Aus dem Institut für Agrarökonomie
der Christian-Albrechts-Universität zu Kiel

Managing ecological-economic systems under uncertainty

–

From concepts to contracts

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Dissertation
zur Erlangung des Doktorgrades
der Agrar- und Ernährungswissenschaftlichen Fakultät
der Christian-Albrechts-Universität zu Kiel

vorgelegt von
Diplom Landschaftsökologin Sandra Derissen
aus Würselen

Kiel, im September 2014

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1. Berichterstatter:	Prof. Dr. Uwe Latacz-Lohmann
2. Berichterstatter:	Prof. Dr. Martin F. Quaas
Tag der mündlichen Prüfung:	12.11.2014

Gedruckt mit Genehmigung
der Agrar- und Ernährungswissenschaftlichen Fakultät
der Christian-Albrechts-Universität zu Kiel

Table of Contents

Table of Figures

English Summary

The dissertation is concerned with the management of ecological-economic systems, especially with policy instruments to promote the ecosystem services such systems provide. Within the research field of environmental and resource economics, resilience has become a key concept to give guidance for the management of ecological-economic systems. However, resilience as an attribute of those systems cannot be alone a sufficient paradigm for a management device e.g. regarding sustainablity, a fallacy which is often made.

For the attempt to maintain or increase a specific ecosystem service, such as water purification, landscape beauty or watershed production, policy instruments have been approved as a means to produce environmental goods and services. Such approval is embedded in the logic of public goods and market environmentalism: If markets do not exist the objective is to construct appraisal methods that come as near to the Free Market ideal as possible. Within design of those instruments the dynamics of the system which produces an ecosystem service are usually not considered.

The aim of the dissertation is therefore to apply the insights regarding the dynamic of ecological-economic systems and make this knowledge applicable and valuable for the design of policy instruments. Since many open questions exist regarding the behavior and the dynamics of ecological-economic systems, the dissertation exhibits a conceptual and analytical character in large parts. In a fist instance, the relationship between resilience and sustainable development of ecological-economic systems is discussed in detail by means of an ecological-economic system featuring multiple stable states. The obtained result is that a deduction from resilience to sustainability, or vice versa, is not possible. On the basis of the same model the effects of factors influencing the stability and resilience of a system, such as complementarity of resources and species interaction are analyzed. Thereby the insights and knowledge about ecological-economic systems are expanded. To make these insights applicable and valuable for the management of ecological-economic systems the design of policy instruments is analyzed.

In this respect the focus is on the so called "Payments for Environmental Services" (PES). Usually a voluntary contract is negotiated between a landowner and some

institution. As for the application and formalization of this contract relationship a principal-agent model is adopted for the purpose of the dissertation. The intention is to analyze and design optimal contracts regarding different risk-preferences of the principal and environmental uncertainty, two factors which might be influenced and determined by the resilience of the system's state.

With the adopted model the much discussed question, about the superiority of performance-based over action-based payments and vice versa is overcome by the proposal of a combined payment scheme. Within a case study, considering the conservation of the Scarce Large Blue (Maculinea teleius), it become obvious that a combination of performance-based and action-based payments within one contract scheme generates positive welfare effects.

Summing up, on the one hand the dissertation reveals new insights about the behavior and the dynamics of ecological-economic system. On the other hand common practices of the design of conservation contracts are revised and developed further.

In a synthesis of the dissertation's results, it is concluded that the design of policy instruments with respect to the dynamics of ecological-economic systems is a very promising and recommendable concept. Under the challenge of uncertainty, risks can be shared more efficiently, which might generate positive welfare effects.

Zusammenfassung

Die vorliegende Dissertation beschäftigt sich mit dem Management ökologisch-ökonomischer Systeme, insbesondere mit Politikinstrumenten, welche die Bereitstellung von Ökosystemdienstleitungen durch solche Systeme unterstützen. Innerhalb der Umwelt- und Ressourcenökonomik gilt Resilienz zunehmend als ein Schlüsselkonzept für das Management von ökologisch-ökonomischen Systemen. Hierbei wird oft jedoch nicht berücksichtigt, dass Resilienz als eine Eigenschaft ökologisch-ökonomischer Systeme noch keine hinrichtende Bedingung für eine Managementscheidung, z.B. in Hinblick auf Nachhaltigkeit, liefert.

Bei dem Bestreben bestimmte Ökosystemdienstleitungen, so genannte "Ecosystem Services", zu fördern und zu erhalten, werden für den Bereich öffentlicher Güter, zu denen viele dieser Dienstleistungen gehören, vor allem marktwirtschaftliche Instrumente eingesetzt. Bei der Anwendung und Gestaltung dieser werden jedoch bisher die Eigenschaften von dynamischen Systemen nicht berücksichtigt.

Das Ziel der Dissertation war es daher, neue Erkenntnisse zur Dynamik ökologisch-ökonomischer Systeme zu gewinnen, und diese bei der Gestaltung von Politikinstrumenten in den Fokus zu nehmen. Da zu dem Verhalten und zu den Eigenschaften ökologisch-ökonomischer Systeme noch viele Fragen offen waren und sind, hat die Arbeit in weiten Teilen einen stark konzeptionellen und analytischen Charakter.

Zunächst wird das Verhältnis des Resilienz-Paradigmas als Eigenschaft ökologisch-ökonomischer Systeme zu der normativen Forderung nach einer nachhaltigen Entwicklung analysiert. Mit Hilfe eines ökologisch-ökonomischen Modells werden die unterschiedlichen logischen Möglichkeiten zwischen Resilienz und nachhaltiger Entwicklung dargestellt. Hierbei wird deutlich, dass der logische Schluss von der Resilienz eines bestimmten Zustandes auf dessen Nachhaltigkeit und umgekehrt nicht möglich ist. Anhand eben dieses Modells werden die Auswirkungen von Faktoren wie der Komplementarität bestimmter Ressourcen und Artinteraktionen auf die Stabilität eines Systems verdeutlicht. Die Erkenntnisse in Bezug auf die Reaktionen ökologisch-ökonomischer Systeme werden damit erweitert. Um diese Erkenntnisse auch für das Management ökologisch-ökonomischer Systeme nutzbar zu machen, werden dazu Politikinstrumente untersucht.

Im Fokus stehen dabei freiwillige Verträge, so genannte "Payments for Environmental Services" (PES). Innerhalb der Dissertation wird anhand eines Prinzipal-Agenten Modells, welches zur Darstellung und Formalisierung der Vertragsbeziehungen dient, analysiert, welche Vertragskombinationen bei verschiedenen Risikoeinstelllungen des Prinzipals und unter Umweltunsicherheit angeboten werden sollten, um ein optimales Ergebnis zu erreichen. Die viel beachtete Diskussion um die Überlegenheit von maßnahmen- bzw. ergebnisorientierten Zahlungen für Ökosystemdienstleitungen wird dabei um den Vorschlag erweitert, innerhalb eines Vertrages ebendiese zu kombinierten. Am Beispiel des Schutzes und der Wiederansiedlung des Ameisenbläulings (Maculinea teleius) wird gezeigt, dass eine Kombination von ergebnis- wie maßnahmenorientierter Honorierungen innerhalb eines Vertrages positive Wohlfahrtseffekte verspricht.

Insgesamt werden damit zum einen neuen Erkenntnisse zu den Eigenschaften und den Reaktionen ökologisch-ökonomischer Systeme gewonnen und zum anderen derzeit übliche Vertragsnaturschutzmodelle weiterentwickelt und verbessert.

In der Synthese dieser Ergebnisse wird gefolgert, dass die Anpassung von Politikinstrumenten an die jeweilige Dynamik eines Systems vielversprechend und empfehlenswert ist. So können unter dem Aspekt der Unsicherheit Risiken besser verteilt und damit Wohlfahrtseffekte erzielt werden.

Chapter 1

Introduction and Paper Summaries

The management of ecological-economic systems is becoming increasingly complex. On the one hand, present social and economic objectives have to be fulfilled. On the other hand, our responsibility with respect to future generations means that we must not deplete natural resources. Additionally, natural systems used and managed by humans for their ecosystem services may exhibit nontrivial dynamics (Baumgärtner et al. 2011). As a consequence, management addressing the long-term conservation and sustainable use of ecosystem services is a significant challenge. To successfully address this challenge, it is important to understand the dynamics of ecological-economic systems.

Within the field of environmental and resource economics, models have been advocated to emphasize the interconnectivities between ecosystems and economic principles. These ecological-economic models are characterized through assumptions on both sides i.e. economic relations as the management regime, and determinants of the ecosystem such as the growth rates of the resources in question. Since these ideas are most common within environmental and resource economics in the recent years, up to this point the theory and research of non-marketed goods and incentives to maintain these goods are mostly unconsidered regarding a system based view. Therefore, characteristics of ecological-economic systems such as the level of resilience or the possibility of non-linear reactions have not yet received any consideration within the design of policy instruments.

The dissertation aims to narrow this research gap by connecting the insights of system dynamics with the design of conservation contracting. More precisely the dissertation is concerned with the management of ecological-economic systems and the provision of ecosystem services under uncertainty. A major focus will be on policy instruments which attempt to obtain or maintain specific ecosystem goods or services provided by such systems. Insights are derived from stylized ecological-economic models and a principal-agent model together with literature analysis. The central focus of the dissertation is thereby neither a methodological discussions nor qualitative or quantitative inquiries and although some of the case studies link to empirical results the main focus is on conceptual insights.

For a foundation the first focus is on the discussion of concepts and terms for the purpose of the dissertation, i.e. the concepts of “resilience”, “sustainability”, and “Payments for Ecosystem Services”. In a second layer, the properties of ecological-economic systems providing ecosystem services are analyzed. In a third step concrete enhancements for instruments of policy design are proposed, while open questions remain for further research on how to design conservation contracts with respect to system properties.

Turning to the field of natural resource management in general, Rist & Moen (2013) distinguish three approaches for providing a guideline for management decisions: The Ecosystem Approach, Adaptive Management and the paradigm of Resilience Thinking[1]. For this thesis the last concept is particularly relevant and will be discussed further.

The focal point of the paradigm of resilience is rooted in Holling's article (1973) on stability and non-linear changes in ecosystems (Folke et al. 2010). Regarding this management perspective Whitten et al. (2012: 331) state: "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". In comparison to other management paradigms the paradigm of resilience assumes the existence of multiple local equilibrium, hysteresis, alternative stable states and regime shifts[2].

Looking at the paradigm of resilience in more detail, two prominent meanings can be distinguished. On the one hand, according to Holling (1973), resilience is understood as "[...] the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior" (Holling & Gunderson 2002: 4). In this sense, resilience is thought of as "the capacity of a system to absorb disturbances and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks." (Walker et al. 2004: 2). On the other hand, following Pimm (1984), resilience is defined as the rate at which a system returns to its equilibrium following a disturbance. This concept has been implemented to calculate the cost of systems' recovery after disturbances (Martin 2004).

The dissertation follows Holling's paradigm of resilience. Ecosystems are thought of as having different "states" or – in alternative terminology – "basins of attraction". A system in a certain state is defined by a set of variables e.g. for an agricultural ecosystem such state variables could be land, crops, livestock, farmers or roads. In one basin of attraction the system has the same essential structure, function, feedbacks and, therefore, identity (Walker et al. 2004)[3]. A regime shift occurs when due to a disturbance a system crosses a threshold into an alternate basin of attraction[4]. Due to this disturbance the system is

[1]For a comprehensive overview and comparison of these different management paradigms, their distinctions and similarities with a special focus on the Resilience Thinking approach, see Rist & Moen (2013).

