text{ is given by Equation (14)} . \tag{23}$$

(ii) The insurance value is globally increasing with resilience,

$$I(0) < I(1) , \tag{24}$$

in particular, it is strictly monotonically increasing depending on ecosystem elasticity:

$$I'(R) > 0 \quad \begin{cases} \text{for } R > \tilde{\tilde{R}} & \text{if} \quad \sigma < 0 \\ \text{for all } R \in (0,1) & \text{if and only if} \quad 0 \leq \sigma \leq \bar{\sigma} \\ \text{for } R < \tilde{\tilde{R}} & \text{if} \quad \sigma > \bar{\sigma} \end{cases} , \tag{25}$$

where $\bar{\sigma}$ and $\tilde{\tilde{R}}$ are as defined in Lemma 1(iii).

(iii) For all $R \in (0,1)$

$$\frac{dI(R)}{d\rho} \left\{ \begin{matrix} \leq \\ \equiv \\ > \end{matrix} \right\} 0 \quad \text{for} \quad R \left\{ \begin{matrix} \leq \\ \equiv \\ > \end{matrix} \right\} \tilde{R} \quad \text{and} \quad \lim_{\rho \to 0} I(R) = 0 , \tag{26}$$

$$\frac{dI(R)}{d\Delta y} \left\{ \begin{matrix} \leq \\ \equiv \\ > \end{matrix} \right\} 0 \quad \text{for} \quad R \left\{ \begin{matrix} \leq \\ \equiv \\ > \end{matrix} \right\} \tilde{R} \quad \text{and} \quad \lim_{\Delta y \to 0} I(R) = 0 , \tag{27}$$

$$\frac{dI(R)}{d\sigma} \left\{ \begin{matrix} < \\ = \\ > \\ = \\ < \end{matrix} \right\} 0 \quad \text{for} \quad \left\{ \begin{matrix} R < {}'R \\ R = {}'R \\ {}'R < R < R' \\ R = R' \\ R > R' \end{matrix} \right. \tag{28}$$

$$\text{and} \quad \lim_{\sigma \to 1} I(R) = \lim_{\sigma \to -\infty} I(R) = 0 , \tag{29}$$

where $\tilde{R}$ is as defined in Lemma 1(iii) and ${}'R < \tilde{R} < R'$.

Proof. See Appendix A.2. □

Result (23) states that the insurance value of resilience may be negative or positive, depending on the level of resilience R. If resilience is below the maximum-income-risk level $\tilde{R}$, an increases in resilience raises the risk premium (Result 13) and therefore, as the insurance value is defined as the reduction in the risk premium (Definition 2), resilience has a negative insurance value for all $R < \tilde{R}$. Only if $R > \tilde{R}$, an increase in resilience reduces the risk premium and the insurance value is positive (Figures 1 and 2, green line).

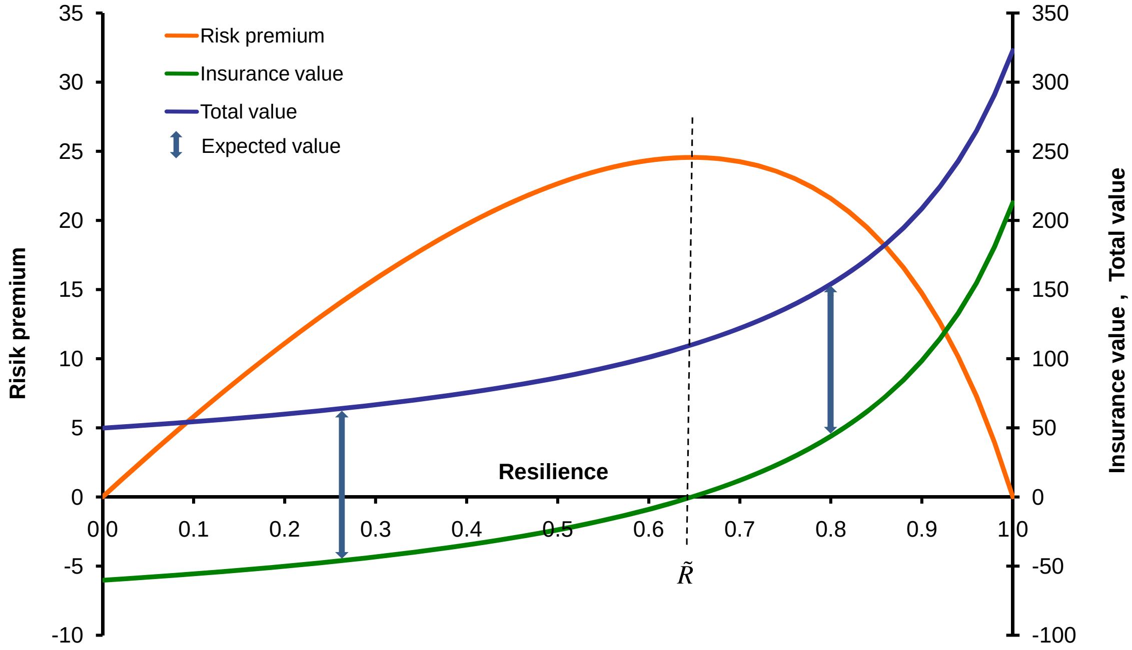

Figure 1: Risk premium (orange curve), insurance value (green curve), expected value (vertical distance between green and blue curves) and total value (blue curve) as a function of resilience for the case of intermediate ecosystem elasticity $0 \leq \sigma \leq \bar{\sigma}$. The dashed line marks the maximum-income-risk level of resilience $R = \tilde{R}$. (Parameter values: $\sigma = 0$, $\Delta y = 110$, $\rho = 0.017$)

Result (24) states that the insurance value of ecosystem resilience globally increases with the level of resilience: it is strictly higher for the maximum level of resilience than for zero resilience. Result (25) states that for a domain of (positive) values of ecosystem elasticity, $0 \leq \sigma \leq \bar{\sigma}$ (including the preeminent case of $\sigma = 0$), the insurance value of ecosystem resilience increases even strictly monotonically with the level of resilience (Figure 1, green line). Only as ecosystem elasticity σ turns negative or exceeds the threshold value $\bar{\sigma}$, it may happen that the insurance value locally decreases.[14] For negative ecosystem elasticity, $\sigma < 0$, it may

[13]For $\sigma = 0$, the statement about the sign of $I(R)$ holds also for $R = 0$ and $R = 1$. Yet, for $\sigma < 0$, one has $I(0) = 0$; for $\sigma \to 1$, one has $I(1) \to 0$.

[14]A parameter value of $\sigma < 0$ in Function (4) implies a relationship between p and R such that the first marginal units of resilience starting from $R = 0$ do not have any significant impact on the reduction of the flip probability p. Only increases in resilience close to the maximum level of $R = 1$ do significantly lower the flip probability p. For such ecosystems, the insurance value of resilience decreases for small levels of resilience and

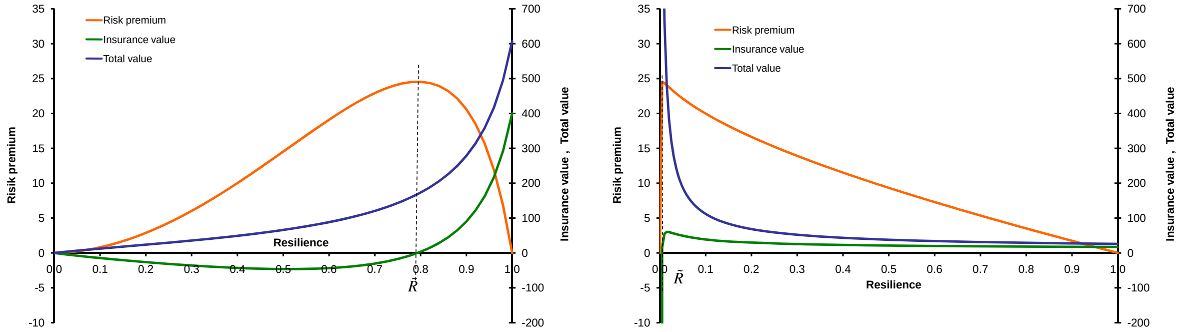

Figure 2: Risk premium (orange curve), insurance value (green curve), expected value (vertical distance between green and blue curves) and total value (blue curve) as a function of resilience for the two extreme cases of negative ecosystem elasticity ($\sigma < 0$, left) and very large positive ecosystem elasticity ($\sigma > \bar{\sigma}$, right). The dashed line marks the maximum-income-risk level of resilience $R = \tilde{R}$. (Parameter values, left: $\sigma = -0.88$; right: $\sigma = 0.92$; both: $\Delta y = 110$, $\rho = 0.017$)

be that the (negative) insurance value locally decreases at levels of resilience smaller than $\tilde{\tilde{R}}$ (Figure 2 left, green line); and for very large positive ecosystem elasticity, $\sigma > \bar{\sigma}$, it may be that the (positive) insurance value locally decreases at levels of resilience greater than $\tilde{\tilde{R}}$ (Figure 2 right, green line).

In economic terms, an increasing insurance value means that the higher the level of resilience, the more valuable – as an insurance – is a marginal increase in resilience. This is unusual and in contrast to normal economic goods, the marginal value of which decreases with their amount: normally, the more abundant a good, the less valuable the next marginal unit. Technically, the increasing marginal value of resilience comes about as the objective function, expected utility (5), when expressed as a function of the level of resilience, is non-concave in R.

Result (26) states how the ecosystem user's degree of risk-aversion affects the insurance value. If the ecosystem user was risk neutral ($\rho = 0$), the insurance value would vanish for

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increases for high levels of resilience close to $R = 1$ (Figure 2 left, green line). Conversely, a parameter value of σ close to its maximum value of 1 means that the first marginal unit of resilience has a huge impact on the reduction of the flip probability p, whereas all later units of resilience only have a negligible effect. Under such circumstances, the insurance value of resilience steeply increases in the vicinity of $R = 0$ from a negative value to its maximum (positive) value and then decreases with R (Figure 2 right, green line).

all levels of resilience R. With increasing risk-aversion, the insurance value increases for high levels of $R > \tilde{R}$, where it is positive, and decreases for low levels of $R < \tilde{R}$, where it is negative. Thus, the more risk-averse the ecosystem user is, the steeper the curve for I (Figure 1, green line). The same goes for the potential income loss Δy (Result 27). For equal income levels in both stability domains, which means no income loss in case of a system flip ($\Delta y = 0$), the insurance value would vanish for all levels of resilience R. With increasing potential income loss Δy, the I-curve gets steeper, as the insurance value decreases for $R < \tilde{R}$ and raises for $R > \tilde{R}$.

Also, $\tilde{R}$ shifts to the right with both increasing risk-aversion ρ and increasing potential income loss Δy. For very high values of ρ or Δy the I-curve appears to be sharply bended around $\tilde{R}$, since the insurance value raises faster with ρ or Δy in the range of $R > \tilde{R}$ than it decreases in the range of $R < \tilde{R}$.

Result (28) states that the insurance value decreases with increasing ecosystem elasticity for low and high levels of resilience, $R < {}'R < \tilde{R}$ and $R > R' > \tilde{R}$, and increases with increasing ecosystem elasticity in between, ${}'R < R < R'$. This can be seen from comparing the green lines in Figures 2 (left), 1 and 2 (right), as σ increases in this order. Result (29) states that the insurance value vanishes as the ecosystem's elasticity approaches either its maximum or its minimum value, which becomes plausible from the underlying property of the risk premium (Result 21). This can also be seen from comparing the green lines in Figures 2 (left and right) and 1.

Having discussed the effect of the ecosystem user's risk preferences and ecosystem properties on the insurance value of resilience, we now turn to discussing how the insurance value of ecosystem resilience relates to its total economic value (Definition 3).

Proposition 2

The total economic value of resilience, $V(R)$, is given by

$$V(R) = -p'(R)\frac{1}{\rho}\frac{e^{\rho\Delta y} - 1}{1 + p(R)\left(e^{\rho\Delta y} - 1\right)} \,, \tag{30}$$

which has the following properties:

(i)

$$V(R) \equiv -p'(R)\Delta y + I(R). \tag{31}$$

(ii)

$$V(R) \geq 0 \quad \text{for all } R. \tag{32}$$

Proof. See Appendix A.3. □

From Equation (31) it becomes obvious that the total economic value of resilience is the sum of two components: the expected increase in income due to a marginal increase in resilience, $-p'(R)\Delta y$, which is always positive,[15] and the insurance value of increased resilience, which may be negative or positive (cf. Proposition 1). This reflects the fact that an increase in ecosystem resilience has two effects on the ecosystem user's income: (i) it raises the expected income; (ii) it may raise or lower the riskiness of income, i.e. deviations from expected income. Thus, the total value of resilience is more than its insurance value, or, put the other way round, the insurance value is a value component over and above the expected value of resilience.

Figures 1 and 2 show the total economic value as a function of resilience (blue line). In the figures, the expected value of resilience, $-p'(R)\Delta y$, is just the vertical difference between the curves for I (green) and V (blue). Whereas the insurance value $I(R)$ of resilience may be positive or negative, depending on the level of resilience R, the expected value of resilience, $-p'(R)\Delta y$, is positive at all levels of resilience R.[16] As a consequence, for $R < \tilde{R}$ where the insurance value is negative, the total economic value of resilience is smaller than its expected value. Yet, at all levels of resilience the total value is positive (Result 32). That means, even if the insurance value should be negative, the mean-increasing value of resilience is large enough to offset this negative effect on the total value.

5 Discussion and conclusion

In this paper we have provided a conceptual clarification of the economic insurance value of ecosystem resilience. We have adopted a general and widely accepted definition of *insurance* as mitigation of the influence of uncertainty on a person's well-being (McCall 1987), and of *insurance value* as a reduction in the risk premium of the person's income risk lottery (Baumgärtner 2007). That way, we have clearly distinguished the insurance value of ecosystem resilience, which is due to its function to reduce the *riskiness* of income ("risk

[15]By Assumption (2), $p'(R) < 0$ for all $R \in (0,1)$.

[16]Note that for $\sigma = 0$, one has $p'(R) = -1 = \text{const.}$, so that the expected value of resilience does not depend on the level of resilience. That is, the vertical difference between the curves for I (green) and V (blue) in Figure 1 is constant.

mitigation"), from other components of its total economic value, which are due to resilience's function to raise the *expected* income from ecosystem services.

Our analysis has yielded several interesting and important results. First, the insurance value of resilience is negative for low levels of resilience and positive for high levels of resilience. That is, ecosystem resilience actually functions as an economic insurance, i.e. it reduces the riskiness of income from ecosystem services, only at sufficiently high levels of resilience; it does *not* function as an economic insurance but – just on the contrary – increases the riskiness of income at low levels of resilience.

Second, the (marginal) insurance value as well as the (marginal) total value of resilience increase globally with the level of resilience – for some ecosystem types (namely those with moderately positive elasticity) even monotonically: the higher the level of resilience, the more valuable is another unit of resilience. This is in contrast to normal economic goods, the (marginal) value of which *de*creases with their quantity. As unusual as this increasing-returns property may be for normal economic goods, it is not implausible and also known from other goods which are of systemic importance and thus give rise to a non-concavity in the social objective function, such as e.g. information (Radner and Stiglitz 1984) or biodiversity conservation (Hunter 2009). The management consequences for such non-convex ecological-economic systems are discussed e.g. by Dasgupta and Mäler (2003).

Third, the insurance value of resilience is one additive component of its total economic value. The other component is the rise in expected income due to a higher level of resilience. So, the insurance value of resilience, which is due to its risk-mitigation function, is a value component over and above the change in the expected value of the income lottery. While the former may be positive or negative, the latter is always positive, and the total economic value of resilience is always positive. One reason for distinguishing between the two value components of ecosystem resilience, and for studying the insurance value separately, might be that in an encompassing management-and-decision context the different functions of resilience may have different substitutes. For example, in many rural areas of developing countries there is no substitute for agro-ecosystem resilience in enhancing the mean level of farming income, but there is now more and more financial insurance available that serves as a substitute for resilience's function to mitigate income risks (Baumgärtner and Quaas 2008, Quaas and Baumgärtner 2008).

While we have made one specific assumption about risk preferences, i.e. constant absolute risk aversion, actually all of our results qualitatively hold more generally for all risk preferences satisfying the von-Neumann-Morgenstern axioms. These axioms, including continuity and context-independence, appear plausible for standard small-risk situations. But one may doubt that they adequately describe people's risk preferences when it comes to catastrophic (i.e. discontinuous) risk that irreversibly threatens the subsistence level of income, as it is the case for many threats to the resilience of life-supporting ecosystems. For such risks, it may be interesting to study how resilience provides insurance under, e.g., safety-first preferences (Roy 1952, Telser 1955, Kataoka 1963).

One general lesson from our analysis for further discussions of resilience as an insurance is that the concept of insurance fundamentally refers to both the objective characteristics of risk in terms of different possible states of nature and people's subjective risk preferences over these states. In particular, explicit reference to people's risk preferences is needed to meaningfully discuss insurance, to specify the economic insurance value of resilience, and to meaningfully distinguish the insurance value from other components of the total economic value of ecosystem resilience.

Acknowledgments

We are grateful to Martin Quaas for helpful discussion and comments.

Appendix

Throughout the appendix, we denote the risk-aversion-weighted income loss by $\lambda \equiv \rho \Delta y$.