[2]A system might shift easily from one stable state to another, but much more energy or effort might be necessary to shift the system back to its initial state. This phenomenon is called "hysteresis" (e.g. Scheffer & Carpenter (2003)).

[3]For a comprehensive outline of the paradigm of resilience and Resilience Thinking see the pages of the Resilience Alliance (http://www.resalliance.org) and the Stockholm Resilience Center (SRC) (2013) (http://www.stockholmresilience.org).

[4]For a comprehensive study of the background theory of "ecosystem resilience" see Brand (2005).

then called “not resilient”.

From a human perspective this state might be identified as desirable or undesirable depending on the desired amount of certain ecosystem services which are either provided or not provided. Therefore management will either aim at achieving or preventing a transition into another state of the system. For example: In the semi-arid rangelands of West Africa, climate conditions are highly variable and uncertain and cause scarce and spatio-temporally variable resource availability (Jakoby 2011). The most important type of land-use is livestock grazing and the landscape was initially covered with a mixture of grassland and shrub vegetation. Recently, the ecosystem underwent radical change as the excessive appearance of bushes on the grazing areas, also often called “bush encroachment”, became a serious and apparently unstoppable problem, which caused a lot of concern:

“It can further be concluded that grazing pressure, even with declining stocking rates, was still inherently too high to utilise the rangelands in a sustainable way and resulted in a form of vicious cycle. Fear has been expressed that the bounds of resilience of the former ecosystem have been exceeded. Only by means of external inputs will the original status of our rangelands be able to be restored” (de Klerk 2004: 22).

In the case of the semi-arid rangelands, the grass-dominated state is preferred to a bush dominated state insofar as this state provides desired ecosystem services, since in Namibia agriculture is regarded as the backbone of the economy and the grass dominated state supports livestock grazing (e.g. de Klerk 2004 and Stehn 2008).

By the same argument, the focus of ecosystem management strategies is often on the maintenance of a given system state, without reflecting the characteristics and desirability of possible alternative stable states. However, another perspective would be to ask how a different state of the system could meet the needs of a society, and if for example, the benefits of transforming the system to an alternative state would outweigh the costs of maintaining the given state. To decide which state is desirable and which is not, clear definitions are necessary: What management expects from any given states, and which services an ecosystem should provide. Such expectations might be e.g. supply of food, energy or recreation, which can be subsumed as ecosystem services as e.g. the Millennium Ecosystem Assessment (MA) indicates (2005). The more precisely such expectations are articulated the clearer it becomes which state of the system management should aim to achieve.

Adopting a sustainability perspective constitutes one way to formulate such criteria. The World Commission on Environment and Development (WCED) defines sustainable development as development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED 1987). More precisely in the terminology of this dissertation a state hereafter corresponds to the claims of sustainability if the utility of the ecosystem services this state provides do not fall below a certain baseline over time[5].

Policy instruments have been approved as a means to procure environmental goods and services. Such approval is embedded in the logic of public goods and market environmentalism: If markets do not exist the objective is to construct appraisal methods that come as near to the Free Market ideal as possible (Vatn 2009 and Corbera 2007). Environmental and resource management encompasses many policy instruments such as taxes, subsidies and tradable permits. In general one can distinguish between regulatory and economic instruments for the enhancement and conservation of environmental goods and services. Regulatory instruments influence the actions of an agent directly, e.g. because of sanctions enforcing certain actions. In contrast, the aim of economic instruments is to influence actions and choices of the agent indirectly by varied restrictions or incentives (Matzdorf 2004).

On this basis, the dissertation recognizes incentives as the focal point for the provision of public goods, such as environmental goods, and as a possibility to maintain or achieve a certain desired state of a system. This broadens the view to the whole literature of Contract Theory, Incentive Theory and principal-agent models[6]. In this context the question of the "right" incentives and optimal contracts between principal and agent is discussed by means of analytical models. Up to this point the existing literature regarding the theory of contracts focuses on labor economics and production of manufactured goods (Bolton & Dewatripont 2005). By contrast, applications to the domain of environmental and resource economics are sparse[7]. One main aim of the dissertation therefore is to apply the analytical framework of contract theory to the field of policy design and especially to the design of conservation contracts.

Conservation contracts usually fall into two subcategories of payment schemes: (i) payments which are bound to a specified performance or outcome, hereafter called performance-based payments and (ii) payments which are bound to some defined management, hereafter

[5]This notion corresponds to a concept of "weak sustainability" (Ott & Döring 2004).

[6]As described in detail within Chapter 8 a person (hereafter called an agent) e.g. a land owner, is rewarded for the environmental service he provides. Within a framework of PES the contracts are usually voluntary. Therefore, one challenge is to incentivize the agent's participation.

[7]While many practical examples can be found a consideration of conservation contracts within a framework of contract theory and on a conceptual level is surprisingly rare, see Zabel & Roe (2009) for one example.

called action-based payments[8].

Up to this point, action-based payments are the most established form of conservation contracting. Their advantages are notable if actions are directly bound to the outcome or if a specific management itself is an aim of the contract, e.g. low use of fertilizer, or a specific mowing regime. With information asymmetry, i.e. the agent is better informed about the best input level, performance-based payments allow the agent to find the best way to generate a desired level of an ecosystem service without determining her actions[9]. However, as the dissertation considers, external influences (e.g. disease, floods, and droughts) affect the provision of those services. Therefore performance-based payments are usually applied for the conservation of a state already existing e.g. high levels of biodiversity, to minimize the risk of the assumed risk-averse agent (Osterburg 2006 and Hampicke 2001). Action-based payments are then applied for cases with uncertain outcomes, especially if the principal can be assumed as risk-neutral. A drawback of this scenario, however, is that information asymmetries and uncertainty of the contract outcome might appear together. In these cases one cannot decide for an exclusively performance-based or an exclusively action-base payment scheme for optimality.

The dissertation thesis investigates a combination of these payment schemes for the case of uncertainty and information asymmetries between principal and agent. With the adopted model the much discussed question concerning the superiority of performance-based over action-based payments and vice versa is overcome by the proposal of a combined payment scheme. Within a case study, considering the conservation of the Scarce Large Blue Butterfly (Maculinea teleius), it becomes obvious that a combination of performance-based and action-based payments within one contract scheme generates positive welfare effects.

Within the literature concerning the PES field, different payment schemes regard different social and economic aspects. A contract design regarding the dynamic and specific situation of an ecological-economic system has not yet been conducted although policy instruments, which aim to prevent or achieve a special environmental service by inducing a certain management strategy, are heavily dependent for their success on the knowledge of the ecosystem and e.g. species interactions, regime shifts and tipping points. The thesis aims to narrow this research gap by connecting the insights of system dynamics to the design of conservation contracts and management strategies as a leading research paradigm

[8] Advantages and disadvantages regarding both payment schemes are discussed within the specification of Chapter 6. Further literature readings e.g. Ferraro & Kiss (2002) and Matzdorf (2004) are recommended.

[9] Within principal-agent models and contract theory the agent is usually denoted as “she” whereas the principal is denoted as “he”.

since up to this point specific characteristics of systems receive no consideration. In detail, connecting the perspectives discussed above led to the following specific research questions and propositions for the dissertation:

Within this context the dissertation focuses on one recently much discussed framework for the maintenance and support of public goods: The concept of "Payments for Environmental" or "Ecosystem Services" (PES) (e.g. Wunder 2005 and Pagiola 2008). The terms "Payments for Environmental Services" and "Payments for Ecosystem Services" are frequently used as synonyms but also as discrete concepts. A literature overview revises the general terminology of environmental and ecosystem services is presented in Chapter 2, asking how "Payments for Environmental Services" might be defined in a consistent way.

In Chapter 3 the management of coupled ecological-economic systems which provide ecosystem services such as the semi-arid rangelands in Namibia is addressed on a conceptual level. Insights considering this research study have been leading to the general question whether resilience of an ecological-economic system state is a precondition for sustainability or not; a proposition which often has been made in the literature. Regarding this question, the logical relationship of resilience as a property of ecological-economic systems and sustainability as a normative claim is analyzed on the basis of an ecological-economic model (Chapter 4).

During the work with the ecological-economic model of Chapter 4 it becomes obvious that the stability landscape of the model changes rapidly with changes of assumptions regarding the relationship between species and the options of assessing the resources[10]. In this respect the model is studied further, to understand the driving forces of ecological-economic systems (Chapter 5). Up to this point characteristics of ecological-economic systems such as the level of resilience or the possibility of non-linear reactions receive no consideration within the design of conservation contracts. Aiming to narrow this research gap, the first question is which combination of payment schemes are preferable regarding a socially optimal result. In Chapter 6 a principle-agent model is used to help elucidate the contract relationship. Within this framework of a model, analysis varying environmental uncertainty and different levels of risk-aversion are considered.

To conclude, the dissertation reveals new insights about the behavior and the dynamics of ecological-economic system and common practices of the design of conservation contracts are revised and developed further. In a synthesis of the dissertation's results, it is

[10] The term "stability landscape" is intended here to address the inherent dynamic and stability properties of the whole system. As similarly used by Walker et al. (2004), imagine a 3-dimensional state space with all equilibria and tipping points of the systems, one can imagine a picture of hills and valleys, shaping the "stability landscape" of the system.

concluded that the design of policy instruments with respect to the dynamics of ecological-economic systems is a very promising and recommendable concept. Under the challenge of uncertainty, risks can be shared more efficiently, which might generate positive welfare effects.

The published papers of the dissertation thesis are here summarized:

Chapter 2: What are PES? - A review of definitions and an extension

Chapter 2 compares and contrasts existing definitions of the term PES in the literature (Derissen & Latacz-Lohmann 2013). Since PES is widely used either as "Payments for Ecosystem Services" or "Payments for Environmental Services" as a first step, definitions of ecosystem services and environmental services are discussed. This discussion points to coexistence of different definitions with different meanings. Some authors differentiate the two terms while others use ecosystem and environmental services synonymously. Similarly, some authors use the terms "Payments for Environmental Services" and "Payments for Ecosystem Services" as synonyms while others distinguish them to describe distinct concepts.

These terminological inconsistencies lead to the question which term to use within this dissertation thesis. The thesis is explicitly concerned with the framework and concept of conservation contracting. In this context, environmental services are regarded as public goods. Such public goods are not supplied in sufficient quantities if individuals act in their own self-interest (e.g. Ferraro & Kiss (2002) on biodiversity). This is why incentives are given to initiate their production. The chapter focuses on a debate in Germany where landowners receive rewards for certain actions or omittance of actions. Thus these "Honorierung ökologischer Leistungen" constitute a payment for the effort of the landowner. Such efforts of landowners are generally not mentioned in definitions of environmental and ecosystem services.

The Millennium Ecosystem Assessment (MA), for example, considers ecosystem services as all the benefits people obtain from nature but not as services some agent conduct for an enlargement or maintenance of such goods. More precisely: "[...] provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits" (MA 2005: 27).

In a similar sense Pagiola (2008) refers to environmental services, for example of forests and landscapes, as of mitigation of greenhouse gas emissions, provisioning services such as energy production and irrigation but also provision of landscape beauty for recreation and tourism. Only the FAO made a distinction in accordance to this consideration described above, stating that "[...] environmental services are externalities, unintentionally provided while producing, for example, food and timber for sale or direct consumption"[11]. Since no consistent meaning could be found according to a literature survey and recent definitions do not sufficiently reflect the man-made nature of biodiversity, the dissertation proposes a consistent definition for the two terms ecosystem and environmental services in the PES context.

Summarizing, the definition of ecosystem services, as applied within the dissertation, corresponds to the definition of the MA. The definition of environmental services corresponds in most part to the definition of the FAO, extending it and explicitly focusing on the meaning of environmental services as intentionally provided services. Therefore within the whole dissertation the term environmental service is used to refer to human induced ecosystem services, whereas services from "nature" without human intervention are called ecosystem services. The term PES is referred to as "Payments for Environmental Services" to underline that humans are rewarded for their actions and services i.e. environmental services, while the connotation of "Payments for Ecosystem Services" was found as redundant.

Chapter 3: Resilienz und nachhaltige Entwicklung von semi-aridem Weideland in Namibia

Chapter 3 focuses on resilience semi-arid rangelands in Namibia (Derissen 2009). Academic discussion of resilience and dynamic of semi-arid rangelands and its implications for management started with Holling's remarks on resilience and grazing ecosystems in 1973 and continuing with Walker & Noy-Meir in the early 80's and their article about the 'Aspects of the Stability and Resilience of Savanna Ecosystems' (Walker & Noy-Meir 1982). Up to this point the management of rangelands based on the paradigm of resilience was firmly established[12].

Within the savanna ecosystem of rangelands the system is thought to have at least two different basins of attraction, one grass-dominated and one bush-dominated state. Concern is raised that the system shifts from the former to the latter. To what extend grazing impacts the shift from the grass-dominated to the bush-dominated state is a matter of intense academic debate (see Vetter 2005, Briske et al. 2003 and Jakoby 2011). Some authors (e.g. de Klerk 2004 and Perrings & Walker 1997) argue that the absence of frequent burning

[11] http://www.fao.org/es/esa/pesal/aboutPES1.html

[12] The management of coral reefs (Hughes et al. 2003) and the stylized model of the shallow lake (Mäler et al. 2003) are examples of topics often raised which link the resilience paradigm to ecosystems.

and climate change are the key factors stimulating bush encroachment. Jakoby (2011) on the other hand, states that livestock grazing in combination with low precipitation has a substantial negative impact on the vigor of grass and vegetation conditions.

The main utilization of the rangelands is livestock grazing. Thus, the society is currently dependent on and aligned to the grass-dominated state which constitutes the desired state of the system. Managing efforts aim at keeping the system in this actual state. From this perspective it is important to identify and assess the relationship and interactions of the system components to find new and adjust approved management strategies to maintain the grass-dominant state. By contrast, this chapter reflects this ambition. It asks for possibilities to use the bush-dominated state in a way that allows that society to gain the same or at least a sufficient amount of utility in comparison to the utility that society gains from a grass-dominated state. In general the question was raised if the current state of a system is necessarily to be protected or are there other adaptation possibilities. The case study sets up the conceptual background for one of the leading research questions of the dissertation: Is resilience of a systems state a precondition for sustainability of an ecological-economic system?

On the basis of the described case study research conjectures have been formulated: As for a case of a system state which is highly resilient but does not provide an ecosystem service on a desired level, resilience might be an obstacle due to a regime shift into a desired state of the system. Additionally a system which can provide a desired level of ecosystem services within two system states would not be dependent on the resilience of one system state. Therefore resilience might be necessary or not for sustainability depending on the initial system state. Chapter 4 verified these hypotheses on a basis of an analytical ecological-economic model featuring multiple basins of attraction.

Chapter 4: The relationship between resilience and sustainability in ecological-economic systems

Chapter 4 analyses the question if resilience is a precondition for sustainability (Derissen et al. 2011). As discussed within the introduction, society regards a system within a certain basin of attraction as desirable or undesirable depending on the flows of goods or services it yields. Before management decisions can be made one has to decide which state of a system should be reached or avoided and under a paradigm of sustainability a desired state should be maintained in the long term. Sustainable development in this sense can be understood as development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED 1987); a definition the thesis refers to. Thus, concerning obligations towards future generations, the primary question of sustainability is to what extent ecosystem services, and therefore certain resources, have to be maintained to enable future generations to meet their needs.

Some authors argue that resilience is a precondition for this long term maintenance, which raises the question of how the concepts of resilience and sustainability are connected in general and how the paradigm of resilience frames sustainability. The dissertation was concerned to present and define resilience and sustainability in both analytical and conceptual way. A system state in terms of the dissertation therefore falls within sustainability if the associated ecosystem service, which a society generates utility from does not fall below a certain baseline, here called the "sustainability threshold". This requirement corresponds to a conception of "weak sustainability" as defined by Ott & Döring (2004). As for the paradigm of resilience the definition of Holling (1973) was chosen, as described. Here resilience is understood as "[...] the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior" (Holling & Gunderson 2002: 4).

It can be assumed that a desired state has a better chance to be maintained in the long run if the system state is resilient to many disturbances. But is the resilience of a given system state therefore necessary for sustainability i.e. the maintenance of the stock flow, in the long run? In this respect the dissertation sets out to contribute to this question while reflecting the relation of resilience in ecological-economic systems to sustainability i.e. the maintenance of natural capital (and capital flow) over time. In the first instance the chapter of the dissertation reflects definitions and perspectives of recent literature: Resilience is often seen as a precondition for a sustainability, for example in Lebel et al. (2006) and (Holling & Walker 2003). Levin et al. (1998: 221) argue that "[...] the concept of resilience offers a useful way of thinking about the sustainability not just of environmental processes, but of social and economic processes as well". On the other hand, if a system state is undesirable, resilience would obviously be an obstacle to sustainability. Examples of such "lock-in" situations which are in the way of change have been identified by Hanna et al. (1996), like for example cultural or institutional settings or the dependence on a specific technology. In this sense Holling & Walker (2003: 1) noted that "[r]esilience, per se, is not necessarily a good thing. Undesirable system configurations (e.g. Stalin's regime, collapsed fish stocks) can be very resilient, [...]".

Up to this point two scenarios therefore have been addressed: A system is either within desired system state and should be maintained there or it is trapped within an undesired state with the necessity of change. The statements quoted above do not take into account the following possibilities, which are addressed by the dissertation: Due to the assumed system properties a certain, desired amount of natural capital can fall below a given sustainability threshold even within a basin of attraction. Additionally, given a system with multiple stable states, more than one basin of attraction can yield the desired level of utility.

As a consequence, a system with a desired system state which is not resilient against a certain disturbance might change its basin of attraction because of such disturbance. In doing so, it could reach a new basin of attraction which does also qualify as a desired state in respect to sustainability. Thus, the system management might still achieve sustainability of the system, even if the basin of attraction has changed. In general, four relationships between resilience and sustainability are logically possible. These are discussed with their assumptions within the dissertation thesis: a) resilience of the system is necessary, but not sufficient, for sustainability; b) resilience of the system is sufficient, but not necessary, for sustainability; c) resilience of the system is neither necessary nor sufficient for sustainability; and d) resilience of the system is both necessary and sufficient for sustainability. Any of those may hold in a given system.

On the basis of a simple dynamic ecological-economic system with multiple stable states the first paper of the dissertation shows that resilience is not a goal in itself. Rather, it only constitutes a necessary and sufficient criterion for sustainability in cases where no other systems state exists, which corresponds to the claim of sustainability (Derissen et al. 2011). As a consequence, one may conclude that resilience is neither desirable in itself nor is it in general a necessary or sufficient condition for sustainability. Thus, in general the deduction from sustainability to resilience, or vice versa, is not possible. In particular, the property of resilience should not be confused with the positive normative connotations of sustainability.

Chapter 5: Consumer preferences determine resilience of ecological-economic systems

Ecosystem shifts may threaten the inter-temporal efficiency of ecosystem services, although such shifts do not necessarily imply such deterioration in ecosystem service provision (see Chapter 4 and Derissen et al. 2011). Furthermore, the resilience of an ecological-economic system and the stability landscape of such systems is neither immutable nor and always stable. Working with the ecological-economic model of Chapter 4 it was becoming quite obvious that the dynamic of the ecological-economic system's responses to changes of several variables of the model with quite different outcomes regarding the stability landscapes (Baumgärtner et al. 2011).

Horan et al. (2011) point out that the stability landscape of an ecological-economic system may be changed through institutional arrangements. As shown within this chapter, altering access to resources, complementarities of ecosystem services and species competition also induce a change to the dynamic of an ecological-economic system. The literature shows that species interaction and competition can destabilize the dynamic property of a system (e.g. Ives & Carpenter 2007 and Scheffer 2009). However, the consequences of changing economic resource use have so far not yet been investigated in detail. Chapter 5 analyzes which driving forces affect the alteration of systems' resilience and proves that consumer preferences change the resilience and stability landscapes of ecological-economic systems[13].

With respect to the ecosystem management, resilience as a descriptive concept does not necessarily provide guidance for the "right" actions and has to be seen as a merely descriptive concept (Derissen et al. 2011). Moreover since in the given meaning a system only is declared resilient ex post, i.e. after a disturbance took place. With respect to this certain disturbance no management advices are possible ex ante. Consequently, another concept is needed to guide management recommendations before a disturbance took place, together with a prescriptive guideline of a desired state. For this purpose it is necessary to measure resilience to estimate the probabilities of a given system state to change into another basin of attraction (Baumgärtner et al. 2011).

To analyze and describe a change of the resilience of an ecological-economic system necessitates measuring resilience. This constitutes a matter of ongoing debate (Brand 2009). Carpenter et al. (2005) assume that resilience cannot be measured directly. Instead they argue, it is necessary to draw on surrogates such as ecological redundancy, response diversity (e.g. Bellwood et al. 2003 and Bengtsson et al. 2003) or the concept of maintained

[13]Consumer preferences are understood as preferences that consumers hold over directly consumed commodities of the ecosystem and the interrelation of the given resources.

system identity (Cumming et al. 2005)[14]. These could be used as indicators for resilience. Within this chapter resilience is not measured directly and quantitatively but the resilience of the ecological-economic system is said to decrease as the distance between corresponding stability basins decreases, and also as the number of alternative basins of attraction increases[15].

Within a stylized ecological-economic model an ecosystem under natural conditions, i.e. with no harvest, is firstly simulated based on a Lodka-Volterra model for a baseline. Secondly, a system under open access and profit-maximizing resource harvesting is analyzed. The setting is similar to the model of Chapter 4: Two resources are harvested for a society's well-being, and consumers' utility is based on the ecosystem services the resources provide. Additionally to that, low complementarity between the resources is assumed with no inter-species competition. The system is then studied under rising complementarity between the ecosystem resources and with an increase of the competition between the species. Finally, the alterations of the system's stability landscape are compared with the system's natural dynamics.

Summarizing the results, Chapter 5 demonstrates that consumer preferences are an important determinant of the dynamic characteristics of coupled ecological-economic systems. In detail three destabilizing effects have been identified for the assumed ecological-economic model which genuinely stems from consumer preferences in an ecological system used for economic purposes: At first profit-maximizing harvesting by competitive firms under open access considerably weakens the resilience of the basin of attraction as compared to the natural dynamics. Second, complementarity of ecosystem services in consumption significantly reduces the resilience of the system's interior equilibrium where both species are in existence. Finally increased competition between species destabilizes the system and may lead to multiple basins of attraction.

[14]For a comprehensive overview of the discussion on the measuring of resilience see Brand (2009).

[15]Notice that in Chapter 4, resilience is described with regard to a specific disturbance.

Chapter 6: Combining performance-based and action-based payments under uncertainty

The final paper of the dissertation thesis presents a concrete design proposal for conservation contracts (Derissen & Quaas 2013). As described within the introduction, usually two subcategories of payment schemes are distinguished: (i) performance-based payments and (ii) action-based payments. Action-based payments are bound to the implementation of predefined actions whereas with a performance-based payment agents are rewarded for the maintenance or achievement of a concrete conservation objective. Within the literature the advantages and disadvantages of both payment schemes are discussed at large which are summarized here[16].