A.1 Proof of Lemma 1

Explicating the general definition of the risk premium (Equation 7) by the CARA-utility function (6) yields

$$-e^{-\rho[(1-p(R))y_H + p(R)y_L - \Pi(R)]} = -(1-p(R))e^{-\rho y_H} - p(R)e^{-\rho y_L} , \qquad \text{(A.33)}$$

which can be rearranged into

$$e^{\rho \Pi(R)} = \frac{(1-p(R))e^{-\rho y_H} + p(R)e^{-\rho y_L}}{e^{-\rho[(1-p(R))y_H + p(R)y_L]}} . \qquad \text{(A.34)}$$

Using $\Delta y = y_H - y_L$, (A.34) can be solved for $\Pi(R)$, which leads to Result (11).

ad (i). Inserting $p(0) = 1$ or $p(1) = 0$ into (11) immediately yields $\Pi = 0$ (Result 12a). Strict positivity of $\Pi(R)$ for all $R \in (0,1)$ (Result 12b) can be demonstrated as follows. Note that

$$1 - p(R) > e^{p(R)\lambda} - p(R)e^{\lambda} \tag{A.35}$$

because the right hand side of this inequality approaches $1 - p(R)$ as $\lambda \to 0$ and strictly monotonically decreases with λ,

$$\frac{d}{d\lambda}\left[e^{p(R)\lambda} - p(R)e^{\lambda}\right] = p(R)\left(e^{p(R)\lambda} - e^{\lambda}\right) < 0\,,$$

since $\lambda > 0$ and $R \in (0,1)$, i.e. $0 < p(R) < 1$. Inequality (A.35) can be rearranged

$$\begin{aligned}
1 - p(R) &> e^{p(R)\lambda} - p(R)e^{\lambda} && \text{(A.36)}\\
1 + p(R)\left(e^{\lambda} - 1\right) &> e^{p(R)\lambda} && \text{(A.37)}\\
\ln\left[1 + p(R)\left(e^{\lambda} - 1\right)\right] &> p(R)\lambda && \text{(A.38)}\\
-p(R)\lambda + \ln\left[1 + p(R)\left(e^{\lambda} - 1\right)\right] &> 0\,, && \text{(A.39)}
\end{aligned}$$

which yields, after dividing by $\rho > 0$, Result (12)b.

ad (ii). Differentiating Result (11) with respect to R yields

$$\Pi'(R) = -\frac{p'(R)}{\rho}\left\{\lambda - \frac{e^{\lambda} - 1}{1 + p(R)\left(e^{\lambda} - 1\right)}\right\}\,. \tag{A.40}$$

By Assumption (2), $p'(R)$ is strictly negative for all $R \in (0,1)$. Hence, the sign of $\Pi'(R)$ is determined by the sign of the term in braces. For $R \to 0$, using $e^x > 1 + x$ for all $x \neq 0$, one has

$$\begin{aligned}
\lim_{R\to 0} \lambda - \frac{e^{\lambda} - 1}{1 + p(R)\left(e^{\lambda} - 1\right)} &= \lambda - \frac{e^{\lambda} - 1}{1 + \left(e^{\lambda} - 1\right)} = \lambda - 1 + e^{-\lambda}\\
&> \lambda - 1 + 1 - \lambda = 0 && \text{(A.41)}
\end{aligned}$$

For $R \to 1$, and again using $e^x > 1 + x$ for all $x \neq 0$, one has

$$\begin{aligned}
\lim_{R\to 1} \lambda - \frac{e^{\lambda} - 1}{1 + p(R)\left(e^{\lambda} - 1\right)} &= \lambda - \frac{e^{\lambda} - 1}{1 + 0} = \lambda - e^{\lambda} + 1\\
&< \lambda - 1 - \lambda + 1 = 0 && \text{(A.42)}
\end{aligned}$$

And $\Pi'(R) = 0$ for

$$\lambda - \frac{e^{\lambda} - 1}{1 + p(\tilde{R})\left(e^{\lambda} - 1\right)} = 0\,. \tag{A.43}$$

This can be uniquely solved for $R = \tilde{R}$ where $\tilde{R}$ is defined through

$$p(\tilde{R}) = \frac{1}{\lambda} - \frac{1}{e^{\lambda} - 1} \,, \tag{A.44}$$

which is equivalent to Result (14), since $p'(R) \neq 0$ for all $R \in (0,1)$. Pulling all this information together, from $\Pi'(0) > 0$ (A.41), $\Pi'(1) < 0$ (A.42), and $\Pi'(R) = 0$ if and only if $R = \tilde{R}$ (A.44), it follows that Result (13) holds.

In order to study the properties of $\tilde{R}$ (Equation 14) introduce

$$F(\lambda) = \frac{1}{\lambda} - \frac{1}{e^{\lambda} - 1} \,, \tag{A.45}$$

so that (A.44) and (14) can be rewritten as

$$p(\tilde{R}) \equiv F(\lambda) \quad \text{and} \quad \tilde{R} \equiv p^{-1}(F(\lambda)) \,. \tag{A.46}$$

Note that

$$\lim_{\lambda \to 0} F(\lambda) = \lim_{\lambda \to 0} \frac{e^{\lambda} - 1 - \lambda}{\lambda(e^{\lambda} - 1)} = \lim_{\lambda \to 0} \frac{e^{\lambda} - 1}{(1+\lambda)e^{\lambda} - 1} = \lim_{\lambda \to 0} \frac{1}{2+\lambda} = \frac{1}{2} \,, \tag{A.47}$$

(apply l'Hôpital's rule twice)

$$\lim_{\lambda \to +\infty} F(\lambda) = \lim_{\lambda \to +\infty} \frac{1}{\lambda} - \lim_{\lambda \to +\infty} \frac{1}{e^{\lambda} - 1} = 0 \,, \tag{A.48}$$

$$F'(\lambda) = -\frac{1}{\lambda^2} + \frac{e^{\lambda}}{(e^{\lambda} - 1)^2} = \frac{1}{e^{\lambda} + e^{-\lambda} - 2} - \frac{1}{\lambda^2} < 0 \quad \text{for all } \lambda \,, \tag{A.49}$$

(as, through Taylor expansion, $e^{\lambda} = \sum_{n=0}^{\infty} \frac{\lambda^n}{n!}$ and therefore

$e^{\lambda} + e^{-\lambda} - 2 = \lambda^2 + \sum_{n=1}^{\infty} \frac{2\lambda^{2n}}{(2n)!} > \lambda^2$ for all λ)

$$F(\lambda) > 0 \quad \text{for all } \lambda \,. \tag{A.50}$$

(follows immediately from A.47–A.49)

From (A.50) it follows immediately that $p(\tilde{R}) = F(\lambda) > 0$ for all λ, which implies, with $p'(R) < 0$ for all R, that $\tilde{R} < 1$ for all λ. On the other hand, from (A.47) and (A.49) one has that $F(\lambda) < 1/2$ for all $\lambda > 0$. Hence, $p(\tilde{R}) = F(\lambda) < 1/2$ for all λ, which implies, with $p'(R) < 0$ for all $R \in (0,1)$, that $\tilde{R} > \underline{R}$ for all λ. This establishes Result (15a).

With (A.46), Assumption 2 ($p'(R) < 0$ for all $R \in (0,1)$) and Property (A.49), it follows that

$$\frac{d\tilde{R}}{d\lambda} = \frac{1}{p'(\tilde{R})} F'(\lambda) > 0 \,. \tag{A.51}$$

From that, with $\lambda \equiv \rho \Delta y$ it follows immediately that $d\tilde{R}/d\rho > 0$ and $d\tilde{R}/d\Delta y > 0$ (Result 15b). Using (4) and (A.46), $\tilde{R}$ can be rewritten as

$$\tilde{R} = p^{-1}(F(\lambda)) = [1 - F(\lambda)]^{\frac{1}{1-\sigma}} , \tag{A.52}$$

from which it it follows that

$$\frac{d\tilde{R}}{d\sigma} = [1 - F(\lambda)]^{\frac{1}{1-\sigma}} \ln[1 - F(\lambda)] \frac{1}{(1-\sigma)^2} < 0 , \tag{A.53}$$

since $0 < F(\lambda) < 1/2$ (from A.47–A.50) and $\sigma < 1$ (by Assumption 4) imply that the first and third factors are strictly positive and the second is strictly negative.

ad (iii). Differentiate (A.40) again with respect to R:

$$\Pi''(R) = -\frac{1}{\rho}\left\{ p''(R)\left[\lambda - \frac{e^{\lambda}-1}{1+p(R)(e^{\lambda}-1)}\right] + \left[p'(R)\frac{e^{\lambda}-1}{1+p(R)(e^{\lambda}-1)}\right]^2 \right\} . \tag{A.54}$$

Under Assumption (4) one has

$$p(R) = 1 - R^{1-\sigma} \tag{A.55}$$

$$p'(R) = -(1-\sigma)R^{-\sigma} \tag{A.56}$$

$$p''(R) = \sigma(1-\sigma)R^{-\sigma-1} \tag{A.57}$$

Inserting (A.55)–(A.57) into (A.54) yields an explicit equation for $\Pi''(R)$ in the elementary parameters of the model, σ, ρ and Δy. Systematic numerical simulation of this equation for all $-\infty < \sigma < 1$ and for various $\rho, \Delta y > 0$ yields Results (16) and (17).

ad (iv). By definition, the risk premium is zero for a risk-neutral decision-maker ($\rho = 0$), and is known to increase with her degree of risk-aversion ρ (e.g. Gravelle and Rees 2004: 463), which yields Result (18).

Setting $\Delta y = 0$ in Expression (11) for $\Pi(R)$ obviously yields $\Pi(R) \equiv 0$. That the risk premium increases with Δy can be seen from the first derivative of $\Pi(R)$ with respect to Δy:

$$\begin{aligned} \frac{d\Pi(R)}{d\Delta y} &= p(R)\left[\frac{e^{\lambda}}{1+p(R)\left(e^{\lambda}-1\right)} - 1\right] \\ &= p(R)\left[\frac{1}{p(R)+(1-p(R))e^{-\lambda}} - 1\right] . \end{aligned} \tag{A.58}$$

As $e^{-\lambda} < 1$ for $\lambda > 0$, and $0 < p(R) < 1$ for $R \in (0,1)$, the denominator in the fraction is strictly smaller than 1, so that the term in brackets is strictly positive and the whole expression is strictly positive, which yields Result (19).

From Result (11) it follows that

$$\frac{d\Pi(R)}{d\sigma} = -\frac{1}{\rho}\left\{\lambda - \frac{e^{\lambda}-1}{1+p(R)\left(e^{\lambda}-1\right)}\right\}\frac{dp(R)}{d\sigma} \, . \tag{A.59}$$

From (A.40) it is apparent that

$$\left\{\lambda - \frac{e^{\lambda}-1}{1+p(R)\left(e^{\lambda}-1\right)}\right\} = -\rho\frac{\Pi'(R)}{p'(R)} \, , \tag{A.60}$$

so that (A.59) becomes

$$\frac{d\Pi(R)}{d\sigma} = \frac{\Pi'(R)}{p'(R)}\frac{dp(R)}{d\sigma} \, . \tag{A.61}$$

As $p'(R) < 0$ for all $R \in (0,1)$, and, with Assumption (4), $dp(R)/d\sigma < 0$ for all $R \in (0,1)$, the sign of $d\Pi(R)/d\sigma$ is determined by the sign of $\Pi'(R)$. With Result (13), Result (20) then follows immediately.

Result (21) follows from Result (11) and noting that model (4) implies

$$\lim_{\sigma\to 1} p(R) = 0 \quad \text{as well as} \quad \lim_{\sigma\to -\infty} p(R) = 1 \quad \text{for all } R \in (0,1) \, . \tag{A.62}$$

A.2 Proof of Proposition 1

Differentiating $-\Pi(R)$ with respect to R immediately yields Result (22).

ad (i). Result (23) follows immediately from Definition (8) and Result (13).

ad (ii). Result (24) can be demonstrated by noting that Result (22) implies

$$I(0) = p'(0)\frac{1}{\rho}\left(\lambda - 1 + e^{-\lambda}\right) \quad \text{and} \quad I(1) = p'(1)\frac{1}{\rho}\left(\lambda - e^{\lambda} + 1\right) \, , \tag{A.63}$$

where

$$\lambda - 1 + e^{-\lambda} > 0 \quad \text{and} \quad \lambda - e^{\lambda} + 1 < 0 \, , \tag{A.64}$$

since $e^x > 1 + x$ for all $x > 0$. Under Assumption (4), one has (A.56), so that

$$\left\{\begin{array}{l} p'(0) = 0 \ \text{ and } \ p'(1) < 0 \\ p'(0) < 0 \ \text{ and } \ p'(1) < 0 \\ p'(0) < 0 \ \text{ and } \ p'(1) \leq 0^{17} \end{array}\right\} \quad \text{if} \quad \left\{\begin{array}{l} \sigma < 0 \\ \sigma = 0 \\ \sigma > 0 \end{array}\right\} . \tag{A.65}$$

Combining (A.63)–(A.65), one has

$$\left\{\begin{array}{l} I(0) = 0 \ \text{ and } \ I(1) > 0 \\ I(0) < 0 \ \text{ and } \ I(1) > 0 \\ I(0) < 0 \ \text{ and } \ I(1) \geq 0^{18} \end{array}\right\} \quad \text{if} \quad \left\{\begin{array}{l} \sigma < 0 \\ \sigma = 0 \\ \sigma > 0 \end{array}\right\} , \tag{A.66}$$

[17] $p'(1) < 0$ for $\sigma < 1$, and $p'(1) \to 0$ as $\sigma \to 1$.

which means that, in any case, $I(0) < I(1)$, which is Result (24). Result (25) follows immediately from Definition (8) and Result (17).

ad (iii). Results (26), (27), (28) follow from Definition (8), the fact that the function $\Pi(R)$ is continuous and differentiable, and Results (12), (18), (19), (20). In, addition, systematic numerical simulations of Equation (23), using model (4), for all $-\infty < \sigma < 1$ and for various $\rho, \Delta y > 0$ have been employed to demonstrate Result (28). Result (29) follows from Definition (8), the fact that the function $\Pi(R)$ is continuous and differentiable, and Result (21).

A.3 Proof of Proposition 2

Explicating the general Definition of the ecosystem user's WTP (Equation 10) by the CARA-utility function (6) yields

$$\begin{aligned} & -(1-p(R))e^{-\rho y_H} - p(R)e^{-\rho y_L} && \text{(A.67)} \\ = & -\left[p(R+\Delta R)e^{-\rho(y_L - WTP(\Delta R))} + (1-p(R+\Delta R))e^{-\rho(y_H - WTP(\Delta R))}\right] && \text{(A.68)} \\ = & -e^{\rho WTP(\Delta R)}\left[p(R+\Delta R)e^{-\rho y_L} + (1-p(R+\Delta R))e^{-\rho y_H}\right] . && \text{(A.69)} \end{aligned}$$

Rearranging leads to

$$-e^{\rho WTP(\Delta R)} = \frac{-(1-p(R))e^{-\rho y_H} - p(R)e^{-\rho y_L}}{[p(R+\Delta R)e^{-\rho y_L} + (1-p(R+\Delta R))e^{-\rho y_H}]} . \tag{A.70}$$

Solving for $WTP(\Delta R)$, using $\Delta y = y_H - y_L$ and $\lambda \equiv \rho \Delta y$, yields

$$WTP(\Delta R) = \frac{1}{\rho} \ln \frac{(1-p(R)) + p(R)e^{\lambda}}{(1-p(R+\Delta R)) + p(R+\Delta R)e^{\lambda}} . \tag{A.71}$$

[18] $I(1) > 0$ for $\sigma < 1$, and $I(1) \to 0$ as $\sigma \to 1$.

Using (A.71) in Definition 3 and applying l'Hôpital's rule, one has

$$
\begin{aligned}
V(R) &= \frac{1}{\rho}\lim_{\Delta R\to 0}\frac{\ln\frac{(1-p(R))+p(R)e^{\lambda}}{(1-p(R+\Delta R))+p(R+\Delta R)e^{\lambda}}}{\Delta R} && \text{(A.72)}\\
&= \frac{1}{\rho}\lim_{\Delta R\to 0}\frac{(1-p(R+\Delta R))+p(R+\Delta R)e^{\lambda}}{(1-p(R))+p(R)e^{\lambda}}\\
&\qquad\times\frac{d}{d\Delta R}\left[\frac{1-p(R)+p(R)e^{\rho\Delta y}}{1-p(R+\Delta R)+p(R+\Delta R)e^{\lambda}}\right] && \text{(A.73)}\\
&= \frac{1}{\rho}\lim_{\Delta R\to 0}\frac{d}{d\Delta R}\left[\frac{1-p(R)+p(R)e^{\lambda}}{1-p(R+\Delta R)+p(R+\Delta R)e^{\lambda}}\right] && \text{(A.74)}\\
&= -\frac{1}{\rho}\lim_{\Delta R\to 0}\frac{\left[1-p(R)+p(R)e^{\lambda}\right]\left[-p'(R+\Delta R)+p'(R+\Delta R)e^{\lambda}\right]}{\left[1-p(R+\Delta R)+p(R+\Delta R)e^{\lambda}\right]^2} && \text{(A.75)}\\
&= -\frac{p'(R)}{\rho}\frac{e^{\lambda}-1}{1+p(R)\left(e^{\lambda}-1\right)}\,. && \text{(A.76)}
\end{aligned}
$$

ad (i). Rearranging Result (30), and using Result (22), immediately yields Result (31).

ad (ii). Expression (A.76) for V is non-negative for all R, since $-p'(R)$ is non-negative and the term $(e^{\lambda}-1)$ is strictly positive for all R. Hence, Result (32) holds.