One predefined action might be efficient in one agricultural business but not in another, and the best action alternatives might change due to changes in seasonal annual weather conditions or because of internal changes of the farm structure. Since the agent is assumed to be better informed about the conditions of her land, a loss of efficiency might be the result in the case of action-based payments.

For performance-based payment schemes it is assumed that better incentives are provided for the agent to innovate and find the most cost-effective action to produce the desired ecosystem good. In addition to this, performance-based payments can support the cooperation between farmers although an advantage can obviously only be presumed if the environmental good is necessarily bound to the protection of a bigger area. Another advantage is that the principal in many cases cannot validate the actions or compliance in detail for an action-based contract, whereas performance-based payments are only paid out if the desired goal was achieved.

With all these arguments, the literature assumes performance-based payments are preferable to action-based payments: “If we want to get what we pay for, we must start tying our investments directly to our goals” (Ferraro & Kiss 2002: 1719). However as indicated within the introduction of the thesis this, as always, depends on the context and goals: In the case of an action-based payment the burden of risk to produce a desired environmental good is at the expense of the principal, since the agent is paid for the predefined actions and not for the achievement of the intended goal. On the other hand performance-based payments bear a greater risk for the agent, since the performance of the environmental good is in most cases not only dependent on their actions but on external circumstances as well, which modifies the performance. In this case the agent eventually would not consent to a performance-based payment scheme since a payment might not compensate their

[16]See for example Hampicke (2001), Ferraro & Kiss (2002) and Matzdorf (2004).

expenditure costs.

Therefore with this argument an action-based contract is chosen in many cases regardless of the disadvantages of action-based payment schemes in case of e.g. cooperation markedability, self-interest of the agent or compliance. Performance-based payments are usually applied for the conservation of an already given state of existent biodiversity (Hampicke 2001). Or in addition to performance-based payments, a base-payment might be offered to compensate the agent's opportunity costs of the conservation contract (Zabel & Roe 2009).

So naturally the question arises: How can contracts be designed to deal with risks of the production and can risk be fairly distributed for the desired environmental good and include the advantages of performance-based payments, such as lower probabilities of cheating, more self-interest and cooperation? Furthermore, since uncertainty and information asymmetries are not interdependent but likely to be positively associated, the dissertation presents a combination of both payment schemes. Within this chapter, insights are developed with the help of a principal-agent model. With different assumptions regarding the risk profile of the principal and with different uncertainty levels for the production of the ecosystem good, the optimal combination of performance-based and action-based payments are examined. Furthermore the model has been modified to demonstrate the situation regarding the case for a risk-averse regulator. Here, the situation changes as follows: While the regulator is not directly affected through environmental uncertainty, he is indirectly through the farmer's choice of action, since the farmer chooses his actions considering environmental uncertainty. Thus, the performance-based payment tends to pay out less favorably compared to the action-based payment for the regulator.

As a result of the model's insights, it can be concluded that an exclusively performance-based payment is optimal only if there is no environmental uncertainty or if both the farmer and the regulator are risk-neutral. An exclusively action-based payment is optimal only if the regulator has full information about the productivity of the action i.e. if there is no information asymmetry. In every other case a combination of performance-based and action-based payments (with different weightings) increases welfare.

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Chapter 2

What are PES? - A review of definitions and an extension

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January 2013

This article was published in *Ecosystem Services* **6** (2013): 12–15.
http://dx.doi.org/10.1016/j.ecoser.2013.02.002

What are PES? - A review of definitions and an extension

Abstract: The term PES is often used to denote market incentives for the provision of public goods within the field of environmental and resource issues. In this context, PES translates into either "Payments for *Environmental* Services" or "Payments for *Ecosystem* Services" - the terms that are not consistently defined in the literature and sometimes used as synonyms. Given the lack of coherent definitions, this note reviews current definitions of payments for ecosystem services and payments for environmental services entertained in the literature, discusses alternative meanings of environmental and ecosystem services in the PES context, and finally proposes a consistent definition. We argue that current definitions of PES found in the literature are insufficient to adequately describe the man-made nature of many environmental goods and services: that nature is 'produced' through human intervention. Building upon the FAO's definition of environmental services, we propose a definition that regards environmental services as services provided through countryside management in a broader sense whilst produced either unintentionally or intentionally.

Keywords: PES, ecosystem services, environmental services, conservation contracting, market-based incentives

JEL-Classification: Q57

2.1 Introduction

Public goods are not normally provided in sufficient quantities, but often produced as by-product or externality of primary production activities. One response to this market failure has been for governments and NGOs to implement policies and incentives which reward landholders to maintain or enhance such goods.

The acronym PES has been widely used to refer to these "nascent market creation incentive mechanisms"(Pascual & Perrings 2007: 256). However, different authors use PES with different definitions in mind, and there exists no agreement in the literature what PES actually means. “ES” is translated either as environmental or as ecosystem services. Concerning this terminological inconsistency, it is interesting to note that ecosystem services is the more explicitly defined term in the literature and most authors agree about its meaning, whereas the definition of environmental services is more ambiguous. This raises the question as to how the terms “Payments for Environmental Services” and “Payments for Ecosystem Services” are to be interpreted.

Against this background, this commentary sets out to: (i) review definitions of ecosystem services and environmental services entertained in the literature; (ii) discuss alternative meanings of environmental and ecosystem services in the PES context; and (iii) propose a consistent definition of the terms.

2.2 Review of terms and definitions

2.2.1 Ecosystem services versus environmental services

Initiated in 2001 by the United Nations Secretary-General Kofi Annan, the Millennium Ecosystem Assessment (MA) aimed at assessing the consequences of ecosystem change for human well-being and establishing the scientific basis for actions needed to enhance the conservation and sustainable use of those systems and their contribution to human well-being[17]. The MA reports provided a definition of ecosystem services which has become widely accepted in the academic community. In fact, most authors have subsequently drawn on the MA’s definition of ecosystem services as a key reference (Corbera et al. 2007, Kroeger & Casey 2007, Pascual & Perrings 2007, Engel et al. 2008, Jack et al. 2008, Chen et al. 2009, Carpenter et al. 2009, Leimona et al. 2009, Norgaard 2010, Pascual et al. 2010, Sommerville et al. 2009, Swallow et al. 2009, Zabel & Roe 2009).

According to the MA, ecosystem services are all the benefits people obtain from ecosystems. These include "[. . .] provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services

[17]http://www.maweb.org/en/About.aspx.

such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits"(MA 2005: 27). Along the same lines, the Food and Agriculture Organization of the United Nations (FAO) describes ecosystem services as: "[...] all benefits that humans receive from ecosystems. These benefits can be direct (e.g. food production) or indirect, through the functioning of ecosystem processes that produce the direct services"(FAO 2012a)[18].

Definitions are less clear cut when it comes to environmental (as opposed to ecosystem) services, and attempts to distinguish the one from the other add to the confusion. There are barely two authors sharing the same definition of the term environmental services. The following review does neither claim nor intend to be exhaustive; it rather aims to provide an overview of definitions and of how the terms environmental and ecosystem services are used and interpreted in the literature.

The FAO states that the terms ecosystem services and environmental services are sometimes used interchangeably. However, the FAO considers environmental and ecosystem services as distinct concepts. On the one hand, environmental services are characterized as externalities, unintentionally provided while producing food or timber for sale or direct consumption, whereas ecosystem services are defined, according to the MA characterizations, as all the benefits that people obtain from ecosystems, as shown above. Environmental services are seen as a subset of ecosystem services (FAO 2012b)[19].

Muradian et al. (2010: 1202) provide another definition which in some way gives an opposite meaning to the FAO's understanding of environmental services while explicitly refering to the distinction of ecosystem services and environmental services by noting "[...] that ecosystem services is a subcategory of the former, dealing exclusively with the human benefits derived from natural ecosystems. Environmental services also comprise benefits associated with different types of actively managed ecosystems, such as sustainable agricultural practices and rural landscapes". Apparently, Muradian et al. (2010) consider environmental services not to be a subset of ecosystem services, as assumed by the FAO. Conversely, they regard ecosystem services as a subset of environmental services.

Myers (1996) also regards ecosystem services as a subset of environmental services, but in his definition the crucial factor for distinguishing between both categories is the scale of the service: Ënvironmental services are also known as ecosystem services, both terms reflecting environmental functions and ecological processes. They can be defined as any functional attribute of natural ecosystems that are demonstrably beneficial to humankind. The term environmental services is preferred since it embraces the larger-scale and often

[18]http://www.fao.org/es/esa/pesal/aboutPES1.html.

[19]http://www.fao.org/es/esa/pesal/aboutPES2.html.

more important services [...]"(Myers 1996: 2764). With reference to Scherr et al. (2004), Wunder (2005) finally considers environmental services to be separable in nature, whereas the term ecosystem services "probably has a more integral interpretation, implying that multiple services cannot always be broken up into additive components"(Wunder 2005: 4).

Pagiola (2008), quoting the Forest Law No.7575 of Costa Rica, refers to the mitigation of greenhouse gas emissions, hydrological services and to the provision of science beauty as environmental services. This definition does not add clarity to the distinction between ecosystem and environmental service since the description seems congruent with the MA's definition of ecosystem services (MA 2005). Therefore one might assume that the terms have been used interchangeably. Myers (1996), Engel et al. (2008) and FAO (2012a,b,c) also note that environmental services and ecosystem services are often used as synonyms.

2.2.2 Payments for ecosystem versus payments for environmental services

At the international scale, the debate about payments for ecosystem or environmental services has attracted increasing attention (e.g. Ferraro & Kiss 2002, Wunder 2005, Pagiola et al. 2007, Engel et al. 2008, Ferraro 2008, Pascual et al. 2010, Rodríguez et al. 2011, Garbach et al. 2012, Xuan To et al. 2012). The debate focuses inter alia on the institutional background needed to implement PES schemes, especially how such schemes can be implemented in the context of weak institutions, and how in developing countries PES schemes might help to benefit the poor and contribute to a more efficient and equal distribution of resources (e.g. Wunder et al. 2008, Bulte et al. 2008, Muradian et al. 2009, Norgaard 2010). Naturally, PES was not the first and only term which refers to the maintenance and enhancement of amenities provided by nature. Other terms include, in chronological order:

- Conservation payments (Ferraro & Simpson 2002)
- Rewards for ecological goods (Gerowitt et al. 2003)
- Agri- environmental payments (Cooper 2003)
- Payment schemes for environmental services (Tomich et al. 2004)
- Agri-environmental subsidies (Wittig et al. 2006)
- Rewards for ecosystem services (Pascual & Perrings 2007)
- Rewards for environmental services (Leimona et al. 2009)
- Compensation and rewards for environmental services (Swallow et al. 2009)
- Incentive Payments (Ferraro & Gjertsen 2009)

- Payments for agrobiodiversity conservation services (Narloch et al. 2011)

However with the acronym PES emerging, the number of sources using PES as a reference has been increasing. It is thus warranted to take a closer look at the PES term. The origin of the PES term can be traced back to a World Bank report in 2000 (World Bank 2000). Here the abbreviation PES emerges for a new policy framework in Costa Rica called "pagos por servicios ambientales"(PSA). Within the glossary of the World Bank report the program is denoted (with an English equivalence) as "Payments for environmental services"(PES). We have not been able to identify an earlier source and thus conclude, with the usual caution, that the term PES originates from the World Bank report.

The list of authors using the acronym PES in the sense of payments for ecosystem services is extensive. Without laying claim to completeness, the term was used by: Corbera et al. (2007), Pascual & Perrings (2007), Bulte et al. (2008), Jack et al. (2008), Clements et al. (2010), Norgaard (2010), Chen et al. (2009) and Milder et al. (2010), van de Sand (2012), Garbach et al. (2012). By contrast, the following authors regard PES as payments for environmental services[20]: Wunder (2005, 2007), Wunder et al. (2008), Zbinden & Lee (2005), Zilberman et al. (2006), Sierra & Russman (2006), Pascual & Perrings (2007), Ferraro (2008), Pagiola (2008), Engel et al. (2008), Muradian et al. (2010), Pascual et al. (2010), Zabel & Roe (2009), Sommerville et al. (2009), Vatn (2010), Rodríguez et al. (2011), García-Amado et al. (2011), Tacconi (2012).

What distinguishes payments for environmental services from payments for ecosystem services? According to Wunder (2005) and Bulte et al. (2008), PES are defined as “Payments for Environmental Services” when amenities provided by the built environment are included. Conversely, PES are defined as “Payments for Ecosystem Services” when emphasis is given to enhancing nature services. The distinguishing criterion thus seems to be whether or not amenities provided by the built environment are included in addition to the generic services provided by nature. “Payments for environmental services” thus is the more encompassing term. “Payments for ecosystem services” is therefore a subset of “Payments for Environmental Services”. This is in contrast to the definition put forward by the FAO. For environmental services the FAO states that: “Opportunities for humans to manage environments may range from penalizing negative externalities to introducing more flexible incentive mechanisms such as Payments for Environmental Services (PES). This tool can encourage the conservation and enhanced provision of regulating and supporting ecosystem services, the basis for all other types of service” (FAO 2012c)[21].

[20] Wunder, Engel, Pagiola and Zabel have joint publications and therefore share the same definition of PES - in the sense of payments for environmental services.

[21] http://www.fao.org/es/esa/pesal/aboutPES4.html.

2.3 Discussion - A framework for a consistent definition of PES

From the previous section it is apparent that no consistent definitions of the terms environmental and ecosystem services exist: Should ecosystem services be understood as a subsystem of environmental services, or vice versa? Do environmental services have a more holistic interpretation as Wunder (2005) states? Should one think of environmental services as services on a larger scale as Myers (1996) suggests, or is there in the end nothing to add to the debate as to accept that environmental and ecosystem services are synonyms and PES is as an umbrella term irrespective of how the acronym is spelled out? In what follows, we set out a framework for a consistent definition of terms.

In our view, there is no necessity to think of ecosystem services in terms other than the MA (2005). We thus adopt this well-established definition as a starting point. According to this definition, direct ecosystem goods and services are thought of as goods and services directly provided by nature as set out above. This is in line with the FAO's definition of ecosystem services as products of nature or as products that are in some sense closer to nature as environmental services.

According to the FAO, environmental services are services provided as side-products of human production. This definition implies the assumption that the environmental goods and services are provided unintentionally. The FAO's definition thus ignores the subset of environmental goods and services that are produced intentionally, for example through conservation-minded land management or eco-certification. This is a serious shortcoming especially in the PES context. PES schemes explicitly target this subset of man-made environmental benefits by paying landholders to provide them. We thus argue that intentionally produced environmental benefits should be included in the definition of environmental services. This holds irrespective of whether the environmental benefits are classified as externalities of primary production (e.g. environmental enhancement through conservation-minded management of productive agricultural land) or whether they are themselves primary products (e.g. environmental benefits from the creation of a swamp on non-productive land). What counts is that these services are produced intentionally.

Our definition of environmental services then encompasses both categories of environmental benefits - those produced intentionally and those produced unintentionally. In a more general sense, our proposed definition classifies countryside benefits as environmental services when human input influences ecosystem goods and services as part of a production process. In contrast to Muradian et al. (2010), we do not therefore regard environmental services as a subsystem of ecosystem services but rather as a systematically different category.

If one accepts this definition the question arises of how the term "Payments for Ecosystem

Services" might be defined in a context where the ecosystem goods and services under consideration are exclusively provided by nature, i.e. without human input. We argue that "Payments for *Ecosystem* Services" is a redundant term in that nature does not need to be paid (and cannot technically be paid) for the flow of goods and services provided to humankind[22]. Since only humans can be paid (and in many cases need to be paid) for the provision of environmental benefits, there can only exist one meaning of the acronym PES: namely "Payments for *Environmental* Services" in the sense of payments *to the provider* of environmental services.

In Germany for example, the term "Honorierung ökologischer Leistungen" is commonly used to denote PES. This term translates into remunerations for ecosystem services (Matzdorf 2004). The focal point here is that remunerations, in contrast to subsidies, are regarded as payments in return for a service provided by the landholder. Service provision implies effort being exerted by the landholder. Landholders are thus remunerated for their intentional actions to enhance the quality of the environment. Within our proposed definition the term "Honorierung ökologischer Leistungen" is thus referring to the category of environmental services that are produced intentionally[23].

2.4 Conclusion

The terms environmental services and ecosystem services have been used inconsistently and in some cases interchangeably in the literature. While the MA's (2005) definition of ecosystem services has become widely accepted in the academic community, no agreement has been reached on the definition and use of the term environmental services. A similar argument holds for the definition of the terms "Payments for Ecosystem Services" and "Payments for Environmental Services". We argue that the definitions of environmental services entertained in the literature are not sufficient to capture the man-made nature of many environmental goods and services: That such benefits, especially those targeted by PES schemes, are usually "produced" through intentional human intervention. Against this backdrop, we propose a definition of environmental services which builds upon the FAO's definition, but broadens it to cater for the man-made nature of environmental services, irrespective of whether these are produced intentionally (e.g. through environmental

[22]Prevention of actions may also be costly to the extent that conservation of the status quo often involves opportunity costs. For example, if a landholder does not drain a piece of species-rich wetland, he will not realize the potential gain from a more profitable use of the land. The argument here remains the same since it is not nature that is rewarded but the ecosystem manager who receives a payment for providing the service of not draining the wetland.

[23]From a political perspective, it is important to draw a clear line of distinction between "remunerations" (or "payments") on the one hand and "subsidies" on the other. While subsidies are simply transfers, remunerations are payments in return for service. This distinction is particularly important in the context of the WTO rules: most "subsidies" are subject to reduction requirements, whereas payments for the provision of environmental services are permitted.

contracting schemes) or unintentionally (by simply farming the land). It follows from this definition that the acronym PES can only refer to "Payments for Environmental Services" since payments can only be made in respect of man-made conservation activities. "Payment for ecosystem services", i.e. those produced by nature without human intervention, is a redundant term because nature does not have a bank account.

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Chapter 3

Resilienz und nachhaltige Entwicklung von semi-aridem Weideland in Namibia

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Mai 2009

This article was published in: Egan-Krieger, T., Schultz, J., Thapa, P. & Voget, L. (eds.): Die Greifswalder Theorie starker Nachhaltigkeit, Metropolis–Verlag, Marburg (2009): 299–312.

3.1 Einleitung

Aufgrund der semi-ariden bis ariden Bedingungen werden die größten Teile von Namibias Landesfläche durch extensive Weidewirtschaft genutzt. Jedoch scheint sich das Ökosystem im Umbruch zu befinden: Das vermehrte Auftreten von Büschen auf den Weideländern (bush encroachment oder Verbuschung) ist ein schwerwiegendes und scheinbar nicht zu stoppendes Phänomen. Die Tiere finden nicht mehr genügend Futter und überweiden die Flächen weiter, und auch in Jahren mit ausreichend Regen können sich die Weiden nicht mehr vollständig regenerieren. Der Verlust für die namibische Wirtschaft durch den Rückgang der Bestockungsraten wird auf 86 Millionen US-Dollar pro Jahr geschätzt (de Klerk 2004).

Die Ursachen der Verbuschung werden in den zu hohen Bestockungsraten in den 1960er Jahren vermutet. Doch hat auch eine gezielte weitere Reduktion der Bestockung das Fortschreiten der Verbuschung in den Weideländern (rangelands) bisher nicht aufhalten oder umkehren können. So wird vermutet, dass durch die Überweidung die Resilienz des Systems bereits überschritten wurde und Namibias semi-aride Weideländer ohne weitere Gegenmaßnahmen von einem Gras-dominierten in einen Busch-dominierten Zustand "umkippen" könnten.

Welche Rolle die Resilienz ökologisch-ökonomischer Systeme bei der nachhaltigen Nutzung semi-arider Weideländer in Namibia spielt oder spielen kann, soll im Folgenden vor dem Hintergrund der Konzeption starker Nachhaltigkeit diskutiert werden. Es geht dabei nicht nur um die Darstellung und Anwendung des Resilienzkonzeptes, sondern auch um seine Kritik: Ich möchte diskutieren, welche Möglichkeiten das Resilienzkonzept bietet, aber auch abgrenzen, was Resilienz gerade nicht sinnvoll sein kann – nämlich ein Entscheidungskriterium für das normativ "Richtige".

In Abschnitt 3.2 werden zunächst die generellen Ursachen und die Wahrnehmung der Verbuschung in Namibia vorgestellt und diskutiert. Abschnitt 3.4 führt in die von mir zugrunde gelegten Begriffe von Resilienz und nachhaltiger Entwicklung und deren Relation ein. In Abschnitt 3.5 wende ich diese theoretischen Überlegungen auf Verbuschung und Bewirtschaftung semi-arider Weideländer in Namibia an und diskutiere die Implikationen von Resilienz für eine nachhaltige Entwicklung semiarider Weideländer, bevor ich meinen Gedankengang in Abschnitt 3.6 zu einem Fazit zusammenfasse.

3.2 Namibia - Land und Wirtschaft

Namibia ist eines der trockensten Länder Afrikas – das trockenste südlich der Sahara. Die mittleren Jahresniederschläge liegen in den südwestlichen Regionen bei weniger als 50 Millimetern pro Jahr und steigen nach Nordosten hin auf 700 Millimeter pro Jahr an.

Insgesamt bezeichnet man die klimatischen Bedingungen als arid bis semi-arid. Aus einer landwirtschaftlichen und ökologischen Perspektive ist neben der Knappheit der Niederschläge vor allem ihre hohe Variabilität ausschlaggebend. Diese beträgt im Norden des Landes ca. 30 Prozent und erreicht im Süden und Westen 70 Prozent. Das heißt, dass die Wahrscheinlichkeit des Regeneintritts in der erwarteten Regenzeit sinkt, je weiter man sich von Nordosten der Küste nähert.