References

Arrow, K.J. (1965). Aspects of the theory of risk-bearing. Yrjö Jahnsson Lecture. Reprinted with modifications in K.J. Arrow, *Essays in the Theory of Risk Bearing*, Markham, Chicago, 1971.

Baumgärtner, S. (2007). The insurance value of biodiversity in the provision of ecosystem services. *Natural Resource Modeling*, **20**(1): 87–127.

Baumgärtner, S. and M.F. Quaas (2008). Agro-biodiversity as natural insurance and the development of financial insurance markets, In A. Kontoleon, U. Pascual and M. Smale (eds), *Agrobiodiversity, Conservation and Economic Development*, Routledge, London, pp. 293—317.

Besley, T. (1995). Savings, credit and insurance, In H.B. Chenery and T.N. Srinivasan (eds), *Handbook of Development Economics, Vol. III*, North Holland, Amsterdam, pp. 2123–2207.

Brand, F. S., and K. Jax (2007). Focusing the meaning(s) of resilience: resilience as a descriptive concept and a boundary object. *Ecology and Society*, **12**(1): 23.

Caillaud, B., G. Dionne and B. Jullien (2000). Corporate insurance with optimal financing contracts. *Economic Theory*, **16**: 77–105.

Carpenter, S.R., B. Walker, J.M. Anderies and N. Abel (2001). From metaphor to measurement: resilience of what to what? *Ecosystems*, **4**: 765–781.

Dasgupta, P. (1993). *An Inquiry into Well-Being and Destitution*. Clarendon Press, Oxford.

Dasgupta, P. and K.-G. Mäler (eds) (2003). *The economics of non-convex ecosystems*. Special issue, *Environmental and Resource Economics*, **26**(4).

Ehrlich, J. and G.S. Becker (1972). Market insurance, self-insurance and self-protection. *Journal of Political Economy*, **80**: 623–648.

Folke, C., S. Carpenter, T. Elmqvist, L. Gunderson, C. S. Holling, B. Walker, J. Bengtsson, F. Berkes, J. Colding, K. Danell, M. Falkenmark, L. Gordon, R. Kasperson, N. Kautsky, A. Kinzig, S. Levin, K.-G. Mäler, F. Moberg, L. Ohlsson, P. Olsson, E. Ostrom, W. Reid, J. Rockström, H. Savenije, and U. Svedin (2002). Resilience and sustainable development: building adaptive capacity in a world of transformations. *Scientific Background Paper on Resilience for the process of The World Summit on Sustainable Development on behalf of The Environmental Advisory Council to the Swedish Government.*

Freeman III, A.M. (2003). *The Measurement of Environmental and Resource Values. Theory and Methods*. Second edition. Resources for the Future, Washington DC.

Gravelle, H. and R. Rees (2004). *Microeconomics*. Third edition. Prentice Hall, Harlow.

Gunderson, L.H. and L. Pritchard Jr. (eds) (2002). *Resilience and the Behavior of Large-Scale Systems*. Island Press, Washington DC.

Hicks, J.R. (1943). The four consumer surpluses. *Review of Economic Studies*, **11**(1): 31–41.

Holling, C.S. (1973). Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics*, **4**: 1–23.

Holling, C.S., S. Carpenter, W.A. Brock and L.H. Gunderson (2002). Discoveries for sustainable futures. In L.H. Gunderson and C.S. Holling (eds), *Panarchy. Understanding Transformations in Human and Natural Systems*, Island Press, Washington DC, pp. 395–418.

Hunter, G. (2009). Noah's nonconcavity: on the existence of nontrivial interior solutions to the problem of cost-effective conservation planning. *Natural Resource Modeling*, **22**: 26–41.

Kataoka, S. (1963). A stochastic programming model. *Econometrica*, **31**: 181–196.

Knight, F. (1921). *Risk, Uncertainty and Profit*. Houghton Mifflin, Boston.

Kreps, D.M. (1990). *A Course in Microeconomic Theory*. Harvester Wheatsheaf, New York.

Levin, S.A., S. Barrett, S. Aniyar, W. Baumol, C. Bliss, B. Bolin, P. Dasgupta, P. Ehrlich, C. Folke, I.-M. Gren, C.S. Holling, A.-M. Jansson, B.-O. Jansson, D. Martin, K.-G. Mäler, C. Perrings and E. Sheshinsky (1998). Resilience in natural and socioeconomic systems, *Environment and Development Economics*, **3**(2): 222–234.

Mäler, K.-G. (2008). Sustainable development and resilience in ecosystems. *Environmental and Resource Economics*, **39**(1): 17–24.

Mäler, K.-G., S. Aniyar and Å. Jansson (2009). Accounting for ecosystems. *Environmental and Resource Economics*, **42**(1): 39–51.

Mäler, K.-G., C.-Z. Li and G. Destouni (2007). Pricing resilience in a dynamic economy-environment system: a capital-theoretic approach. *Beijer Discussion Paper* No. 208 (available at http://www.beijer.kva.se/discussions.php).

Mayers, D. and C.W. Smith Jr. (1990). On the corporate demand for insurance: evidence from the reinsurance market. *Journal of Business*, **63**(1): 19–40.

McCall, J.J. (1987). Insurance. In J. Eatwell, M. Milgate and P. Newman (eds), *The New Palgrave. A Dictionary of Economics*, Volume 2, Palgrave, Houndmills, pp. 868–870.

Perrings, C. (1995). Biodiversity conservation as insurance. In T. Swanson (ed.), *The Economics and Ecology of Biodiversity Decline*, Cambridge University Press, Cambridge, pp. 69–77.

Perrings, C. und D.I. Stern (2000). Modelling loss of resilience in agroecosystems: rangelands in Botswana. *Environmental and Resource Economics*, **16**: 185–210.

Perrings, C. und B.W. Walker (1995). Biodiversity loss and the economics of discontinuous change in semiarid rangelands. In: C. Perrings, K.-G. Mäler, C. Folke, C.S. Holling und B.-O. Jansson (eds), *Biodiversity Loss. Economic and Ecological Issues*, Cambridge University Press, Cambridge, pp. 190–210.

Quaas, M.F. and S. Baumgärtner (2008). Natural vs. financial insurance in the management of public-good ecosystems. *Ecological Economics*, **65**(2): 397–406.

Pratt, J.W. (1964). Risk aversion in the small and in the large. *Econometrica*, **32**(1–2): 122–136.

Radner, R. and J.E. Stiglitz (1984). A nonconcavity in the value of information. In M. Boyer and R.E. Kihlstrom (eds), *Bayesian models in economic theory*, North-Holland, New York, pp. 33–52.

Roy, A.D. (1952). Safety first and the holding of assets. *Econometrica*, **20**(3): 431–449.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker (2001). Catastrophic shifts in ecosystems. *Nature*, **413**: 591–596.

Telser, L.G. (1955). Safety first and hedging. *The Review of Economic Studies*, **23**: 1-–16.

[UNCCD] Secretariat of the United Nations Convention to Combat Desertification (2005). Fact Sheet 3 – The Consequences of Desertification. http://www.unccd.int.

[UNO] United Nations Organisation (2002). *United Nations Convention to Combat Desertification*. http://www.unccd.int.

Varian, H.R. (1992). *Microeconomic Analysis*. 3rd ed., W.W. Norton, New York and London.

Walker, B., L. Pearson, M. Harris, K.-G. Mäler, C.-Z. Li and R. Biggs (2007). Incorporating resilience in the assessment of inclusive wealth: an example from South East Australia. *Beijer Discussion Paper* No. 209 (available at http://www.beijer.kva.se/discussions.php).

Chapter 4: Consumer preferences determine resilience of ecological-economic systems

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Abstract: We perform a model analysis to study the origins of limited resilience in coupled ecological-economic systems. We demonstrate that under open access to ecosystems for profit-maximizing harvesting forms, the resilience properties of the system are essentially determined by consumer preferences for ecosystem services. In particular, we show that complementarity and relative importance of ecosystem services in consumption may significantly decrease the resilience of (almost) any given state of the system. We conclude that the role of consumer preferences and management institutions is not just to facilitate adaptation to, or transformation of, some natural dynamics of ecosystems. Rather, consumer preferences and management institutions are themselves important determinants of the fundamental dynamic characteristics of coupled ecological-economic systems, such as limited resilience.

JEL-Classification: Q01, Q20, Q57

Keywords: consumption, ecological-economic systems, ecosystem services, natural resource management, preferences, resilience

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1 Introduction

Natural systems that are used and managed by humans for the ecosystem services they provide may exhibit non-trivial dynamics. This makes the long-term conservation and sustainable use of such systems a huge challenge.

In particular, a coupled ecological-economic system may be characterized by limited resilience (Holling 1973). That is, it exhibits multiple stability domains (“basins of attraction”) that differ in fundamental system structure and controls as well as in the level and quality of ecosystem services provided to humans. These stability domains are separated by thresholds in the system's state variables. Theoretically, the resilience of the system in some state can be measured by the stability basin’s width – also known as its “latitude” (Walker et al. 2004). As a result of exogenous natural disturbances or ill-adapted human interference with the system, the system may flip from one stability domain into another one with different basic functions and controls (Holling 1973, Levin et al. 1998, Carpenter et al. 2001, Scheffer et al. 2001). Examples encompass a diverse set of ecosystem types that are highly relevant for economic use, such as boreal forests, semi-arid rangelands, wetlands, shallow lakes, coral reefs, or high-seas fisheries (Gunderson and Pritchard 2002).

As the system undergoes a regime shift and flips from one basin of attraction with more desirable ecosystem service provision (from the anthropocentric point of view based on valuation of ecosystem services) to a basin of attraction with less desirable ecosystem service provision, humans will assess this change as a deterioration in ecosystem service provision, or even as a “catastrophic” shift (Scheffer et al. 2001). Such system flips may threaten the intertemporal efficiency of resource management and the intergenerational equity of ecosystem services use from this system, and may thus impair a sustainable development (Arrow et al. 1995, Perrings 2001, Perrings 2006, Mäler 2008, Derissen et al. 2011).

Many studies analyzing the role of resilience for the long-term development of coupled ecological-economic systems explain limits to resilience, i.e. the existence of multiple and limited basins of attraction in a dynamic system, by *natural* characteristics of the system which exist prior to any human interference with the system, such as e.g. ecological properties of shallow lakes or the interaction between grass and shrub species in semi-arid rangelands. Human management of the system then has to be adapted to this natural characteristic, or transform the dynamic characteristics of the natural system, so as to achieve sustainability (e.g. Berkes and Folke 1998, Gunderson et al. 2001, Berkes et al. 2002). How the stability landscape of a coupled ecological-economic system is determined by, and may be changed through, institutional arrangements has been analyzed by e.g. Horan et al. (2011).

In this paper, we point out that under open access to ecosystems for profit-maximizing harvesting firms – which describes many exploited ecosystems – consumer preferences may induce similar characteristics into a dynamic system. Here, the term "consumer preferences" denotes the preferences that consumers hold over the different commodities that are directly consumed, including ecosystem services, based on the individual utility conferred by such consumption – in contrast to preferences for particular ecosystem states or properties that may indirectly result from consumers' behaviour ("green consumerism").

A decrease in the resilience of some desired state in a coupled ecological-economic system, i.e. a decrease in the corresponding stability basin's width or an increase in the number of alternative basins of attraction, may arise due to particular consumer preferences for ecosystem services, even if the underlying ecological processes are rather simple and management institutions are stable. To demonstrate this, we present a model of a simple multi-species ecosystem that may be harvested for economic purposes by profit-maximizing resource-extracting firms. We model biological interactions as competition between the species. We show that multiple basins of attraction may be introduced into the system's dynamics, and, thus, the width of some desired state's basin of attraction may decrease, solely as a consequence of changes in consumer preferences. We also analyze how the resilience properties of the coupled ecological-economic system depend on the consumers' preferences for ecosystem services and on the degree of biological interaction between species. Thus, we clearly distinguish the effects of economic use and consumer preferences from the effect of ecological interactions on the system's resilience properties.

2 Model

Consider the following model, which gives a highly stylized description of dynamic ecological-economic systems. Society consists of n identical individuals whose well-being derives from the consumption of manufactured goods (y) and two different ecosystem services, say fish (c) and timber (h). Assume that all three goods are essential for individual well-being and that the two ecosystem services are complementary in human well-being. Then, a representative household's well-being can be described by the utility function

$$u(y, c, h) = y^{1-\alpha} \left[c^{\frac{\sigma-1}{\sigma}} + h^{\frac{\sigma-1}{\sigma}} \right]^{\alpha \frac{\sigma}{\sigma-1}} \quad (1)$$

.

Parameter α (with $0<\alpha<1$) expresses the representative household's dependence on ecosystem services, where a higher value of α describes a higher relative importance of ecosystem services for the household's utility. Parameter σ (with $\sigma>0$) represents the elasticity of substitution between the consumption of fish and timber: a smaller value of σ implies a higher degree of complementarity of fish and timber. In the limit $\sigma\to 0$, fish and timber would be perfect complements and utility would be determined by the relatively scarcer ecosystem service only. In the opposite limit $\sigma\to\infty$, fish and timber would be perfect substitutes and utility would be determined only by the sum of both ecosystem services.

The dynamics of the stocks of fish (x) and wood (w) is described by the following system of differential equations

$$\frac{dx}{dt} = f(x, w) - C \quad (2)$$

,

$$\frac{dw}{dt} = g(w, x) - H \quad (3)$$

,

where the functions $f(x,w)$ and $g(w,x)$ describe the intrinsic growth of the stocks of fish and wood, and C and H denote the aggregate amounts of fish and timber harvested. For expositional simplicity, we specify $f(x,w)$ and $g(w,x)$ in a standard manner as logistic growth functions with competitive interaction between species (e.g. Scheffer 2009: Appendix A4):

$$f(x,w) = \rho_x \left(1 - \frac{x + \gamma_x w}{\kappa_x}\right) x \quad , \tag{4}$$

$$g(w,x) = \rho_w \left(1 - \frac{w + \gamma_w x}{\kappa_w}\right) w \quad , \tag{5}$$

where ρ_i denotes the intrinsic growth rate and κ_i the carrying capacity of the stocks of fish (i=x) and wood (i=w), respectively, and γ_i denotes the impact of competition on species i (i=x,w) from the other species. The specification of logistic growth functions and this particular form of biological interaction is by no means essential for the results derived below. But using a well-known functional form of the biological growth functions $f(x,w)$ and $g(w,x)$ helps to clarify the argument and to highlight the role of consumer preferences for the dynamics of the ecological-economic system.

The consumption of ecosystem services relies on the harvest of fish and timber. There are m_x identical fish-harvesting firms and m_w identical timber-harvesting firms, where the exact numbers are endogenously determined according to market conditions in these two sectors. Let e_x and e_w denote the effort, measured in units of labor, spent by some representative fish-harvesting-firm and some representative timber-harvesting-firm. The maximum amounts of fish and timber that can be harvested from the respective stocks by individual firms are described by Gordon-Schaefer production functions

$$c^{\text{prod}} = \nu_x \, x \, e_x \quad , \tag{6}$$

$$h^{\text{prod}} = \nu_w \, w \, e_w \; , \tag{7}$$

where v_x and v_w denote the productivity of harvesting fish and timber, respectively. Then, the aggregate amounts of fish and timber harvested are simply

$$C = m_x\, c^{\text{prod}} \quad , \tag{8}$$

$$H = m_w\, h^{\text{prod}} \quad . \tag{9}$$

Assume that each household inelastically supplies one unit of labor, so that total labor supply of the economy is equal to human population size n. Households work either in one of the resource harvesting sectors or in the manufactured-goods sector. Assuming that labor is the only factor input for the production of manufactured goods, and that production is through a constant-returns-to-scale technology, i.e. each unit of labor produces $\omega > 0$ units of output, aggregate output of manufactured goods is

$$Y = \omega\ (n - m_x\, e_x - m_w\, e_w) \quad . \tag{10}$$

3 Analysis

In order to show that under open access to ecosystems for profit-maximizing harvesting firms consumer preferences about ecosystem services essentially matter, we analyze the resilience properties of the coupled ecological-economic system for different scenarios in terms of resource-management and consumer preferences. To this end we employ local and global stability analysis based on graphical representation of the system's dynamics in state space. The analytics behind the graphical representation are derived in the Appendix.

3.1 Natural dynamics

In the absence of any resource harvesting by society, the system's dynamics is completely determined by the natural dynamics of the two resources stocks of fish and wood, described by Equations (2)–(5) with *C=H=0*. This scenario goes back to Lotka

(1932) and Volterra (1926) and sets the benchmark against which we then study the influence of harvesting and consumer preferences on resilience.