Aufgrund dieser klimatischen Verhältnisse, hat Namibia insgesamt ein geringes Potential für den Anbau von Feldfrüchten. Die benötigte Niederschlagsmenge für Regenfeldbau beträgt unter diesen klimatischen Bedingungen mindestens 500 Millimeter pro Jahr. Nur acht Prozent der Landesfläche entsprechen diesem Kriterium. Daher prägen landesweit vor allem Rinderzucht und Kleinviehhaltung die Landwirtschaft. Von der Resilienz und nachhaltige Entwicklung von semiaridem Weideland landwirtschaftlich nutzbaren Fläche entfallen 48 Prozent auf Fleischrinderfarmen, 15 Prozent auf Mischbetriebe (Klein- und Großtierhaltung, teilweise mit Ackerbau) und 37 Prozent auf Betriebe mit Kleintierhaltung. Zwar stellt die Landwirtschaft, und das heißt vor allem die semiaride Weidewirtschaft, nur einen geringen Anteil am Bruttosozialprodukt dar (2003 nur rund 4,8 Prozent), doch sind rund 70 Prozent (2006) der 2,1 Millionen Menschen Namibias ganz oder teilweise im landwirtschaftlichen Sektor tätig oder von diesem abhängig. Da neben Bodenschätzen hauptsächlich Fisch- und Fleischprodukte exportiert werden, spielen landwirtschaftliche Erzeugnisse zusätzlich bei den Exporteinnahmen eine große Rolle. Durch die Ausfuhr von landwirtschaftlichen Produkten (vor allem Tafeltrauben, Fisch und Rindfleisch) erwirtschaftete Namibia im Jahr 2006 252 Millionen US-Dollar. Ein Viertel der Güter wird dabei in die Länder der südafrikanischen Zollunion exportiert, vor allem nach Südafrika. Durch die hohen Gewinnspannen für Rindfleisch ist monetär aber die Europäische Union (vor allem Großbritannien) der wichtigste Exportpartner der namibischen Weidewirtschaft, mit einem Gesamterlös von über 124 Millionen US-Dollar pro Jahr (2006)[24]. Ohne die Verbuschung der Weideländer, so wird geschätzt, könnten die Erlöse für die gesamte Weidewirtschaft allerdings um rund 86 Millionen US-Dollar pro Jahr höher ausfallen (de Klerk 2004: 5). So ist es kein Wunder, dass die fortschreitende Verbuschung der Weideländer mit Sorge beobachtet wird.

3.2.1 Verbuschung als ein Problem der Desertifikation

Veränderungen der Savannenökosysteme, vor allem das verstärkte Auftreten von Büschen, verringern die Grasproduktivität der semi-ariden Weideländer, das heißt das Futterangebot für Rinder und damit auch die Bestockungsrate. Die Reduktion des Grasbestandes kann außerdem zu einer Verstärkung von Bodendegradierung und Erosion führen, in anderen

[24] https://www.cia.gov/library/publications/the-worlfactbook/print/wa.html

Worten zu Desertifikation.

Unter Desertifikation wird allgemein die Landdegradierung in trockenen Klimaten verstanden. Das Übereinkommen zur Bekämpfung der Desertifikation, die United Nations Convention to Combat Desertification (UNCCD), definiert Desertifikation als "land degradation in arid, semiarid and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (UNCCD Art.1a: 4) [25]. Landdegradierung ist also in diesem Falle definiert als eine spezielle Form der Desertifikation in ariden und semiariden Gebieten. Die Food and Agriculture Organization (FAO) versteht unter Desertifikation eine durch menschlichen Einfluss hervorgerufene, irreversible Degradation natürlicher Ressourcen in ariden und semi-ariden Klimaten, hebt also zusätzlich die Unumkehrbarkeit der Schädigung hervor[26].

Die Verbuschung in Namibia ist zwar umkehrbar, wird aber als ein Teil der Desertifikation verstanden und beschrieben als "the invasion and/or thickening of aggressive undesired woody species, resulting in an imbalance of the grass:bush ratio, a decrease in biodiversity, a decrease in carrying capacity and concomitant economic losses" (de Klerk 2004: 2). Das Argument für die Anwendbarkeit des Desertifikationsbegriffs lautet hier, dass Verbuschung zwar prinzipiell umkehrbar, im Rahmen von ökonomisch relevanten Zeiträumen jedoch als irreversibel zu betrachten ist. Die namibische Verfassung von 1990 sieht ausdrücklich vor, dass der Staat auch durch den Erhalt von Ökosystemen, essentiellen ökologischen Prozessen und der biologischen Diversität in Namibia zum Wohl der Bevölkerung beiträgt. Dies umfasst auch Maßnahmen gegen die Verbuschung der Weideländer. So lautet eine Grundzielsetzung der National Agriculture Policy des Ministry of Agriculture, Water and Rural Development of Namibia: "[...] endeavour to ensure that appropriate bush control technologies and inputs are available from the private sector at the lowest possible prices" (Ministry of Agriculture, Water and Rural Development 1995: 29).

Zusammenfassend lässt sich festhalten, dass Verbuschung in Namibia als ein Problem der Landdegradierung und als eine Art der Desertifikation gesehen wird. Entsprechende Gegenmaßnahmen werden durch die UNCCD abgedeckt und gefördert. Weitreichende politische Maßnahmen unterstreichen die Absicht, Weideland in seinem Gras-dominierten Zustand zu erhalten.

3.2.2 'Cattle or Climate' – Die Ursachen der Verbuschung

Vor der generellen Debatte um nachhaltiges Management von Savannenökosystemen steht die Frage, welche Faktoren Verbuschung überhaupt auslösen und begünstigen. Die Zu-

[25] Siehe http://www.unccd.int/Lists/SiteDocumentLibrary/conventionText/conv-eng.pdf

[26] http://www.fao.org/desertification/default.asp?lang=en

sammenhänge zwischen einer erhöhten Bestockungsrate und der Verbuschung sind nicht abschließend erforscht. Mehrere Theorien stehen einander in ihren Empfehlungen entgegen: Die Anhänger der Equilibrium-Theorie oder Gleichgewichtstheorie beziehen die Veränderungen des Savannenökosystems auf die hohe Bestockung mit Vieh. Vergleicht man die Bestockungsraten von heute aber mit denen der 1960er Jahre, wird deutlich, dass die Bestockungsraten von um fast 70 Prozent zurückgegangen sind (2001). Dennoch schreitet die Verbuschung voran.

Anhänger der der Non-Equilibrium-Theorie von Savannendynamiken argumentieren daher, dass die hohe Variabilität der Niederschläge bei weitem der bestimmende ökologische Faktor sei. Folglich seien Vegetation und Viehdynamik in einem solchen Ausmaß entkoppelt, dass auch hohe Bestockungsraten keinen Einfluss auf die Vegetationsbedeckung haben (Behnke & Scoones 1993; Elias & Swift 1988). Als Resultat empfehlen zum Beispiel Westoby et al. (1989: 266) den Farmern ein opportunistisches Verhalten: "[F]armers should response to climatic variability by adjusting stocking rates in order to make maximum use of grass production".

Als Opfer klimatischer Ereignisse hätten Landwirte in diesem Szenario keine Verantwortung für die Verbuschung und könnten diese auch nicht verhindern. Anhänger der Equilibrium-Theorie dagegen empfehlen niedrige Bestockungsraten, um die Dominanz der aufkommenden Büsche gegenüber den Gräsern zu verringern. Auch empirische Studien in Südafrika bekräftigen die These, dass hohe Bestockungsraten die Vegetationsdynamik der Weideländer beeinflussen können (Fynn & O'Connor 2000).

Eine dritte Theorie sieht die Hauptursache der Verbuschung in der Verhinderung natürlicher Feuerereignisse. Regelmäßige natürliche Feuerereignisse dezimieren die Büsche. Dabei ist es wichtig, dass die Feuer eine bestimmte Temperatur erreichen. Für diese heißen Buschfeuer ist ein großer Anteil an abgestorbenem Gras als Brennmaterial vonnöten. Bei hohen Bestockungsraten sammelt sich jedoch kein abgestorbenes oder trockenes Gras an. Auftretende Feuer sind daher nicht heiß genug und schädigen junge Büsche nicht in ausreichendem Maß und in ausreichender Zahl. Dies mag die wichtigste ursächliche Verbindung zwischen Verbuschung und Viehzucht sein (vgl. Mendelson et al. 2006).

Unabhängig von den genauen Ursachen der Verbuschung, deren Kenntnis aber grundlegend ist für Managemententscheidungen, muss als schlichte Tatsache festgehalten werden, dass sich die Büsche weiter ausbreiten und geringere Bestockungsraten diese Entwicklung bisher nicht aufhalten oder umkehren konnten. In der Sprache der Resilienz-Debatte lässt sich die Situation so beschreiben, dass die Resilienz des Gras-dominierten Zustandes überschritten wurde und die Konvergenz des Systems hin zu einem Busch-dominierten Zustand nicht mehr ohne weiteres umgekehrt werden kann: "It is probable that only complete tree removal, some soil surface modification and reseeding with desirable grasses will overcome this inertia. Such a situation may constitute a different domain of attraction as outlined by Walker & Noy-Meir (1982) and could indicate that the bounds of resilience of the former system have been exceeded" (de Klerk 2004: 28).

Im folgenden Abschnitt betrachte ich die Grundlagen der Resilienzdebatte und den Zusammenhang zwischen nachhaltiger Entwicklung und Resilienz.

3.3 Konzeptioneller Hintergrund

3.3.1 Konzeption von Resilienz

Eine wichtige Annahme zu den Eigenschaften dynamischer Systeme in der Ökologie lautet, dass in diesen Systemen mehrere lokale stabile Gleichgewichte mit mehreren stabilen Zuständen existieren. Da Systeme außerdem von einem dieser stabilen Zustände in einen anderen übergehen können, wenn wie einen bestimmten Schwellenwert überschreiten, ist es möglich, dass sich die zentralen Eigenschaften des Systems grundsätzlich ändern.

Die Annahme alternativer stabiler Zustände in Ökosystemen gab in der Ökologie den Anlass zu der Frage, wo sich ein bestimmtes Ökosystem relativ zu den Schwellenwerten seiner stabilen Zustände befindet. Um den Zustand eines Ökosystems in dieser Hinsicht zufriedenstellend zu beschreiben, muss nicht nur festgestellt werden, ob es Gleichgewichtszustände gibt und wie stabil sie sind, sondern auch, wie groß die Kapazität eines Systems ist, Schocks zu absorbieren und in seinem gegenwärtigen Zustand zu verbleiben.[27] Ich berufe mich hier also auf die Definition von Holling & Gunderson (2002: 4): "[Resilience is] the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior". Ein System zeigt sich demnach resilient gegenüber einer bestimmten Störung, wenn es sich nach der Störung noch im selben stabilen Zustand befindet.

3.3.2 Konzeption von Nachhaltigkeit

Maßgebliches Leitbild der Nachhaltigkeitspolitik für das "richtige" Handeln und den "richtigen" Umgang mit Natur und Umwelt war und ist seit der UN-Konferenz für Umwelt und Entwicklung in Rio de Janeiro 1992 die Nachhaltigkeitsdefinition der sogenannten Brundtland-Kommission: "Sustainable Development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987).

In der wissenschaftlichen Nachhaltigkeitsdebatte lassen sich die Konzepte der starken und der schwachen Nachhaltigkeit unterscheiden, die beide auf der oben genannten De-

[27] Eine genauere technische Definition gibt Holling (1973), zitiert in Holling & Gunderson (2002: 4).

finition aufbauen. Die Konzepte unterscheiden sich grundsätzlich durch ihre Annahmen darüber, in welchem Maße Naturkapital benötigt wird, um menschliche Grundbedürfnisse zu befriedigen, bzw. inwieweit sich Naturkapital durch Sachkapital substituieren lässt (vgl. Ott & Döring 2004). Nur wenn man davon ausgeht, dass Naturkapital vollständig durch Sachkapital ersetzbar sei, ist der Erhalt von Naturkapital irrelevant und sind damit die Überlegungen zur Resilienz ökologischer Systeme überflüssig[28]. Ich folge dem Greifswalder Ansatz starker Nachhaltigkeit (Ott & Döring 2004) darin, dass kritische Grenzwerte von Naturkapital, sowohl einzelner Bestände als auch ihrer Kombinationen, nicht unterschritten werden dürfen.