If the dynamics of the two resource stocks are independent of each other, i.e. if there is no inter-species competition (γ_x=γ_w=0), both stocks converge to their respective carrying capacities. The isoclines dx/dt=0 and $\dot{w}$ = 0 thus are the straight lines with w = κ_w and x = κ_x, respectively. This dynamics is represented by the upper phase diagram in Figure 1 for parameter values ρ_x=ρ_w=0.5 and κ_x=κ_w=1. The green line is the isocline for dx/dt=0, the red line is the isocline for dw/dt=0. Below (above) the dx/dt= 0-isocline the dynamics is characterized by dx/dt>0 (<0). Likewise, left (right) of the dw/dt= 0-isocline the dynamics is characterized by dw/dt> 0 (<0). In each segment of state space, the green and red arrows indicate this direction of dynamics. At the intersection of the isoclines (point D: x=1, w=1), one has dx/dt=dw/dt=0 and the arrows indicate that this is a stable equilibrium.

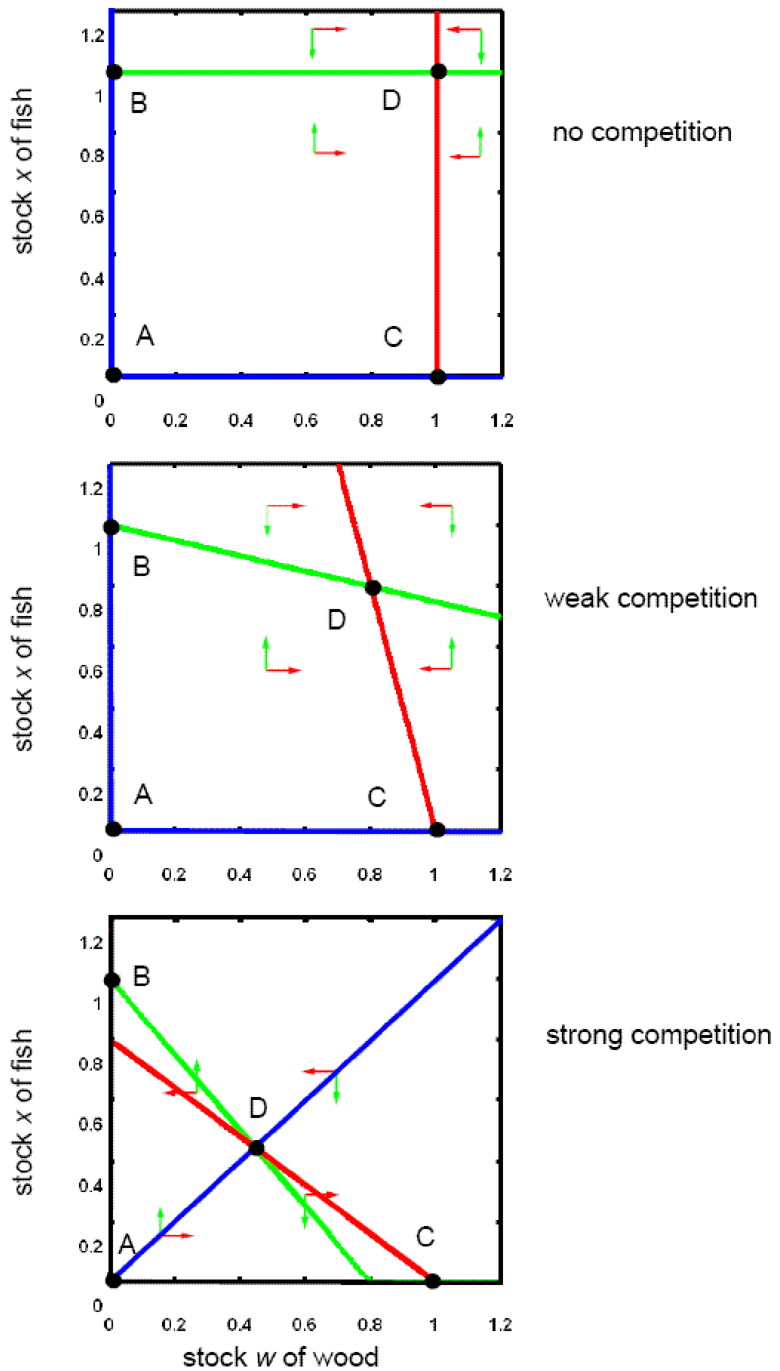

Figure 1: Phase diagrams in state space for the ecosystem's natural dynamics without any harvesting ($C=H=0$). Dynamics is characterized by $dx/dt>0$ (<0) below (above) the green line, and $dw/dt>0$ (<0) left (right) of the red line. Blue lines indicate saddlepaths. The upper diagram displays the case of independent species ($\gamma_x=\gamma_w=0$). In the middle diagram inter-species competition is weaker than intra-species competition ($\gamma_x=\gamma_w=0.25$), and in the lower diagram, inter-species competition is stronger than intra-species competition ($\gamma_x=\gamma_w=1.25$). Parameter values for all diagrams: $\rho_x=\rho_w=0.5$, $\kappa_x=\kappa_w=1$.

Other than D, the system has three more equilibria: A (x=w=0), B (x=1, w=0) and C (x=0, w=1). In the absence of inter-species competition (γ_x=γ_w=0), it is obvious from the state-space representation (Figure 1, upper diagram) that A is an unstable equilibrium, while B and C are locally saddlepoint-stable equilibria. The basin of attraction corresponding to the only stable equilibrium, D, comprises the entire state space with the exception of the axes (x=0, w≥0) and (x≥0, w=0). >From any system state in this domain will the system automatically converge towards equilibrium D. So, equilibrium D is (almost) globally stable – where the "almost" refers to the exception of the axes. In terms of resilience, (almost) every state of the natural system is therefore characterized by (almost) unlimited resilience.

If the system exhibits inter-species competition, neither stock reaches its full carrying capacity due to competition from the other species (Figure 1, middle and lower diagrams). As long as inter-species competition is weaker than intra-species competition (γ_i<1), however, the ecosystem still exhibits one almost globally stable equilibrium at point D (Figure 1, middle diagram). In terms of resilience, (almost) every state of the natural system with moderate ecological interaction ($0 \leq \gamma_i < 1$) is therefore characterized by (almost) unlimited resilience.

If inter-species competition is stronger than intra-species competition (γ_i>1, Figure 1, lower diagram), this changes fundamentally as point D no longer represents an almost globally stable equilibrium. D is now only saddlepoint-stable, but B and C are locally stable. Hence, the system exhibits two corresponding basins of attraction: the area northwest of the saddlepath is the basin of attraction for equilibrium B, the area southwest of the saddlepath is the basin of attraction of equilibrium C. Due to an exogenous disturbance, the system may flip from one basin of attraction to another. This means, ecological interaction in the form of strong inter-species competition has a destabilizing effect on the ecosystem.

3.2 Profit-maximizing harvesting under open access to ecosystems significantly weakens resilience

We now include the impact of economic resource use. That is, we no longer study an isolated natural system (as in the last section), but a coupled ecological-economic system with profoundly different resilience properties. In this section, we study this impact for one given level of mild complementarity between ecosystem services in consumption, and without inter-species competition. In the next section, we then systematically study variations in these two parameters – complementarity and inter-species competition.

We suppose for the economic part that profit-maximizing firms can harvest the resource species from their natural stocks under open-access and competitively sell these ecosystem services as market products to consumers. This is the currently dominant economic institution for the use of ecosystem services. Compared to the scenario without resource harvesting and with not-too-strong inter-species competition (cf. Figure 1, upper and middle phase diagrams), the stability properties of the ecosystem are now fundamentally altered (for the mathematical derivation, see Appendix). This dynamics is represented by the state-space diagram shown in Figure 2 for parameter values $\rho_x=\rho_w=0.5$, $\kappa_x=\kappa_w=1$, $\gamma_x=\gamma_w=0$, $v_x= v_w=1$, $\alpha=0.6$, $\sigma=0.4$ and $n=1$.

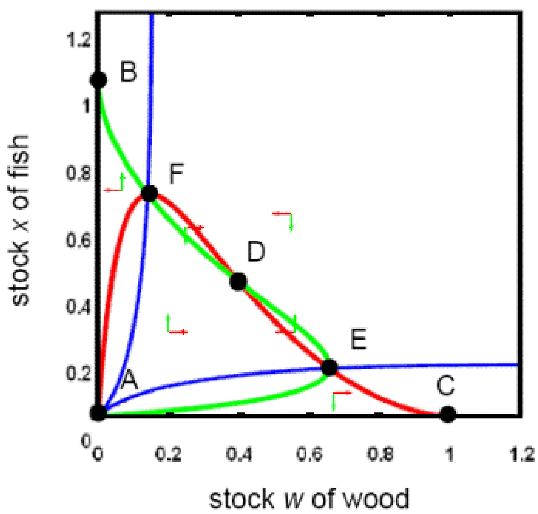

Figure 2: Phase diagram for the ecosystem's dynamics under open access and profit-maximizing harvesting. Dynamics is characterized by $dx/dt>0$ (<0) left (right) of the green line, and $dw/dt>0$ (<0) below (above) the red line. A is an unstable equilibrium; E and F are locally saddlepoint-stable equilibria; B, C and D are locally stable equilibria; the corresponding basins of attraction are the area northeast of the upper saddlepath (for B), the upper saddlepath (for F), the area in between the two saddlepaths (for D), the lower saddlepath (for E), and the area southwest of the lower saddlepath (for C). Parameter values: $\rho_x=\rho_w=0.5$, $\kappa_x=\kappa_w=1$, $\gamma_x=\gamma_w=0$, $v_x= v_w=1$, $\alpha=0.6$, $\sigma=0.4$, $n=1$.

Again, the green line is the isocline for $dx/dt=0$, the red line is the isocline for $dw/dt=0$. Left (right) of the $dx/dt=0$-isocline the dynamics is characterized by $dx/dt>0$ (<0). Likewise, below (above) the $dw/dt=0$-isocline the dynamics is characterized by *dw/dt*>0 (<0). In each segment of state space, the green and red arrows indicate this direction of dynamics. While A ($x=w=0$) is still an unstable equilibrium, B ($x=1$, $w=0$) and C ($x=0$, $w=1$) are now locally stable equilibria. D is still a stable equilibrium, but it is now only locally stable. In addition, there are two new equilibria, E and F, which are locally saddlepoint-stable. The basins of attraction associated with the stable equilibria are as follows: the area northwest of the upper saddlepath (for B), the upper saddlepath (for F), the area in between the two saddlepaths (for D), the lower saddlepath (for E), and the area southeast of the lower saddlepath (for C).

It is obvious that the particular resource management institution considered here – open access to ecosystems of profit-maximizing harvesting firms – has fundamentally altered the resilience properties of the ecosystem. While in the absence of resource harvesting and not too-strong inter-species competition there exists only one (almost) globally stable equilibrium, so that (almost) every state of the system is characterized by (almost) unlimited resilience, under open access to ecosystems of profit-maximizing harvesting firms the system has three locally stable equilibria. Each of those has an associated basin of attraction which comprises only a limited part of the state space, so that the system may flip from one basin of attraction to another one as a result of exogenous disturbance. In particular, equilibrium D (with both resource species in existence) and any state in its basin of attraction have only limited resilience, and any of those states may be disturbed in a way that the system flips into another basin of attraction with another locally stable equilibrium characterized by extinction of one or the other species.

3.3 Complementarity and relative importance of ecosystem services in consumption decrease resilience

Consumer preferences about ecosystem services and manufactured goods are a significant determinant of an ecosystem's resilience properties. This is demonstrated here by illustrating for the institutional setting considered previously – open access to ecosystems of profit-maximizing harvesting firms – how a change in the elasticity of substitution σ between the consumption of fish and timber, and how a change in the relative importance of ecosystem services α, affect the resilience properties of the ecosystem.

In the previous section, the analysis of that setting was carried out for an elasticity of substitution between the consumption of fish and timber of σ=0.4, which reflects a mild complementarity (cf. Figure 2). Figure 3 illustrates the resilience properties of the ecosystem when – everything else being equal – the elasticity of substitution changes to σ=0.95 (low complementarity) and σ=0.05 (high complementarity).

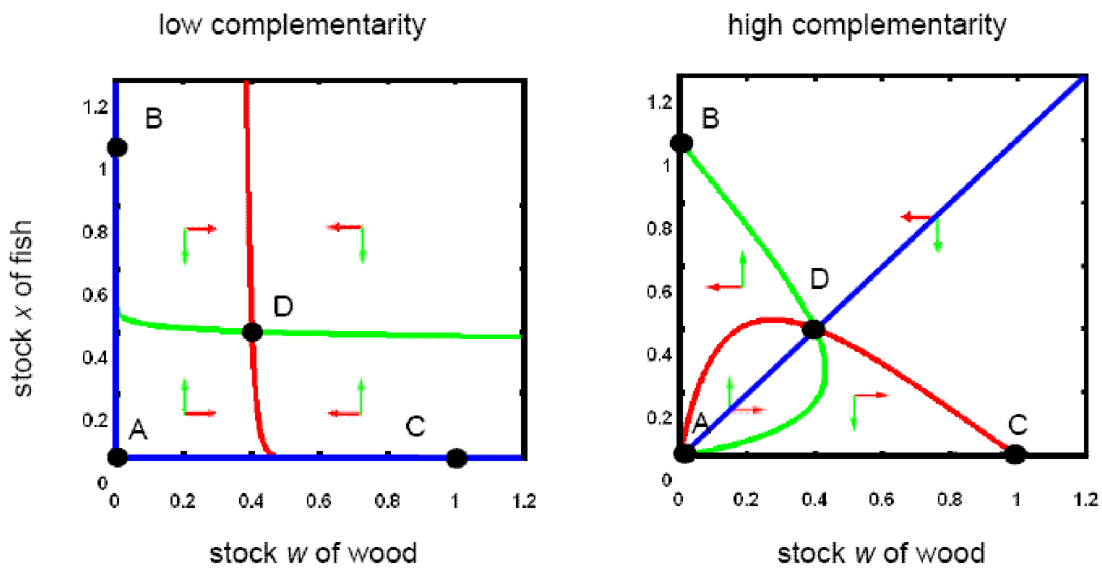

Figure 3: Phase diagrams for the ecosystem's dynamics under open access and profit-maximizing harvesting for low complementarity (σ=0.95, left diagram) and high complementarity (σ=0.05, right diagram) between ecosystem services in consumption. Dynamics is characterized by dx/dt>0 (<0) below (above) the green line, and dw/dt>0 (<0) left (right) of the red line. In the left phase diagram, A is an unstable equilibrium, B and C are locally saddlepoint-stable equilibria, D is the only and (almost) globally stable equilibrium; the corresponding basin of attraction comprises the entire state space with the exception of the axes (x=0, w≥0) and (x≥0, w=0). In the right phase diagram, A is an unstable equilibrium, B and C are locally stable equilibria; the corresponding basins of attraction consisting of the areas northeast (B) and southwest (C) of the saddlepath; D is a saddlepoint-stable equilibrium whose basin of attraction is just a one-dimensional line. Parameter values for both diagrams: $\rho_x=\rho_w=0.5$, $\kappa_x=\kappa_w=1$, $\gamma_x=\gamma_w=0$, $v_x=v_w=1$, $\alpha=0.6$, $n=1$.

From Figure 3 (left diagram) it is apparent that even for open access and profit-maximizing resource harvesting, with low complementarity between ecosystem services in consumption the resilience properties of the system are very similar as in the natural dynamics without human resource management and with moderate inter-species competition. That is, with low complementarity between ecosystem services in consumption, and a low relative importance of ecosystem services, resource harvesting only lowers the species' abundances at the stable equilibrium D (cf. Figure 1), but this equilibrium and every state of the system in its basin of attraction are characterized by (almost) unlimited resilience.

With increasing complementarity between the two ecosystem services in consumption, i.e. a decreasing value of σ, the resilience of this equilibrium reduces. The reason for this decrease in resilience is a vicious circle brought about by the complementarity between ecosystem services. Since the benefits from ecosystem services use are limited by the scarcer service, more effort is spent on harvesting this resource. The increased harvesting effort, in turn, reduces the abundance of that resource even further, thus leading to self-re-enforcing dynamics. At a certain threshold value of σ (σ =1/3 for the parameter values used to compute the figures) the locally stable equilibrium D in Figure 3 (left diagram) loses its stability and turns into an only saddlepoint-stable equilibrium (Figure 3, right diagram). The basin of attraction for this equilibrium is just a one-dimensional line. This means, its resilience is extremely reduced and the state of the system is very brittle and sensitive to exogenous disturbance.

Consumer preferences influence the ecological-economic system's resilience properties also via the relative importance of ecosystem services in the consumer's utility function, α. If ecosystem services are relatively unimportant in the utility function, as compared to the manufactured good, the system shows almost unlimited resilience. In contrast, increasing the relative importance of ecosystem services destabilizes the system. If the relative importance of ecosystem services is very large, the ecosystem's resilience sharply declines and small exogenous perturbations may lead to extinction of one of the species.

Figure 4 illustrates this result. Taking Figure 2 again as a reference point, the phase diagrams of Figure 4 show how changes in the relative importance of ecosystem services in the consumer's utility-function alter the resilience properties of the system. Everything else being equal, decreasing the value of α from 0.4 to 0.25 stabilizes the system in that interior equilibrium D is now almost globally stable (Figure 4, left diagram). Conversely, increasing the relative importance of ecosystem services in the consumer's utility function by raising α from 0.4 to 0.75 entails destabilization of the system: the interior equilibrium's basin of attraction now consists only of the saddlepath, so its resilience is sharply reduced and the system is very sensitive to exogenous disturbance (Figure 4, right diagram).

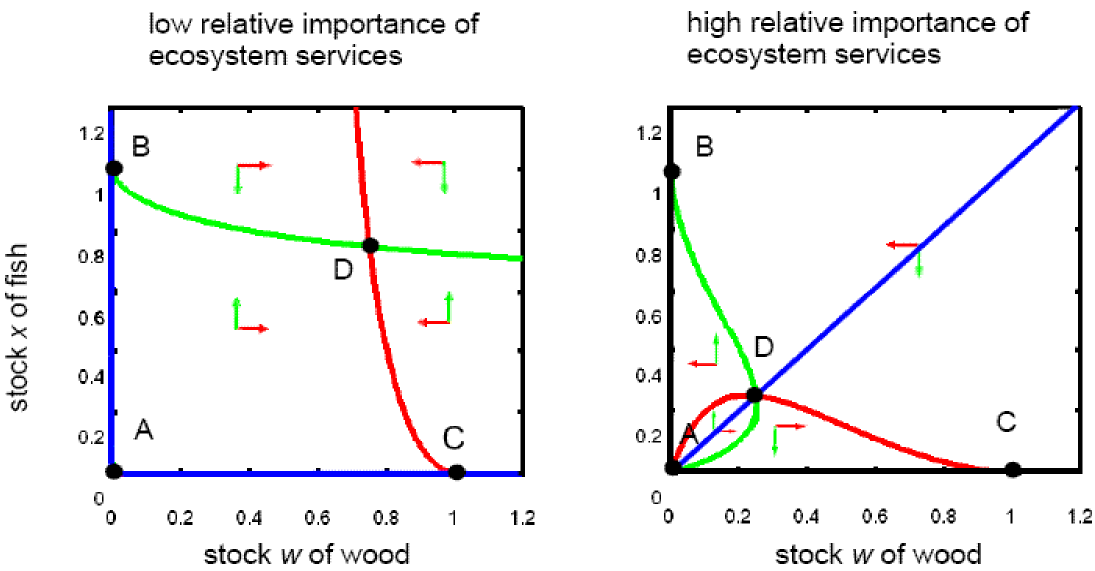

Figure 4: Phase diagrams for the ecosystem's dynamics under open access and profit-maximizing harvesting for different levels of relative importance of ecosystem services, α. Dynamics is characterized by $dx/dt>0$ (<0) left (right) of the green line, and $dw/dt>0$ (<0) below (above) the red line. Blue lines indicate saddlepaths. In both diagrams, A is an unstable equilibrium. In the left diagram, relative importance of ecosystem services is low (α=0.25) and D is an (almost) globally stable equilibrium, while B and C are only saddlepoint-stable. In the right diagram, relative importance of ecosystem services is high (α=0.75) and D is only saddlepoint-stable while B and C are locally stable, the corresponding basins of attraction consisting of the areas northeast (B) and southwest (C) of the saddlepath. Parameter values for both diagrams: $\rho_x=\rho_w=0.5$, $\kappa_x=\kappa_w=1$, $\gamma_x=\gamma_w=0$, $v_x= v_w=1$, $\sigma=0.4$, $n=1$.

In passing we note that increasing the productivity of the harvest technology, v_x and v_w, has qualitatively exactly the same effect as increasing the relative importance of ecosystem services in the consumer's utility function, α: in a market economy and under open access to ecosystems, both changes lead to an increase in harvesting pressure, which reduces the potential for sustainable resource use. Similarly, decreasing the resources' intrinsic growth rates, ρ_x and ρ_w, lowers their ability to recover from harvesting and destabilizes the system in qualitatively the same way.

The general insight from the analysis so far is that resilience of the interior equilibrium with both resource species in existence (point D) tends to decrease (i) with increasing complementarity, i.e. decreasing elasticity of substitution, between the two ecosystem services in consumption and (ii) with increasing relative importance of ecosystem services for the consumer's well-being. In other words, while complementarity and relative importance of ecosystem services in consumption reduce the resilience of the interior equilibrium with both resource species in existence, substitutability and relative unimportance of ecosystem services in consumption tend to make this equilibrium and all system states in its basin of attraction more resilient. This general insight continues to hold with inter-species competition. This is shown in the remainder of the section.

Whereas in Figures 2 to 4 there was no inter-species competition, in the analogously constructed phase diagrams of Figure 5 there is weak inter-species competition (γ_i=0.25). Figure 5 shows that the destabilizing effect of complementarity in consumption also occurs under inter-species competition. The same holds for the destabilizing effect of relative importance of ecosystem services (not shown).

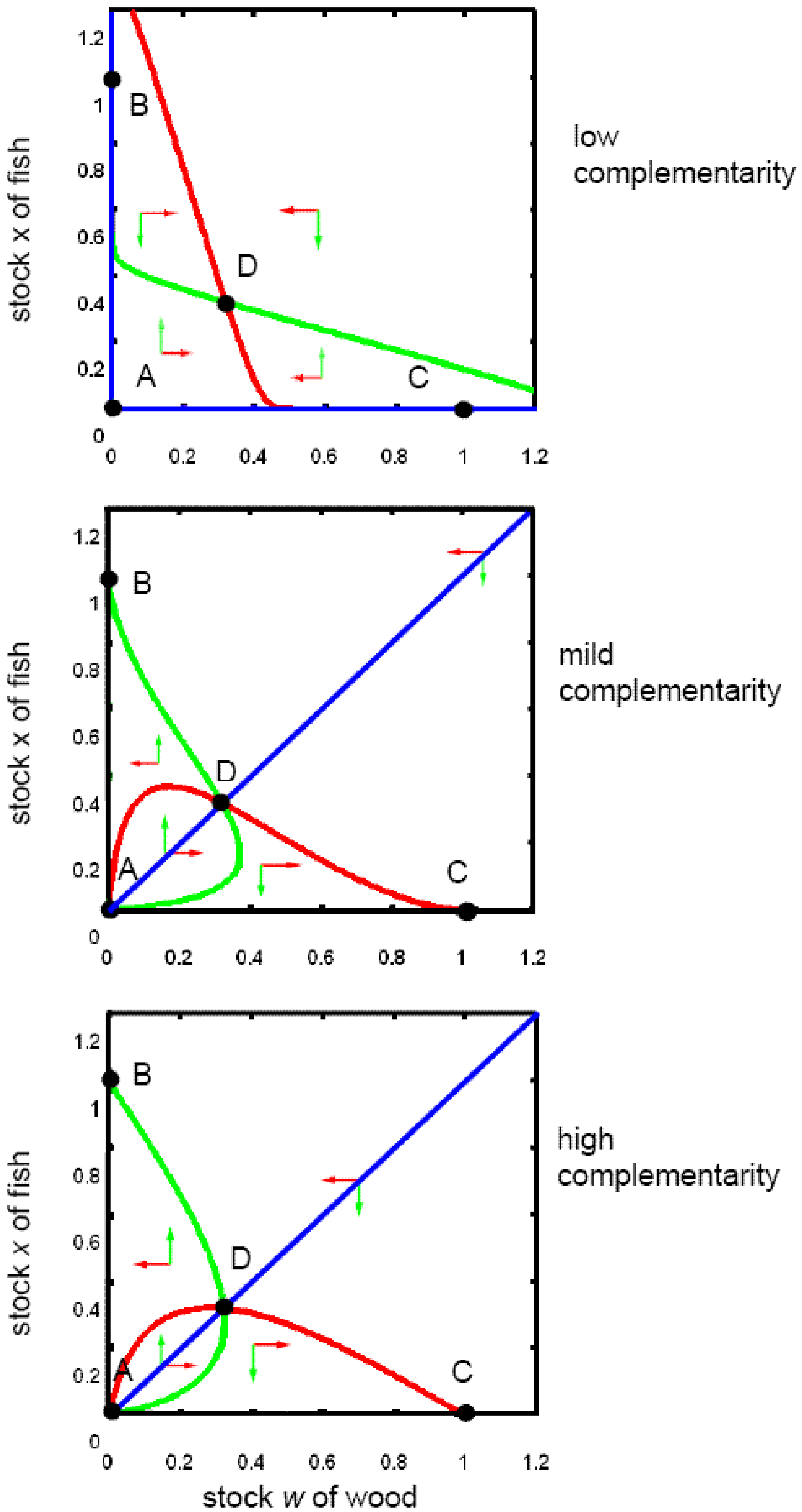

Figure 5: Phase diagrams for the ecosystem's dynamics with inter-species competition for different levels of complementarity between ecosystem services in consumption, σ. Dynamics in each diagram is characterized by $dx/dt>0$ (<0) left (right) of the green line, and $dw/dt>0$ (<0) below (above) the red line. Blue lines indicate saddlepaths. The upper diagram shows the case of low complementarity ($\sigma=0.95$), the middle diagram displays mild complementarity ($\sigma=0.4$) and the lower diagram high complementarity ($\sigma=0.05$).

Parameter values for all diagrams: ρ_x=ρ_w=0.5, κ_x=κ_w=1, γ_x=γ_w=0.25, v_x= v_w=1, α=0.6, n=1.

In all three phase diagrams of Figure 5, equilibrium A, where both species are extinct, is unstable. In the case of low complementarity (σ=0.95, upper diagram, Figure 5), D is an almost globally stable equilibrium, whereas B and C are only saddle-point stable. Thus, there is only one basin of attraction and co-existence of both species is likely. At a certain threshold value of σ (about σ=0.62 for the parameter values used to compute the figures) the locally stable equilibrium D loses its stability and turns into a saddlepoint-stable equilibrium: D lies on a saddle-path and B and C are locally stable equilibria. In other words, if complementarity is high enough, there are two basins of attraction and the interior equilibrium D exhibits very limited resilience (σ=0.4, middle and σ=0.05, lower diagram, Figure 5). Note that compared to Figures 2–4, the threshold value of σ in Figure 5 is higher (i.e. threshold-complementarity is lower) due to the additional destabilizing effect of species competition.

The destabilizing effect of increasing inter-species competition also occurs under resource harvesting. This is shown in Figure 6 for a given level of resource complementarity. Without inter-species competition (γ_x=γ_w=0, upper diagram, Figure 6), the interior equilibrium D with both resource species in existence is locally stable, but exhibits limited resilience due to open-access resource harvesting. The resilience of this interior equilibrium sharply decreases with the introduction of species competition (γ_x=γ_w=0.25, middle diagram, Figure 6): equilibrium D's basin of attraction shrinks to a one-dimensional-line. Thus the system is very brittle and sensitive to exogenous disturbances. Once dislodged from point D, the system will converge to either point B or C, where only one of the species exists. Both B and C remain locally stable equilibria. Further increasing the strength of inter-species competition (γ_x=γ_w=1.25, lower diagram, Figure 6) entails lower abundances of both species at the saddlepoint-equilibrium D.

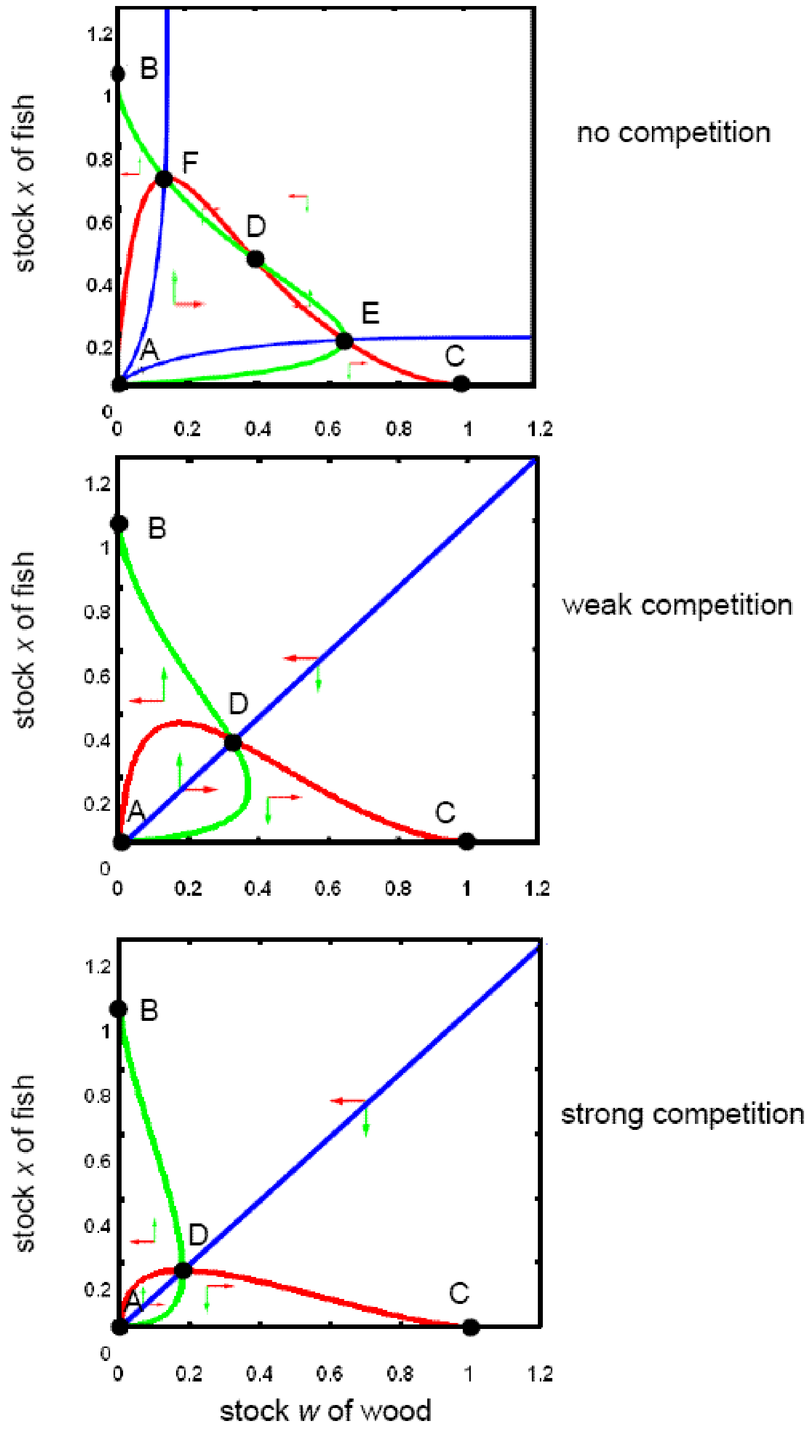

Figure 6: Phase diagrams for the ecosystem's dynamics at a given level of resource complementarity and increasing inter-species competition, γ_i. Dynamics in each diagram is characterized by $dx/dt>0$ (<0) left (right) of the green line, and $dw/dt>0$ (<0) below (above) the red line. Blue lines indicate saddlepaths. The upper diagram displays the case of independent species ($\gamma_x=\gamma_w=0$). Competition occurs in the middle ($\gamma_x=\gamma_w=0.25$) and increases in the lower ($\gamma_x=\gamma_w=1.25$) diagram. Parameter values for all diagrams: $\rho_x=\rho_w=0.5$, $\kappa_x=\kappa_w=1$, $v_x= v_w=1$, $\alpha=0.6$, $\sigma=0.4$, $n=1$.

Comparing Figure 6 to Figure 1 shows that the effects on resilience of increasing inter-species competition are also present under economic resource use. In Figure 6 however, as equilibrium D’s resilience is already decreased by resource harvesting and consumer preferences, low levels of species competition are sufficient to significantly further decrease the resilience of state of the system. Put another way, open-access economic resource use, relative importance of ecosystem services and complementarity in consumption entail a decrease of resilience which may be even larger with stronger species competition.

4 Discussion and conclusion

Our analysis has demonstrated that consumer preferences are an important determinant of the dynamic characteristics of coupled ecological-economic systems, such as limited resilience. In particular, we have clearly distinguished the effects of economic use and consumer preferences from the effect of ecological interactions on the system’s resilience properties.

We have identified three destabilizing effects that genuinely stem from consumer preferences in an ecological system used for economic purposes: First, we have shown that profit-maximizing harvesting by competitive firms under open access to the ecosystem considerably weakens the resilience of the interior equilibrium of the coupled ecological-economic system as compared to the natural dynamics. Second, we have shown that complementarity of ecosystem services in consumption significantly reduces the resilience of the system’s interior equilibrium where both species are in existence. The economic logic behind this result is the following: out of two complementary ecosystem services, the scarcer one is limiting the benefits from ecosystem service use. Hence, under an institutional setting of open access, this ecosystem service is the one to which harvesting is directed primarily. The increased harvesting effort, in turn, reduces the abundance of that resource even further, thus leading to self-re-enforcing dynamics. Third, we have shown that an increased relative importance of ecosystem services for the consumer’s well-being destabilizes the system. The economic logic behind this result is the following: if consumers’ well-being derives to a larger degree from ecosystem services, the share of their budget spent on ecosystem services increases. In a market economy and under open access to resource, this leads to an increase in harvesting pressure, which reduces the potential for sustainable resource use. Conversely, if the consumer's well-being does not, or only to a small degree, derive from consuming ecosystem services, harvesting pressure on the ecosystem is very low and it displays an almost globally resilient interior equilibrium. These three preference-effects act in addition to the ecological mechanisms that are well-known to destabilize an ecological-economic system and to give rise to multiple basins of attraction and limited resilience: increased competition between species and low intrinsic growth rates (e.g. Scheffer 2009).