Die weiteren Ergebnisse dieses Beitrags sind selbstverständlich nur vor dem Hintergrund der gewählten Definitionen als einschlägig zu betrachten. Die Definition von Nachhaltigkeit im Sinne der WCED (1987), die Präzisierung starker Nachhaltigkeit sensu Ott & Döring (2004) und die Definition des Resilienzkonzeptes im Sinne Hollings (1973), auf die ich mich hier berufe, sind aber weithin akzeptiert.

3.3.3 Zum Verhältnis von Resilienz und nachhaltiger Entwicklung

Aufbauend auf einer Vielzahl von Resilienz- und Nachhaltigkeitsdefinitionen finden sich in der Literatur mindestens ebenso viele Versuche, diese beiden Konzepte miteinander in Beziehung zu setzen. So formulieren zum Beispiel Common & Perrings (1992: 28) in einem frühen Aufsatz: “A system may be said to be Holling-sustainable, if and only if it is Holling-resilient”.

Oder es wird im Sinne des Drei-Säulen-Konzeptes von Nachhaltigkeit der Schluss gezogen: “[A] resilient socio-ecological system is synonymous with a region that is ecologically, economically, and socially sustainable” (Holling & Walker 2003: 1).

Wenn wir von diesen Definitionen ausgehen, gibt es keinen plausiblen Grund, Resilienz als notwendig oder gar hinreichend für die nachhaltige Entwicklung ökologisch-ökonomischer Systeme zu betrachten. So schreiben Holling & Walker (2003: 1f.) weiter: “Resilience, per se, is not necessarily a good thing. Undesirable system configurations (e.g. Stalin’s regime, collapsed fish stocks) can be very resilient, and they can have high adaptive capacity in the sense of reconfiguring to retain the same controls on function”. Viel näher liegt der Schluss, dass unerwünschte, aber resiliente Zustände gerade als nicht nachhaltig betrachtet werden sollten (Beispiele in Brander 1998). In diesem Sinne formulieren auch Carpenter et al. (2001: 766): “Unlike sustainability, resilience can be desirable or undesirable”.

Andere Autoren betrachten Resilienz zwar nicht als hinreichendes, aber als ein notwen-

[28]In Arbeiten, die Resilienz als Basis oder zur Operationalisierung von Nachhaltigkeit vorschlagen, wird zwischen starker und schwacher Nachhaltigkeit meist nicht differenziert.

diges Kriterium für Nachhaltigkeit. So vertreten Arrow et al. (1995: 93) in einem vielbeachteten Science-Artikel die Position: "[E]conomic activities are sustainable only if the life-support ecosystems upon which they depend are resilient"[29]. Auch Perrings (2006) stellt fest: "A development strategy is not sustainable if it is not resilient: i.e. if it involves a significant risk that the economy can be flipped from desirable state (path) into an undesirable state (path) and if that change is either irreversible or only slowly reversible" (Perrings 2006: 418).

Die entscheidende Einschränkung ist hier, dass der Übergang in den unerwünschten Zustand irreversibel oder nur langsam reversibel ist. Kann das System schnell wieder in einen erwünschten Zustand zurückkehren, gilt es trotz seiner geringen Resilienz als nachhaltig. Die Resilienz eines Systemzustandes kann damit generell keine notwendige Bedingung für eine nachhaltige Entwicklung darstellen. Darüber hinaus sind Zustände denkbar, bei denen gerade der Übergang in einen anderen Systemzustand eine nachhaltige Entwicklung überhaupt erst möglich macht. Vier Fälle sind demnach bei der Beziehung von Resilienz und nachhaltiger Entwicklung im Sinne der obigen Definition möglich:

1. Resilienz ist notwendig und hinreichend für eine nachhaltige Entwicklung ökologisch-ökonomischer Systeme, wenn es genau einen Gleichgewichtszustand des Systems gibt, der den Erfordernissen einer nachhaltigen Entwicklung entspricht.

2. Resilienz ist hinreichend aber nicht notwendig für eine nachhaltige Entwicklung ökologisch-ökonomischer Systeme, wenn mehrere Gleichgewichtszustände des Systems existieren, die den Erfordernissen einer nachhaltigen Entwicklung entsprechen.

3. Resilienz ist notwendig aber nicht hinreichend für eine nachhaltige Entwicklung ökologisch-ökonomischer Systeme, wenn es keinen Gleichgewichtzustand eines Systems gibt, der den Erfordernissen einer nachhaltigen Entwicklung entspricht.

4. Resilienz ist weder notwendig noch hinreichend für eine nachhaltige Entwicklung ökologisch-ökonomischer Systeme, wenn der Ausgangszustand eines Systems außerhalb der Anforderungen einer nachhaltigen Entwicklung liegt, also der kritische Wert eines Naturkapitalbestandes bereits unterschritten wurde.

[29] Allerdings könnte diese Formulierung auch so interpretiert werden, dass sich die Forderung nach Resilienz nur auf solche Ökosysteme bzw. Ökosystemfunktionen bezieht, die life-support-Funktionen aufweisen, mit anderen Worten auf kritisches Naturkapital bezogen werden. Damit wäre Resilienz nicht per se notwendig, sondern nur im Sinne des Erhalts von kritischem Naturkapital relevant. Dies entspricht meiner vorliegenden Argumentation.

3.4 Resilienz als notwendige Bedingung für die nachhaltiger Entwicklung semi-arider Weideländer

In den vorigen Abschnitten habe ich Resilienz als deskriptives Konzept vorgestellt, während nachhaltige Entwicklung als ein normatives definiert wurde. In der Sprache der Resilienzdebatte lassen sich auf dieser Grundlage im Falle der namibischen Weideländer mindestens zwei Systemzustände unterscheiden: In einem der beiden Zustände herrscht Grasvegetation vor und ist die Haltung von Rindern ökonomisch möglich. Wird dieser Zustand gestört, etwa durch zu hohe Bestockung und das Unterdrücken natürlicher Buschbrände, kann das System zu einem Zustand (Entwicklungspfad) der Verbuschung übergehen. (Denkbar sind natürlich auch Zustände, in denen zum Beispiel eine Wüste entsteht.) Ohne weitere Eingriffe würde sich das Savannensystem in Namibia von einem verbuschten Zustand nicht mehr in einen Zustand mit vorherrschender Grasvegetation zurückentwickeln. Damit wären die ehemals hohen Bestockungsraten als eine Störung anzusehen, denen gegenüber das Weideland nicht resilient war: "It can further be concluded that grazing pressure, even with declining stocking rates, was still inherently too high to utilise the rangelands in a sustainable way and resulted in a form of vicious cycle. Fear has been expressed that the bounds of resilience of the former ecosystem have been exceeded. Only by means of external inputs will the original status of our rangelands be able to be restored" (de Klerk 2004: 22).

Mit Blick auf die Nachhaltigkeitsdebatte stellt sich nun die Frage, ob das System in seinem Gras-dominierten Zustand erhalten bleiben müsste, um den Anforderungen starker Nachhaltigkeit gerecht zu werden. Zentral wäre hier, konkret definieren zu können, inwieweit bei einem Übergang in einen Busch-dominierten Zustand kritisches Naturkapital betroffen ist oder wie in dieses investiert werden kann. Im Sinne des Greifswalder Ansatzes starker Nachhaltigkeit ist also zu überprüfen, welche der genannten vier Möglichkeiten (voriger Abschnitt) bei dem Übergang von einem Gras-dominierten in einen Busch-dominierten Zustand semi-arider Weiden zutrifft.

Die Resilienz, das heißt in diesem Fall der Erhalt der semi-ariden Weidelandschaft, wäre für die nachhaltige Entwicklung in Namibia nicht notwendig, wenn ein alternativer stabiler Zustand existierte, der den Erfordernissen einer nachhaltigen Entwicklung entspricht. Dies scheint denkbar zu sein. Nicht nur bieten die aufkommenden Büsche anderen Tier- und Pflanzenarten ein Habitat, auch können die Büsche als Brennmaterial und zur Stromerzeugung verwendet werden. Da das Abholzen der Büsche sehr zeit- und arbeitsintensiv ist, wird nicht selten mit den positiven Auswirkungen auf die namibische Beschäftigtenquote geworben, zu denen eine stärkere kommerzielle Nutzung der Büsche führen würde. Jedoch sind bislang interessanterweise keine Zahlen bekannt, die das Potential der Nutzung und Ernte der Büsche und deren weitere Wertschöpfung umfassend darstellen, so dass die Verbuschung ausschließlich als Verlust für die Rindfleischindustrie und als gesamtwirtschaftliches Problem wahrgenommen wird. Alternative Nutzungsmöglichkeiten der Büsche sind zwar bekannt, und sie werden im kleinen Maßstab auch umgesetzt, eine grundlegende Umorientierung der Landnutzung ist allerdings nicht zu erkennen.

Unabhängig von dieser Frage muss jedoch angesichts der oben skizzierten politischen Zielsetzungen festgehalten werden, dass es bisher keinen gesellschaftlich akzeptierten alternativen Zustand semi-arider Weideländer gibt. Dies heißt jedoch nicht, dass kein Zustand denkbar ist, der den Vorgaben starker Nachhaltigkeit genügt, langfristig das Nutzenniveau der Gesellschaft mindestens konstant zu halten und kritisches Naturkapital zu bewahren. An dieser Stelle klaffen die normativen Vorgaben einer nachhaltigen Entwicklung und das gesellschaftliche Ideal, das die Rinderhaltung als einzig akzeptable Bewirtschaftungsform vorsieht, offensichtlich auseinander. Die Frage, wie jenseits einer optimierenden Ein-Zustands-Politik agiert werden müsste, ist daher notwendiger Gegenstand weiterer Forschung.

3.5 Schlussfolgerungen

Veränderungen der Savannenökosysteme, vor allem das verstärkte Auftreten von Büschen, werden als ein Hauptproblem für die Bewirtschaftung semi-ariden Weidelands gesehen: Die Verbuschung des Weidelands verringert die Grasproduktivität und die mögliche Bestockungsrate. Außerdem führt eine Reduktion des Grasbestandes zu einer Verstärkung von Bodendegradierung und Erosion. Die Mechanismen der Verbuschung semi-ariden Weidelands sind bis heute nicht abschließend geklärt. Es scheint jedoch klar zu sein, dass zu hohe Bestockungsraten nicht die einzige Ursache darstellen, sondern eine Ursachenkombination vorliegt. Das Unterdrücken von Feuerereignissen und das vollständige Abgrasen der Weiden führen zu seltenen und schwachen Bränden, die die Büsche nicht mehr ausreichend dezimieren. Eine Reduktion der Bestockungsraten hat die Verbuschung der Weideländer bisher nicht aufhalten können, und es scheint, als sei die Resilienz des Gras-dominierten Zustandes überschritten, so dass das Ökosystem nun langfristig in einen Busch-dominierten Zustand übergeht.

Die Resilienz, das heißt in diesem Fall der Erhalt der semi-ariden Weidelandschaft in einem Gras-dominierten Zustand, ist für eine nachhaltige Entwicklung in Namibia nicht notwendig, falls ein alternativer stabiler Zustand existiert, der den Erfordernissen einer nachhaltigen Entwicklung entspricht. Es scheint ein Systemzustand mit vorherrschender Buschvegetation, mit einer eigenen Diversität von Arten und Lebensgemeinschaften und mit einer untergeordneten Weidewirtschaft denkbar zu sein, in dem die Büsche etwa zu

Erzeugung von Strom verwendet werden. Allerdings ist der öffentliche Diskurs in Namibia bislang völlig auf die Nutzung der betreffenden Flächen als Weideland fixiert.