While our model analysis was based on specific functional forms and certain properties of the particular functions used, of course, determine the results obtained, our results would qualitatively survive a fair amount of generalization. As for the utility function (1), the crucial property, upon which our results critically depend, is the complementarity between the two ecosystem services and the substitutability of aggregate ecosystem services by manufactured goods. As for the logistic growth functions (4) and (5) for both biological resources, the crucial property, upon which our results critically depend, is that the intrinsic growth rate is bounded as the stock declines to zero. Other models with this property, such as e.g. the Beverton-Holt (1957) or the Ricker (1954) models used to describe the dynamics of fish stocks, would yield qualitatively the same results. In contrast, if the intrinsic growth rate increased to infinity as the stock level declines to zero one would obtain qualitatively very different results. Assuming the existence of a minimum viable population level for one or both biological resources would make the whole system even more instable, as we have demonstrated elsewhere (Derissen et al. 2011), and would therefore reinforce our results. As for the Gordon-Schaefer-harvest functions (6) and (7), the crucial property, upon which our results critically depend, is that harvest positively depends on the stock level. Any other harvest function with this property would yield qualitatively the same results. As for the institutional setting, strong complementarity between ecosystem services reduces the resilience of the ecological-economic system also when resources are optimally managed, provided the discount rate applied is relatively large (Quaas et al. 2011).

In the joint endeavor of natural and social scientists as well as practitioners of resource management to understand and manage coupled ecological-economic systems for sustainability, our results call for truly interdisciplinary and integrated analysis of such systems and their management.

Acknowledgments

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References

Arrow, K. J., B. Bolin, R. Costanza, P. Dasgupta, C. Folke, C. S. Holling, B.-O. Jansson, S. Levin, K.-G. Mäler, C. Perrings, and D. Pimentel. 1995. Economic growth, carrying capacity, and the environment. *Science* 268(5210):520-521.

Berkes, F., and C. Folke, editors. 1998. *Linking Social and Ecological Systems. Management Practices and Social Mechanisms for Building Resilience*. Cambridge University Press, Cambridge, UK.

Berkes, F., J. Colding ,and C. Folke, editors. 2002. *Navigating Social-Ecological Systems: Building Resilience for Complexity and Change.* Cambridge University Press, Cambridge, UK.

Beverton, R.J. H., and S. J. Holt. 1957. *On the Dynamics of Exploited Fish Populations*. Fishery Investigations Series II, Xix. Ministry of Agriculture, Fisheries and Food.

Carpenter, S. R., B. Walker, J. M. Anderies, and N. Abel. 2001. From metaphor to measurement: resilience of what to what? *Ecosystems* 4:765-781.

Derissen, S., M. F. Quaas, and S. Baumgärtner. 2011. The relationship between resilience and sustainable development of ecological-economic systems. *Ecological Economics* 70(6):1121-1128.

Gunderson, L. H., and C. S. Holling, editors. 2001. *Panarchy. Understanding Transformations in Human and Natural Systems.* Island Press, Washington DC, USA.

Gunderson, L. H., and L. Pritchard Jr., editors. 2002. *Resilience and the Behavior of Large-Scale Systems.* Island Press, Washington DC, USA.

Holling, C. S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4:1-23.

Horan, R.D., E.P. Fenichel, K.L.S. Drury, and D.M. Lodge. 2011. Managing ecological thresholds in coupled environmental-human systems. *Proceedings of the National Academy of Sciences* 108(18):7333-7338.

Levin, S. A., S. Barrett, S. Aniyar, W. Baumol, C. Bliss, B. Bolin, P. Dasgupta, P. Ehrlich, C. Folke, I.-M. Gren, C. S. Holling, A.-M. Jansson, B.-O. Jansson, D. Martin, K.-G. Mäler, C. Perrings, and E. Sheshinsky. 1998. Resilience in natural and socioeconomic systems. *Environment and Development Economics* 3(2):222-234.

Lotka, A.J. 1932. The growth of mixed populations: two species competing for a common food supply. *Journal of the Washington Academy of Sciences* 22:461-469

Mäler, K.-G. 2008. Sustainable development and resilience in ecosystems. *Environmental and Resource Economics* 39:17-24.

Perrings, C. 2001. Resilience and sustainability. Pages 319-341 *in* H. Folmer, H. L. Gabel, S. Gerking, and A. Rose, editors. *Frontiers of Environmental Economics.* Edward Elgar, Cheltenham, UK.

Perrings, C. 2006. Resilience and sustainable development. *Environment and Development Economics* 11(4):417-427.

Quaas, M. F., D. Van Soest, and S. Baumgärtner. 2011. Resilience of natural-resource-dependent economies. 13th Annual BIOECON Conference, Geneva. http://www.bioecon-network.org/pages/13th_2011/Quaas.pdf

Ricker, W.E. 1954. Stock and recruitment. *Journal of the Fisheries Research Board of Cananda* 11:559-623.

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke, and B. Walker. 2001. Catastrophic shifts in ecosystems. *Nature* 413:591-596.

Scheffer, M. 2009. *Critical Transitions in Nature and Society*, Princeton University Press, Princeton.

Walker, B. C.S. Holling, S. Carpenter, and A. Kinzig. 2004. Resilience, adaptability and transformability in social-ecological systems. *Ecology and Society* 9(2):5 (available at http://www.ecologyandsociety.org/vol9/iss2/art5/).

Volterra, V. 1926. Variations and fluctuations of the number of individuals in animal species living together. Reprinted 1931. In: R.N. Chapman, *Animal Ecology,* McGraw Hill, New York.

Appendix

Taking manufactured goods as the numeraire, the representative household's utility maximization problem is

$$\max_{y,c,h} u(y,c,h) \quad \text{subject to} \quad \omega = y + p_x c + p_w h \;, \tag{A.1}$$

where p_x and p_w are the market prices of fish and timber, respectively. With utility function (1), this leads to Marshallian demand functions for fish and timber:

$$c(p_x, p_w, \omega) = \alpha\, \omega\, \frac{p_x^{-\sigma}}{p_x^{1-\sigma} + p_w^{1-\sigma}} \quad \text{and} \tag{A.2}$$

$$h(p_x, p_w, \omega) = \alpha\, \omega\, \frac{p_w^{-\sigma}}{p_x^{1-\sigma} + p_w^{1-\sigma}} \;. \tag{A.3}$$

Profits of representative firms harvesting fish and timber are given by

$$\pi_x = p_x\, c^{\text{prod}} - \omega\, e_x = (p_x\, \nu_x x - \omega)\; e_x \qquad \text{and} \tag{A.4}$$

$$\pi_w = p_w\, h^{\text{prod}} - \omega\, e_w = (p_w\, \nu_w w - \omega)\; e_w \;, \tag{A.5}$$

where production functions (6) and (7) have been employed in the second equality. In open-access equilibrium, which is characterized by zero profits, i.e. $\pi_x = 0$ and $\pi_w = 0$ for all firms, we thus have the following relationships between equilibrium market prices and resource stocks of fish and wood:

$$p_x = \frac{\omega}{\nu_x} x^{-1} \quad \text{and} \tag{A.6}$$

$$p_w = \frac{\omega}{\nu_w} w^{-1} \;. \tag{A.7}$$

Inserting these expressions into demand functions (A.2) and (A.3), we obtain open-access per-capita resource demands of fish and timber as functions of the respective resource stocks:

$$c(x,w) = \alpha\, \frac{(\nu_x x)^{\sigma}}{(\nu_x x)^{\sigma-1} + (\nu_w w)^{\sigma-1}} \quad \text{and} \tag{A.8}$$

$$h(x,w) = \alpha\, \frac{(\nu_w w)^{\sigma}}{(\nu_x x)^{\sigma-1} + (\nu_w w)^{\sigma-1}} \;. \tag{A.9}$$

General market equilibrium, when aggregate supply equals aggregate demand on the markets for both ecosystem services, is characterized by the conditions

$$C = m_x c^{\text{prod}} = nc(x,w) \quad \text{and} \tag{A.10}$$

$$H = m_w h^{\text{prod}} = nh(x,w) \; . \tag{A.11}$$

Inserting these market-clearing-conditions into equations (2) and (3) yields the following system of coupled differential equations that characterize the dynamics of the ecological-economic system in the general market equilibrium:

$$\frac{dx}{dt} = f(x,w) - nc(x,w) \quad \text{and} \tag{A.12}$$

$$\frac{dw}{dt} = g(w,x) - nh(x,w) \; , \tag{A.13}$$

where $f(x,w)$ and $g(w,x)$ are given by Equations (4) and (5), and $c(x,w)$ and $h(x,w)$ are given by Equations (A.8) and (A.9). The phase diagrams in the main text graphically display the dynamics in state space determined by the system of Equations (A.12) and (A.13).

Chapter 5: How real options and ecological resilience thinking can assist in environmental risk management

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Abstract: In this paper, we describe how real option techniques and resilience thinking can be integrated to better understand and inform decision making around environmental risks within complex systems. Resilience thinking offers a promising framework for framing environmental risks posed through the non-linear responses of complex systems to natural and human-induced disturbance pressures. Real options techniques offer the potential to directly model such systems including consideration of the prospect that the passage of time opens new options while closing others. The implications (cost) of risk can be described by option prices that describe the net present values generated by alternative regimes in the resilience construct, and the shadow prices of particular attributes of resilience such as the speed of return from a shock and the distance or time to transition. Examples are provided which illustrate the potential for integrated resilience and real options approaches to contribute to understanding and managing environmental risk.

Keywords: Resilience thinking, real options, risk, uncertainty, thresholds, transitions

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1 Introduction

Environmental risks are no less a feature of modern society than of times past, and the exposure of society is growing due to population growth and climate change in particular. The pertinence of questions about risk is illustrated by the debate on climate change action: what are the risks? And, what are the consequences of these risks? Alternatively, how much and when should we invest in avoiding or abating the consequences? Our focus in this paper is to describe the potential to integrate resilience theory and real options approaches to inform the cost and management options of complex environmental risks in ecological systems (such as via pre-emptive and responsive investments in protection, restoration, or adaptation).

Resilience thinking offers a way of framing the responses of complex social and ecological systems to human and other interventions (Walker and Salt 2006). The concept of resilience as set out by Holling (1973) comprises two key elements for the purposes of risk research. First, resilience relates to the capacity of the system to absorb shocks or continuous disturbances while maintaining stability (remaining within a single basin of attraction) – generally represented as the reliable delivery of a set of ecosystem or other services. The second element relates to the consequences of disturbance which results in transition of the system to a different state (or basin of attraction). These transitions may be irreversible and non-desirable in the sense that they may lead to a less preferred state (a state which delivers a less valued mix of services). Resilience is directly linked to the concept of risk via an interpretation of transition potential as a probability; albeit often non-measurable (uncertain).

Real options are the application of the methods of finance to a wider range of real world problems to understand the implications of decisions (i.e., to make, abandon, or adjust investments, policies or other interventions) and the passage of time in creating new options and foreclosing others. These features of real options approaches can accommodate resilience thinking by considering the uncertainty of transition, time, and the concepts of irreversibility and path dependent transitions between states. Real option approaches offer a tool for 'valuing' the consequences of transitions from one state to another as an option price relating to the difference in net present values between two different states of a system. Real options approaches also estimate shadow prices of particular attributes of resilience that can be characterised as environmental risks including the speed of return following disturbance, the distance to a threshold, and the expected time to system transition.

For policy purposes there are three useful interpretations of the real options approach to understanding and dealing with risk and resilience. First, the approach can inform questions about whether and when investment in avoiding or mitigating risk is worthwhile by reference to the option price. Second, real options shadow prices can help inform what types of risk reducing investment are most likely to be useful. Third, the approach can be used to inform how we respond to system resilience and the consequent risk to ecosystem service provision in terms of switching decisions about management or investment. A simple ecological example relating to system transition caused by eutrophication of a freshwater lake illustrates these three interpretations. First, option prices can help inform investments in avoiding the risk of eutrophication, such as whether it is worthwhile investing and when to invest. Second, shadow prices can help identify the marginal value of investment in protecting or enhancing particular aspects of resilience and the consequent risk. Finally, the difference in values derived from each system and shadow prices can help inform adaptation decisions to eutrophication where it is not worthwhile or possible to manage environmental risk.

The paper is set out as follows. In the next section we use the concept of resilience thinking to frame and conceptualise how systems respond to disturbances and how to operationalise the concept to aid in evaluating risk. In section three we focus on the information for managing complex systems using a real options approach and we provide a simplified illustration of the approach and the outputs that would result. In section four we use examples of real options applications to describe how these approaches can be used to inform resilience thinking. We conclude with a brief summary of the potential for integrated resilience thinking and real options models to inform environmental risk management, noting the difficulties in practical applications.

2 Resilience as a construct for evaluating risk

2.1 The concept of resilience in social and ecological systems

The "Resilience Alliance" defines 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). It is an ecological construct originally used to frame non-linear ecological responses to disturbance. This concept of resilience is rooted in Holling's (1973) seminal article on stability and non-linear changes in ecosystems. Resilience is measured by the duration and magnitude of disturbance a system can absorb before it switches into another, qualitatively different stable state with a different functional structure and which provides a different, potentially less valued, set of goods and services. A resilient system can absorb exogenous shocks without changing its basic processes. However, a loss of resilience makes the system prone to disturbances and small changes in exogenous conditions may trigger a fundamental change in the system's functional structure (the notion of a non-linear response to disturbance).

The best-known example of alternating stable states in ecosystems is probably found in eutrophication of shallow lakes. Initially, a shallow lake's pristine stable state is characterized by clear water and rich submerged vegetation. Human farming activities often increase the nutrient concentration in shallow lakes. This nutrient loading has no perceivable impact on the water clarity until a critical threshold is reached and the lake undergoes an abrupt transition to the turbid water stable state. In this system configuration, submerged vegetation is almost completely absent and different feedback cycles keep the lake in the new stable state (see for example Carpenter et al. 1999). Other examples include switches between a coral-dominated and an algae-dominated stable state in coral reefs, and switches between a grassy stable state and one dominated by shrubs in semi-arid savannahs (see also Scheffer et al. 2001 for an overview).[1]

2.2 Implications of resilience thinking for system analysis

"Resilience thinking" provides a framework for understanding and evaluating the risk of a system change, and in particular the potential for a non-linear response to disturbances and transitions between potential system states. The transitions between stable states may be evaluated along the two dimensions of desirability and reversibility. First, stable states may be favourable or unfavourable. Hence, transitions are either beneficial or detrimental and system management will either aim at inducing or preventing transitions. An example for a stable state that is highly resilient yet not beneficial from a human point of view is the Sahara desert. Until around 6000 years ago, the North African climate was much wetter than today and large areas were covered with vegetation including wetlands and lakes. Since the abrupt transition to the desert state (a "catastrophic shift" in the terminology of Scheffer et al. 2001), a strong albedo-related vegetation feedback keeps the system in this stable state (see for example Knorr and Schnitzler 2006).

Second, not all transitions are reversible in any practical sense. The forward and backward transitions may occur at different values of the system variables; termed "hysteresis" (Dixit

1989, 1992 albeit an equivalent economic interpretation). Whereas the forward transition might have been triggered by an incremental variable change, the backward transition will *not* be induced by an equivalent incremental change and indeed may require a much larger variable change. Thus, the system exhibits path dependence. An irreversible transition implies a system so hysteretic that a backward shift cannot be effectuated. Clearly, the costs of reversing a transition increase with the system's hysteresis. That is, although a backward shift might be theoretically possible, it could be too costly to achieve.

Although common in a variety of ecosystems, non-linear transitions between multiple stable states may also arise from human management of the ecosystem (see Quaas et al. 2008 for example). In a simple system without complex ecological interactions, limited resilience may be induced into a dynamic state by particular human preferences and resource management institutions. Thus, resilience thinking is ideal to frame the dynamics of coupled social-ecological systems – and these dynamics do not only concern the interactions *within* a single system but also *across* different systems. Following Gunderson and Holling (2002), human and natural systems form clumped, interdependent structures on different scales in time and space. This in turn suggests that the concept of resilience is also relevant for purely man-made systems. Understood as a general concept indicating the risk of a transition occurring, the "resilience perspective" (Walker and Salt 2006) might be applied to any economic or social system that exhibits different stable states. Evidence for non-linear transitions in societies exists throughout human history (see for example Tainter 1988; Diamond 2005; Scheffer 2010) and in economic systems (Dixit 1989, 1992).

2.3 Can resilience be considered as risk?

At this point it is worth defining how we term risk and related terms in this paper. Common parlance tends to refer to risk as the likelihood (not specified as a probability) of an event with adverse consequences and for which there are opportunities to manage the risk in some way. Economists typically apply a more formal definition dividing risk and uncertainty along the lines proposed by Knight (1921) whereby risk is described as having known outcomes with known probabilities and uncertainty implies unknown probabilities and also unknown outcomes. However, Knight's definition is difficult to apply. If there is true uncertainty, decisions are impossible. Instead, people tend to think of systems in transition as uncertain and systems near equilibrium as risky. Kolmogorov (1931) showed how to derive probabilities for such stochastic dynamic systems. Systems in transition have transition probabilities and systems near equilibrium have the more familiar probabilities of Knight's definition. In this paper we regard risk as also incorporating uncertainty for systems in transition as well as near an equilibrium. Where uncertainty is mentioned we are specifically precluding the possibility that either all outcomes or their probabilities are knowable.[2]

Risk can also be scale related in which case systemic risk relates to an entire system rather than a component of that system (and similarly must be managed at the whole of system scale). The preceding discussion implies that the concept of resilience relates to systemic risk explicitly relating to a switch of the system from one state to another (Scheffer et al. 2002). If a system is currently located in a favourable stable state, the risk of a non-linear transition to a less favourable stable state decreases with its resilience. Accordingly, resilience is inversely related to the degree of threat a system is prone to (Brand 2009). That is, resilience may be interpreted as an indicator of the degree of systemic risk: the higher the resilience of a given system configuration, the lower the risk of system transition.

Resilience thinking can be applied to systemic risk either via the probability of a transition from one state to another or to the time that elapses until the system switches to another stable state. First, consider the probability interpretation. Perrings (1998) defines transition probabilities between different states and relates this to the system's resilience. The transition probabilities are conceptualized as a direct measure of resilience: "By this interpretation, the greater the probability that the system in one state will change to some other state, the less resilient is the system in the first state" (Perrings 1998: 8). Hence, resilience is defined as the dependent variable, relying on information about transition probabilities. Baumgärtner and Strunz (2009) and Walker et al. (2010) take another approach and conceptualize resilience as a measurable state variable which determines the probability of a system transition.[3] In this view, resilience is the independent variable and the transition probability results as a function of the current level of resilience. This rests on the assumption that resilience, albeit not a directly observable system property, can be measured by means of surrogates (Bennett et al. 2005).

Second, consider the time interpretation. Pimm (1991) defined resilience as the time the system needs to reach its equilibrium after a disturbance. Since this concept of resilience concentrates on predictability and stability, it has been termed "engineering resilience" (Holling 1996). By focussing on system dynamics close to a single equilibrium steady state, this approach is not adequate to analyse transitions between multiple stable states. Hertzler and Harris (2010) connect resilience to the expected time that elapses until a system subject to disturbance undergoes a transition to another state. They use a real options approach in which probabilities result from stochastic dynamic systems. Unlike the Knightian distinction between measurable risk and uncertainty, time separates risk in the near term and uncertainty in the distant future. As time passes and information is collected uncertainty can usually be resolved. However, many real options studies do not even mention probabilities for three reasons. First, they may not exist. Second, even if they exist, transition probabilities have to be solved for numerically. And third, instead of using probabilities, stochastic dynamic systems are usually modelled directly based on the underlying processes that drive variability (see McDonald and Siegel 1985; Dixit and Pindyck 1994 as examples).

3 Real Options techniques for evaluating the benefits from resilience

3.1 Resilience and the real options approach

Since the social and economic systems to which resilience thinking is applied often provide valued services, the question arises as to the value of resilience. In particular, there may be alternative investment options that would enhance (or reduce) resilience, or in managing the consequences of system transition for the services generated. That is, from the perspective of an environmental manager the usefulness of resilience thinking lies in informing investment decisions about avoiding or preparing for (or promoting) environmental risks posed by system thresholds. It is in thinking about these investment choices, or options, that the real options approach is useful.

Real options are often defined as the application of techniques developed in finance to non financial problems. In the real world, as in finance, every decision (i.e., to make, abandon, or adjust investments, policies or other interventions) and the passage of time creates new options and forecloses others. Real options techniques can be used to model the time dimension of resilience as well as risk in the absence of directly measurable probabilities in a tractable form to inform policy and investment decisions. The ‘option’ which is the focus of the technique is the value of an abstract concept: the flexibility to keep options open while uncertainty is resolved (Dixit 1989). This reveals the notion that options can be sequential and can involve switches between different forms of intervention such as investment, policy, or management. An option represents the opportunity to undertake a specific action but it is not an obligation.

The link to resilience is the value of retaining flexibility by avoiding or reducing the risk of undesirable system transitions. Nevertheless the application of resilience thinking within real options approaches is not necessarily straightforward. Resilience concepts usually refer to systems that remain in a particular state (basin of attraction) in the absence of further disturbances. In a stochastic world, this is called stationarity. Real options approaches do not require stationarity. In fact most real options models assume geometric Brownian motion which is non-stationary. The famous Black-Scholes formula (Black and Scholes 1973) and the investment analysis by Dixit and Pindyck (1994) are two examples. Nor do stochastic systems need to change continuously. Real options can also be analysed in discrete time using discrete probability distributions (see Copeland and Tufano 2004; Luehrman 1998; and Trigeorgis 1996 as examples).

Careful consideration also needs to be given to how risk concepts relating to resilience are best quantified within a real options approach. Resilience is an abstract concept, but could it be valued in the same way that options are valued within the real options approach? The option price describes the net difference in values with and without the management intervention, and can be interpreted as the benefit (cost) from reducing the adverse consequence of risk. Similarly, the resilience related risk constructs (speed of return to equilibrium, distance from a threshold, probability of crossing a threshold, and expected time to cross a threshold) can be described by the relevant shadow price within an appropriately constructed stochastic real option model (though each shadow price is likely to require a different model formulation). The relevant shadow price represents the benefit (or cost) of an additional unit of the particular attribute which can be compared against the cost (or benefit) of changing management to deliver the desired outcome. For example, if we are interested in the risk of a continuous disturbance (such as represented by Brownian motion) causing system transition, we might be particularly interested in whether the speed of return is exceeded by the degree and frequency of disturbance, and if so on the expected time until a threshold is crossed. Alternatively, we may be interested in the price of an option that provides for enhanced flexibility in transitioning a threshold in the face of discrete events (such as represented by a Poisson process).

There are two analytical ways of applying these real options concepts to understanding the consequences and opportunities of environmental risks. The first is a simple diagrammatic approach without any mathematics (Hertzler 2007). Hertzler’s approach involves the use of decision diagrams (similar to decision trees) to set out the available options and consequences and map how these may change with the passing of time or as different options are taken or refused. Anyone can use this approach as a framework for thinking about systems over time and under risk. The second is highly mathematical. It involves modelling nonlinear stochastic dynamic systems, subject to thresholds and irreversibilities. The mathematics is challenging, but the biggest challenge and the potential benefits from the approach, are in understanding how real options approaches model risk and uncertainty. In the remainder of this paper we focus on illustrating the potential of combining resilience thinking with real options to understand and manage environmental risks rather than on the mathematical technicalities.

3.2 Applying the real options approach to resilience thinking

The easiest way to explain the real options approach to managing environmental risk in a resilience context is by way of an illustrative example, in our case using the classic resilience example of eutrophication of shallow lakes. The key conceptual relationships are illustrated in Figure 1 based on three biophysical assumptions (following Scheffer et al. 2002): turbidity increases with nutrient levels; vegetation reduces turbidity; and vegetation disappears at a critical nutrient concentration threshold.

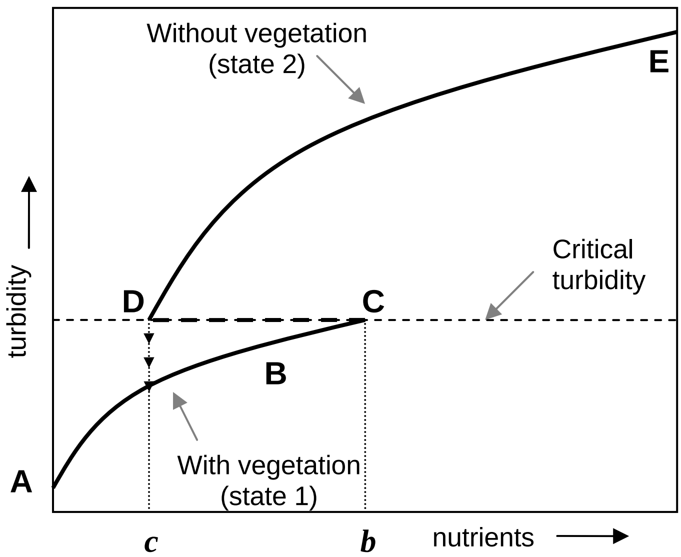

Figure 1: Two stable states in a shallow lake as a function of nutrients and turbidity

Source: Adapted from Scheffer et al. (2002)

The risk of the lake changing state is dependent on the initial state, the vegetation level and the nutrient level. In the vegetated state 1 (A-B-C), increased nutrients lead to increased turbidity until *b* is reached triggering a catastrophic collapse in vegetation and a shift to response curve D-E (the non-vegetated state 2). Return to the vegetated state is only possible if nutrients are reduced below *c*. The nutrient level may be driven by a chain of human decisions and environmental factors, but for simplicity it is assumed that the system is driven entirely by farmer decisions about fertilizer levels (the source of changes to nutrient levels) and stochastic rainfall events (which transport the nutrients to the lake). From two possible starting points in state 1 ('A' with low nutrients and low turbidity, and 'B' with low turbidity and a moderate nutrient concentration) a simple representation of a range of system outcomes can be shown using a decision tree like structure as shown in Figure 2. Attaching probabilities to the rainfall events for given fertiliser levels would estimate the environmental risk associated with eutrophication in this example. Attaching economic benefits to the different outcome states will allow consideration of the value of the 'option' to apply less fertiliser in the system which can then be compared against the costs to agricultural production.

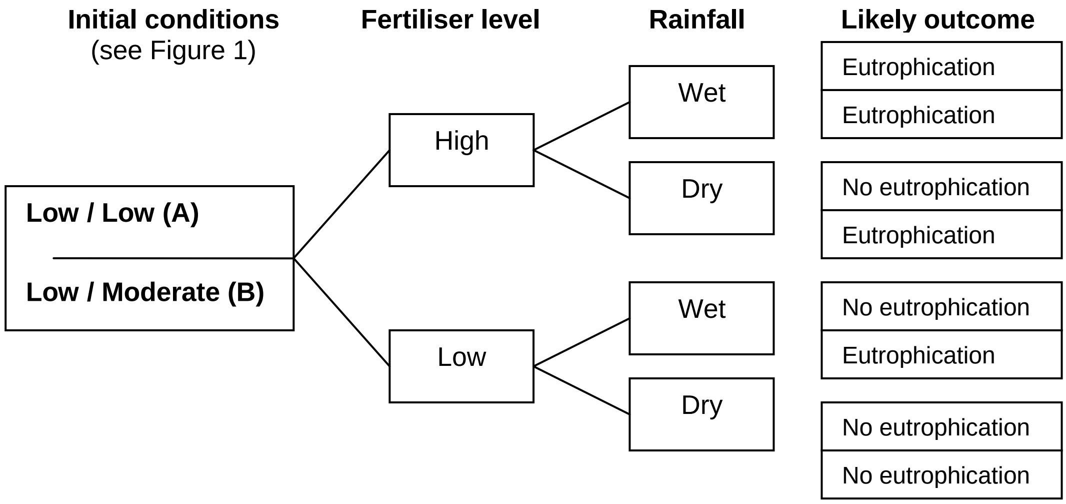

Figure 2: Illustrative decision diagram illustrating the environmental risks associated with different options

Decision diagrams such as shown in Figure 2 provide an excellent way to conceptualise the environmental risks associated with different states of nature. However, such diagrams may not represent the inherent complexity in the real world. In particular, decision diagrams quickly become complex when applied to a large number of options or across a larger number of events and always represent conceptual simplicity at the expense of knowledge about the gaps between defined options, events and outcomes. For example, the initial conditions may change over time depending on previous decisions; rainfall events will not just fall into high and low, but across some distribution from extreme wet to drought and so on.

One way of representing the ecosystem values from the system in different states is shown conceptually in Figure 3. Agricultural benefits increase with increased nutrients, albeit at a diminishing rate and are not state dependent. Community benefits (recreation, fishing, aesthetic, drinking water) sharply diminish with the eutrophic state 2. Total benefits are simply the sum of agricultural and community values. The community and total values derived in each state overlap between *c* and *b* representing the hysteresis effect shown in Figure 1.

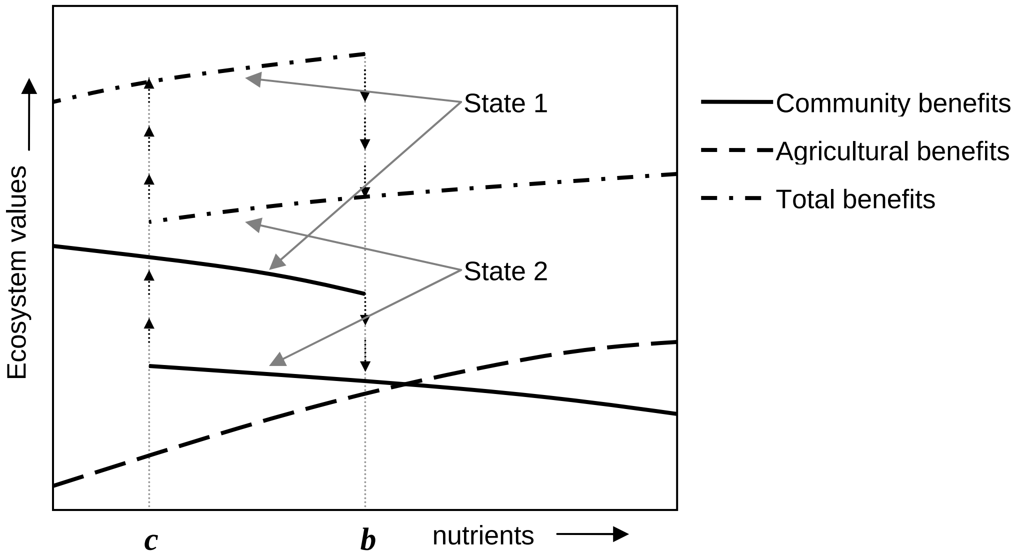

Figure 3: Conceptual model of economic values from shallow lake in two states

Managers and policy makers are most interested in maximising the net benefits that arise and comparing these against the cost of the management options available to maintain these benefits. The net benefits are described for each state by an option price associated with remaining in that regime and represents the value of 'keeping your options open' – the value of flexibility. Unfortunately these values may be difficult to estimate because they are dependent on both the current state of the system and the alternate states of the system. For example, if we are currently in state 1 in Figure 3, we would need to know the option price in state 2 in order to solve for the option price in state 1. In other words, the model must be solved backwards from some terminal value, calculating the option prices for all possible outcomes and using these to calculate the option prices at the present point in the system. Hence, as we will discuss later, option prices are time dependent unless the system is in a stable equilibrium. Option prices have the advantage of being expected values which avoids the need to directly calculate probabilities for each possible outcome. Decision makers can then use these option prices to inform their decisions.

3.3 Describing real options

The real options approach can be used to identify the price of the option of avoiding the state switch in Figure 1; that is the maximum we should be willing to pay to manage the environmental risk of state transition. Conceptually the price of an option also illustrates a fundamental but confusing aspect of real options: how to value an option separately from the value of an entire investment. The value of an option in the resilience construct is generally (but not always) time dependent and since it represents an opportunity (or a right in finance markets terms) rather than an obligation, it will be non-negative.[4]

An illustration of the option values that will result in the shallow lake example we have been following is shown in Figure 4 for avoiding a transition from state 1 to state 2 (non-eutrophic to eutrophic state) based on total benefits from the system. Option values for remaining in state 1 are shown as the heavy solid line in Figure 4. As the state transition is approached the likelihood of transition rises (the environmental risk is higher) and the option price (expected value of an investment ensuring we remain in state 1) becomes larger, eventually reaching the expected value of the future difference in benefits between state 1 and state 2. Once the threshold is breached the option value is bounded by zero (the option only relates to remaining in state 1 and has no value in state 2). Figure 4 also illustrates the value of resolving or reducing uncertainty in a real options framework as time progresses. The lighter lines in Figure 4 represent option values at previous points in time. As we learn more about the system and the environmental risks posed by differing activities, the option value of remaining in state 1 is resolved.

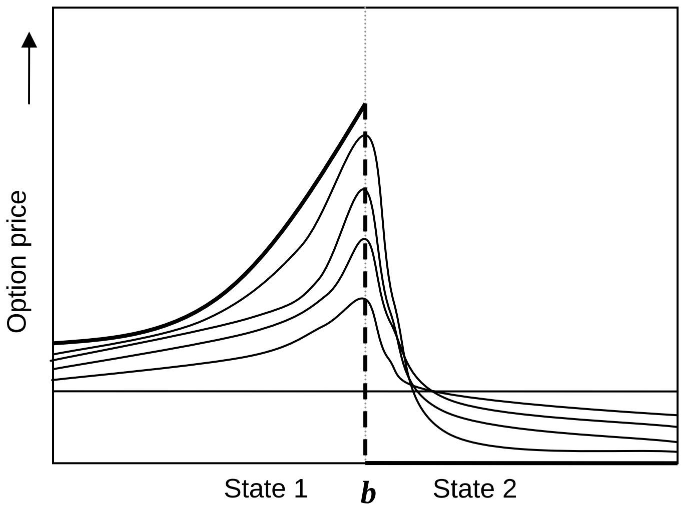

Figure 4: Option price to avoid transition from state 1 to state 2.

A similar exercise can be constructed for the price of an option in state 2 that would return the system to state 1 as shown in Figure 5 (without option prices at previous points in time). As previously, the option price increases close to the threshold due to the expected benefits that would result from transition. At points further from the threshold the agricultural losses would be greater. Note that the transition point changes from *b* to *c* as a result of the hysteresis in the system response to management. The interpretation of the option price differs however in this instance as it no longer represents a maximum willingness to pay to manage environmental risk, and instead represents the maximum price of an investment with benefits defined by the difference between state 1 and state 2.

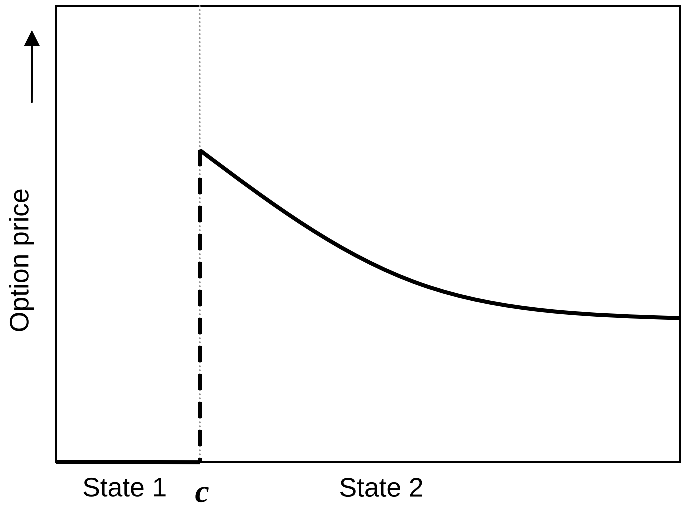

Figure 5: Option price to promote transition from state 2 to state 1.

In the (unlikely) event that both options are available (avoiding transition from 1 to 2, and promoting transition from state 2 to 1) the option value in the zone of hysteresis will be the lesser of the two as illustrated in Figure 6. This would be the equivalent of having the option to invest in make-good insurance rather than further investments in managing environmental risk and avoiding system transition.

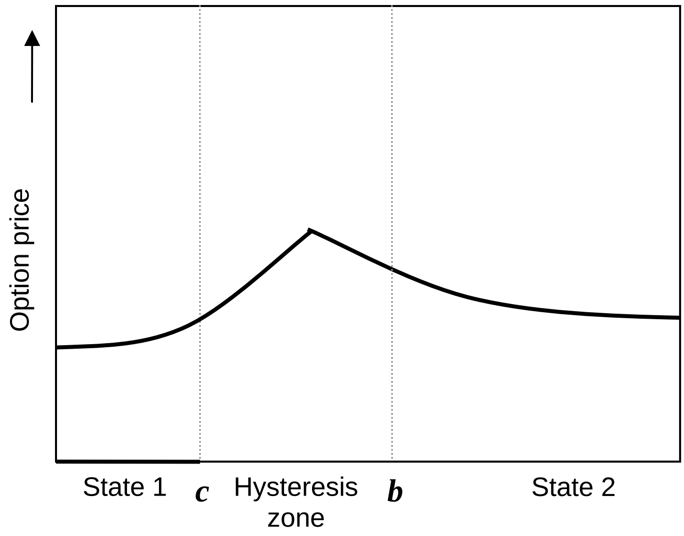

Figure 6: Option prices in the presence of options to avoid transition state 1 to 2 and promote transition state 2 to 1

If the relative benefits from agriculture rise over time, the option value of remaining in state 1 will decline and eventually, the system will be allowed to switch. Conversely if the relative community benefits from the lake rise over time, the option value of remaining in state 1 will increase and the option value of investments to make state 1 more resilient will rise correspondingly.

4 Discussion

The previous discussion has characterised environmental risk management through real options models via an illustration of option prices to avoid or promote transition across regimes. As previously noted, real options models also allow the estimation of shadow prices which represent the marginal value of a specific attribute of resilience, such as the distance from a threshold, the probability of transition, and the expected time to transition. These shadow prices also represent option values in the sense that the cost of an investment (an option) can be compared against the marginal benefit that would result. In this section we describe the ways in which the outputs generated from real options models can influence management of environmental risk and illustrate our points with examples from a sample of the real options literature.

4.1 Option prices as an indicator of the value of resilience

Real options can help us choose appropriate indicators of the dynamics of a system (and of environmental risk) because it is important that we know the resilience of the system to perturbations and the value of this resilience. Just as an option value is the amount we are willing to pay to avoid risk, the option value of system resilience is the maximum amount we are willing to pay to avoid the risk of an adverse transition. If we are willing to pay a large amount it means that the services provided by the system in its current state are valuable to us and we are willing to invest in additional resilience to ensure their continued provision

The real options model illustrated previously was constructed to explicitly provide an option price reflecting the difference in values between two different basins of attraction. The option price is effectively the difference in the net present values of the two alternate regions: the value of resilience in the preferred region. The investment or policy conclusions are as follows. If there is a risk management investment or policy option that would cost less than the option price and which would ensure the system remains in the preferred region, it would be advisable to make that investment. If the only available investments or policies would cost more than the option price, it is preferable (and indeed beneficial from a net present value perspective) to allow transition.

Since real options has rarely been applied in a context that explicitly uses resilience thinking, it is difficult to identify examples which calculate a true option price of a transition between stable states that may be used as a proxy for the value of resilience. Leroux et al. (2009), for example, estimate a price similar to an option price for the value of irreversibly converting land from conservation to agriculture in Costa Rica. In this example, Leroux et al. parameterise their model to generate estimates of a quasi-option value relating to the impact of the ecological concept of an extinction debt (the delayed impact of landuse conversion on biodiversity). If one were to cast Leroux et al.'s study within the framework of resilience thinking the quasi-option value of the extinction debt might be considered as an option price relating to managing the extinction risk resulting from conversion of habitat to agriculture.

Most other real options approaches which could be conceptualised in resilience thinking are focused on aspects of optimal switching points, most commonly in terms of when to commence or discontinue a particular action or policy. Morgan et al. (2007) for example, consider the question of when to discontinue forest harvesting activities with the objective of protecting a population of Caribou under conditions of uncertainty about other impacts on the forest such as wildfire. The decision to stop harvesting is taken at the point just before the probability of extinction exceeds a specified target probability. In a similar way Bakshi and Saphores (2004) consider the (option) value of reintroductions in analysing wolf management decisions.

Possibly the most important optimal switching models in this class (though it is difficult to consider these in terms of shifts between stable states in resilience thinking except in the very long run) consider the implications of uncertainty and climate change policy. Pindyck (2000) considers the implications of uncertainties over future costs and ecosystem responses along with the irreversibilities associated with sunk benefits and costs of environmental regulation. Baranzini et al. (2003) extend the application of real options to include consideration of climate catastrophes. In both cases uncertainty leads to delays in the adoption of climate adaptation policy. The inclusion of increased risk of climate catastrophes by Baranzini et al. (via a Poisson jump process) has the opposite effect by increasing the return from the adoption of adaptation policy. The use of discrete transition probabilities offers significant promise in considering problems of resilience where an extreme event (flood, fire, storm etc.) may trigger a shift in the system from one stable state to another.

4.2 Shadow prices as marginal values for resilience attributes

A key contribution of real options is that it facilitates estimation of marginal values representing the shadow price of resilience (whatever indicator is used for resilience). For example, we could choose speed of return from a disturbance as the indicator of resilience, the distance from a transition, the probability of a transition, or the expected time to cross the threshold. Pimm (1991) for example, proposed the speed at which a system returns to equilibrium as an indicator of resilience. A faster system would be deemed more resilient. It is an ambiguous indicator which leaves out the distance the system may have to travel to push it over the threshold (Holling's 1973 theory) and the frequency and magnitude of the disturbances. Perrings (1998) proposed the probability that a system will cross a threshold as an indicator of resilience. This is an unambiguous indicator but would require solving for transition probabilities at all possible times and states of the system. Hertzler and Harris (2010) proposed the expected time until a system crosses a threshold, given the current state of the system. This is also an unambiguous indicator but requires solving option pricing equations.

Parameters describing speed of return, distance, transition probabilities or time to transition form important aspects of many of the papers described in the previous section, though again we are faced with the consideration that none of these papers explicitly described the problem using resilience thinking. Morgan et al. (2007) incorporate a mean reversion process, while Bakshi and Saphores (2004) incorporate a population growth parameter. In both cases the first derivative of these parameters would provide a marginal value associated with speed of return from a disturbance as an indicator of resilience. Bakshi and Saphores (2004) note that the inclusion of a mean reverting process would provide for limits to predator carrying capacity in the case of wolves, which can also be interpreted as a maximum resilience in a specific ecological state.

Published papers seldom calculate the marginal value of additional information though many of the papers discussed note the implications of uncertainty about ecological or other processes. Leroux et al. (2009) is an exception in that their model can be interpreted as estimating the value that additional information would provide to decisions about optimal conservation subject to environmental risk.

5 Concluding remarks

In this paper, we have described the way in which real option techniques can be applied to resilience concepts to identify the consequences of environmental risk and the management options available to decision makers. Resilience thinking offers a promising framework for considering environmental risk in the context of the non-linear responses to disturbance of complex systems such as ecological and socio-economic systems. In particular, it offers a framework for describing the dynamics of such systems and the existence of thresholds between states or regions of a system, each with different functions, structures, and feedbacks. As transitions from one state or region to another may involve substantive shifts in the values generated from the system, resilience thinking provides a consistent and robust framework for managers and policy makers to better identify and understand the risks and consequences of regime change.

Real options techniques offer the potential to directly model the benefits that result from whole or parts of complex systems. They present managers and policy makers with improved information about the costs (and benefits) of environmental risks and of delaying risk management activities. Investment and disinvestment decisions about whether to manage environmental risks subject to thresholds, hysteresis and irreversibilities can be rigorously analysed.

The formulation and outputs from the real options approach depends critically on the nature of the problem being considered and on the potential intervention opportunities available. Where opportunities to directly influence transition probabilities are available, option prices relating to the difference in net present values generated by alternate regimes may present the most useful output. Targeted information can be obtained from shadow prices, or the marginal values, of particular attributes of resilience such as the speed of return from a shock, the distance to transition or the time to transition. The examples presented in the paper detail the potential for real options analysis to contribute to policy and investment questions in these areas.

Despite the positive assessment presented in this paper, we note the difficulty in applying resilience thinking and real options models to practical problems. These include limited data, analytical complexity, and difficulty in adequately describing the consequences of threshold transition. As an illustration of these difficulties many of the examples identified in the literature incorporate sensitivity and scenario analyses to account for the uncertainty associated with key parameters. In conclusion, there seems to be a rich opportunity to researchers to integrate the fields of resilience thinking and real options models with substantive potential to improve investment and policy decisions under risk and uncertainty.

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Notes:

[1] The "Resilience Alliance" is assembling a database of examples of alternating stable states in ecosystems (Walker and Meyers 2004).

[2] This definition remains rather limited and does not address the notion of subjective risk compared to objective risk and other debates in risk literature.

[3] Note that we have used 'state' in two ways in this paper: the first in resilience thinking to describe the basin of attraction in which the system lies; and the second to refer to the more detailed 'state of a dynamical system' which also contains sufficient information to describe its future behaviour (i.e. its future basin of attraction).

[4] We also note that American options (that may be activated at any point in time prior to state transition) are the type most likely to be of interest rather than European options (that can only be exercised at a terminal time).

References

Bakshi, B., and JD.M. Saphores (2004). *Grandma or the wolf? A real options framework for managing human-wildlife conflicts.* Available online at: http://www.realoptions.org/papers2004/SaphoresHA.pdf.

Baranzini, A., M. Chesney, and J. Morisset (2003). The impact of possible climate catastrophes on global warming policy. *Energy Policy* **31(**8): 691-701.

Baumgärtner, S., and S. Strunz (2009). The economic insurance value of ecosystem resilience. *University of Lüneburg Working Paper Series in Economics*: Working Paper No. 132.

Bennett, E.M., G.S. Cumming, and G.D. Peterson (2005). A systems model approach to determining resilience surrogates for case studies. *Ecosystems* **8**: 945–957.

Black, F., and M. Scholes (1973). The Pricing of Options and Corporate Liabilities. *Journal of Political Economy* **81(**3): 637–654.

Brand, F. (2009). Critical natural capital revisited: ecological resilience and sustainable development. *Ecological Economics* **68**: 605–612.

Carpenter, S. R., D. Ludwig, and W.A. Brock (1999). Management of eutrophication for lakes subject to potentially irreversible change. *Ecological Applications* **9**: 751–771.

Copeland, T., and P. Tufano (2004). A Real-World Way to Manage Real Options. *Harvard Business Review, March*: 1–12.

Diamond, J. (2005). *Collapse: How Societies Choose to Fail or Succeed.* New York: Viking.

Dixit, A.K. (1989). Entry and Exit Decisions under Uncertainty. *The Journal of Political Economy* **97(**3): 620-638.

Dixit, A.K. (1992). Investment and Hysteresis. *The Journal of Economic Perspectives* **6(**1): 107-132.

Dixit, A.K., and R.S. Pindyck (1994). *Investment Under Uncertainty.* Princeton: Princeton University Press.

Gunderson, L., and C.S. Holling, (Eds) (2002). *Panarchy*: *Understanding transformations in human and natural systems*. Washington DC: Island Press.

Hertzler, G. and M. Harris (2010). Resilience as a real option, Paper presented at the Institute for Environmental Decisions Conference: *Environmental Decisions: Risks and Uncertainties. Ascona, Switzerland, April 2010.*

Hertzler, G. (2007). Adapting to Climate Change and Managing Climate Risks by Using Real Options. *Australian Journal of Agricultural Research* **58**: 985-992.

Holling, C.S. (1973). Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* **4**: 1-23.

Holling, C.S. (1996). Engineering resilience vs. ecological resilience. In *Engineering within Ecological Constraints*, ed. P.C. Schulz, 31-43. Washington DC: National Academy Press.

Knight, Frank H. (1921). *Risk, Uncertainty, and Profit*. New York: Hart, Schaffner, and Marx.

Knorr, W. and K.G. Schnitzler (2006). Enhanced albedo feedback in North Africa from possible combined vegetation and soil-formation processes. *Climate Dynamics* **26**: 55-63

Kolmogorov, A.N. (1931). On Analytic Methods in Probability Theory. In *Selected Works of A. N. Kolmogorov; Volume II: Probability Theory and Mathematical Statistics,* ed. A.N. Shiryaev, 62–108. Dordrecht: Kluwer, 1992.

Leroux, A.D., V.L. Martin, and T. Goeschl (2009). Optimal conservation, extinction debt, and the augmented quasi-option value. *Journal of Environmental Economics and Management* **58**: 43.57.

Luehrman, T.A. (1998). Investment Opportunities as Real Options: Getting Started on the Numbers. *Harvard Business Review, July-August*: 3-15.

McDonald, R.L., and D.R. Siegel (1985). Investment and the Valuation of Firms When There is an Option to Shut Down. *International Economic Review* **26**: 331-349.

Morgan, D.G., S. Ben Abdallah, and P. Lasserre (2007). A real options approach to forest-management decision making to protect caribou under the threat of extinction. *Ecology and Society* **13(**1): 27.

Perrings, C. (1998). Resilience in the Dynamics of Economy-Environment Systems. *Environmental and Resource Economics* **11**: 503-520

Pimm, S. L. (1991). *The Balance of Nature*. Chicago: University of Chicago Press.

Pindyck, R. (2000). Irreversibilities and the timing of environmental policy. *Resource and Energy Economics* **22**: 233-259.

Quaas, M.F., S. Baumgärtner, S. Derissen and S. Strunz (2008). Institutions and preferences determine resilience of ecological-economic systems. *University of Lüneburg Working Paper Series in Economics*, Working Paper No. 109.

Scheffer, M. (2010). *Critical transitions in nature and society*. Princeton: Princeton University Press.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker (2001). Catastrophic shifts in ecosystems. *Nature* **413**: 591–596.

Scheffer, M., S., F. Wesley, W.A. Brock and M. Holmgren (2002). Dynamic Interactions of Societies and Ecosystems – Linking Theories from Ecology, Economy and Sociology, in *Panarchy*: *Understanding transformations in human and natural systems*. Gunderson, L., and C.S. Holling, (Eds). Washington DC: Island Press.

Tainter, J.A. (1988). *The Collapse of Complex Societies.* Cambridge: Cambridge University Press.

Trigeorgis, L. (1996). *Real Options: Managerial Flexibility and Strategy in Resource Allocation.* Cambridge Mass: MIT Press.

Walker, B. and J. Meyers (2004). Thresholds in ecological and social-ecological systems: a developing database. *Ecology and Society* **9(**2): 3.

Walker, B., C.S. Holling, S. Carpenter and A. Kinzig (2004). Resilience, adaptability and transformability in social-ecological systems. *Ecology and Society* **9(**2): 5.

Walker, B., and D. Salt (2006). *Resilience Thinking: Sustaining Ecosystems and People in a Changing World*. Washington DC: Island Press.

Walker, B., L. Pearson, M. Harris, K-G Maler, C-Z Li, R. Biggs and T. Baynes (2010). Incorporating Resilience in the Assessment of Inclusive Wealth: An Example from South East Australia. *Environmental and Resource Economics* **45**: 183-202.