Ob ein Busch-dominierter Zustand ebenfalls Nachhaltigkeitskriterien entspräche und wie er im Vergleich zum Gras-dominierten Zustand zu bewerten wäre, kann beim derzeitigen Stand der Forschung nicht abschließend geklärt werden. Dazu müssen kritische Naturkapitalbestände identifiziert und die Kosten und Nutzen des Übergangs auch für zukünftige Generationen geprüft werden. Fest steht jedoch, dass ein Gras-dominiertes Weideland nicht der einzige stabile Zustand dieser Ökosysteme ist und die alternativen Zustände unvoreingenommen in die Managemententscheidungen einbezogen werden sollten.

Diese Unterscheidungen sind bisher nicht in die Überlegungen zu nachhaltigem Management semi-arider Gebiete eingeflossen. Eine Abwägung der Vor- und Nachteil eines solchen regime shift könnte für die nachhaltige Bewirtschaftung semi-arider Gebiete wegweisend sein.

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Chapter 4

The relationship between resilience and sustainability of ecological-economic systems

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December 2010

This article was published in *Ecological Economics* **70** (2011): 1121–1128.
http://dx.doi.org/10.1016/j.ecolecon.2011.01.003

The relationship between resilience and sustainability of ecological-economic systems

Abstract: Resilience as a descriptive concept gives insight into the dynamic properties of an ecological-economic system. Sustainability as a normative concept captures basic ideas of intergenerational justice when human well-being depends on natural capital and services. Thus, resilience and sustainability are independent concepts. In this paper, we discuss the relationship between resilience and sustainability of ecological-economic systems. We use a simple dynamic model where two natural capital stocks provide ecosystem services that are complements for human well-being, to illustrate different possible cases of the relationship between resilience and sustainability, and to identify the conditions under which each of those will hold: a) resilience of the system is necessary, but not sufficient, for sustainability; b) resilience of the system is sufficient, but not necessary, for sustainability; c) resilience of the system is neither necessary nor sufficient for sustainability; and d) resilience is both necessary and sufficient for sustainability. We conclude that more criteria than just resilience have to be taken into account when designing policies for the sustainable development of ecological-economic systems, and, vice versa, the property of resilience should not be confused with the positive normative connotations of sustainability.

Keywords: ecosystem resilience, dynamics, management of ecological-economic systems, sustainability

JEL-Classification: Q20, Q56, Q57

4.1 Introduction

Speaking about resilience and sustainability is speaking about two highly abstract and multi-various concepts, each of which has a great variety of interpretations and definitions. Here we adopt what seems to be the most general and at the same time most widely accepted definitions of resilience and sustainability[30]. We understand sustainable development as the Brundtland Commission defines it as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987). In this definition, sustainability is a normative concept capturing basic ideas of intra- and intergenerational justice[31]. Concerning obligations towards future generations, the primary question of sustainability then is to what extent do natural capital stocks have to be maintained to enable future generations to meet their needs[32].

In contrast, resilience is a descriptive concept. In a most common definition that goes back to Holling (1973), resilience is thought of as "[...] the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior" (Holling & Gunderson 2002: 4). The underlying idea is that a system may flip from one domain of attraction into another one as a result of exogenous disturbance. If the system will not flip due to exogenous disturbance, the system in its initial state is called resilient. Although Holling-resilience can be quantitatively measured (Holling 1973), we focus on the qualitative classification, where a system in a given state is either resilient, or it is not[33].

In the literature, many connections have been drawn between resilience and sustainability (e.g. Folke et al. 2004, Walker & Salt 2006, Mäler 2008). In some contributions, resilience is seen as a necessary precondition for sustainability. For example, Lebel et al. (2006: 2) point out that "[s]trengthening the capacity of societies to manage resilience is critical to effectively pursuing sustainable development". Similarly, Arrow et al. (1995: 93) conclude

[30] Evidently, as definitions are not universal and are appropriate for a certain objective only (Jax 2002), the relationship between resilience and sustainability depends on the particular definitions of these two terms.

[31] We do not consider the issue of intragenerational justice in this paper, but focus on an operational notion of sustainability that captures intergenerational justice.

[32] The term "natural capital" was established to distinguish services and functions of ecosystems from other capital stocks (Pearce 1988).

[33] An alternative definition of resilience is due to Pimm (1984), who defines resilience as the rate at which a system returns to equilibrium following a disturbance. Resilience, according to Pimm's definition, is not defined for unstable systems. Nevertheless, it is a useful concept for ecological-economic analysis. Martin (2004), for example, suggests a quantitative measure of resilience *sensu* Pimm, namely the costs of the restoration of the system after a disturbance where costs are defined as the "minimal time of crisis", i.e. the minimal time the system is violating pre-specified state-restrictions and, thus, is outside the viability kernel (Béné at al. 2001).

that "economic activities are sustainable only if the life-support ecosystems upon which they depend are resilient", and Perrings (2006: 418) states that "[a] development strategy is not sustainable if it is not resilient".

Some authors explicitly define or implicitly understand the notions of resilience and sustainability such that they are essentially equivalent: "A system may be said to be Holling-sustainable, if and only if it is Holling-resilient" (Common & Perrings 1992: 28), or similarly: "A resilient socio-ecological system is synonymous with a region that is ecologically, economically, and socially sustainable" (Holling & Walker 2003: 1). Levin et al. (1998) claim in general that "[r]esilience is the preferred way to think about sustainability in social as well as natural systems", thus also suggesting an equivalence of resilience and sustainability.

In contrast to this view, it has been noted that "[r]esilience, per se, is not necessarily a good thing. Undesirable system configurations (e.g. Stalin's regime, collapsed fish stocks) can be very resilient, and they can have high adaptive capacity in the sense of re-configuring to retain the same controls on function" (Holling & Walker 2003: 1). In other words, resilience is not sufficient for sustainability, and it can therefore not be taken as an objective of its own.

While systems with multiple stable states are widely discussed, a systematic analysis of the relationship between the concepts of resilience and sustainability in a system with multiple stable states has not yet been conducted. To illustrate this research gap, the statements quoted above do not take into account the following possibilities: if some particular management does not conserve a system's resilience, such that under exogenous disturbance the system may flip from an undesirable state into a desirable one, or from a desirable state into another desirable state, the system management might still achieve sustainable development of the system, even though it is not resilient. As a consequence, one may conclude that resilience is neither desirable in itself nor is it in general a necessary or sufficient condition for sustainable development.

In general, four relationships between resilience and sustainability are logically possible, and any of those may hold in a given system: a) resilience of the system is necessary, but not sufficient, for sustainability; b) resilience of the system is sufficient, but not necessary, for sustainability; c) resilience of the system is neither necessary nor sufficient for sustainability; and d) resilience of the system is both necessary and sufficient for sustainability. In order to clarify and illustrate the different possibilities, and to identify the conditions under which each of those will hold, we use a simple dynamic ecological-economic model where two natural capital stocks provide ecosystem services that are complementary in the satisfaction of human needs.

This model is not meant to represent a real ecological-economic system, or to give a fully general representation of ecological-economic systems. Rather, it is meant to illustrate the complexity of relationships between resilience and sustainability even in a simple dynamic model. In contrast to other models used to study resilience of ecological-economic systems, such as the shallow lake model (e.g. Scheffer 1997, Mäler et al. 2003) or rangeland models (e.g. Perrings & Stern 2000, Anderies et al. 2002, Janssen et al. 2004), it features more than two domains of attraction and the possibility of more than one desirable state. In traditional models of bistable systems only two relationships of resilience and sustainability are possible: (i) the system is resilient in a desired state, such that the system's resilience has to be maintained for sustainability; and (ii) the system is resilient in its current state which is, however, not a desired one, such that resilience prevents sustainability. A situation in which the system is not resilient in a desired state but nevertheless on a path of sustainability cannot – by construction of these traditional models – possibly occur. Our dynamic model, as simple as it is, overcomes this shortcoming and may therefore add another valuable dimension to the model-based study of resilience and sustainability.

The outline of the paper is as follows. In the following section, we present the model (Section 4.2), analyze its basic dynamics (Section 4.3), introduce formal definitions of resilience and sustainability (Section 4.4), and discuss the possible relationships between resilience and sustainability (Section 4.5). In Section 4.6, we discuss our findings and draw conclusions concerning the sustainable management of ecological-economic systems.

4.2 Resilience and sustainability in a simple dynamic model of an ecological-economic system

The model describes the use of two natural capital stocks – say, fish and wood – and features multiple equilibria with different domains of attraction. The deterministic dynamics of the two stocks of fish (x) and wood (w) are described by the following differential equations, referring to the growth of the stocks of fish $\dot{x}$ and wood $\dot{w}$:

$$\dot{x} = f(x) - C = r_x \, (x - v_x) \left(1 - \frac{x}{k_x}\right) - C \tag{1}$$

$$\dot{w} = g(w) - H = r_w \, (w - v_w) \left(1 - \frac{w}{k_w}\right) - H \tag{2}$$

where r_x and r_w denote the intrinsic growth rates, v_x and v_w the minimum viable populations, and k_x and k_w the carrying capacities of the stocks of fish and wood, respectively.

Let $x_0 = x(0)$ and $w_0 = w(0)$ denote the initial state of the system. The differential equations (1) and (2) do not contain ecological interactions, although, of course, in reality ecological interactions may exist. As a consequence, interactions of the stock dynamics are only due to the interrelated harvests of fish and timber. We use C and H to denote the aggregate amounts of harvested fish and timber; $f(\cdot)$ and $g(\cdot)$ describe the intrinsic growth of the two stocks. We assume logistic growth functions for simplicity and because using a well-known functional form of the growth functions helps to clarify the argument.

Suppose that myopic profit-maximizing firms harvest the resources under open-access to ecosystems and sell these ecosystem services as market products to consumers at prices p_x and p_w for fish and timber, respectively. Assuming Schaefer production functions, the amounts of fish and timber harvested from the respective stocks by individual firms are described by

$$C = q_x \, x \, e_x \quad \text{and} \tag{3}$$

$$H = q_w \, w \, e_w \ , \tag{4}$$

where e_x and e_w denote the aggregate effort, measured in units of labor, spent by fish-harvesting-firms and timber-harvesting-firms, respectively, and q_x and q_w denote the productivity of harvesting fish and timber, respectively. Firms can enter and exit the two industries at no costs.

Society consists of n identical utility-maximizing individuals who derive utility from the consumption of manufactured goods (y) as well as from the consumption of the two ecosystem services, fish (c) and timber (h). We assume that all three goods are essential for individual well-being. The utility function of a representative household is

$$u(y,c,h) = y^{1-\alpha}\left[c^{\frac{\sigma-1}{\sigma}} + h^{\frac{\sigma-1}{\sigma}}\right]^{\alpha\frac{\sigma}{\sigma-1}} , \tag{5}$$

where $\alpha \in (0,1)$ is the household's elasticity of marginal utility for consumption of ecosystem services, and σ is the elasticity of substitution between the consumption of fish and timber. Both ecosystem services are assumed to be complements in satisfying human needs, i.e. $\sigma < 1$.

Each household inelastically supplies one unit of labor either to one of the resource-harvesting firms or to the manufactured-goods sector. We assume that labor is the only input in the manufactured-goods sector and that production exhibits constant returns to scale. Assuming that labor supply is large enough, the wage rate is thus determined by the marginal product of labor in the manufacturing sector, which we denote by λ. Taking the manufactured good as numeraire, the household's budget constraint then is

$$\lambda = y + p_x c + p_w h \ . \tag{6}$$

As an exogenous disturbance to this system we consider an unforeseen, one-time random shock $\Delta = (\Delta_x, \Delta_w)$ that affects the state of the system at time t_Δ. The random shock Δ is distributed over the support $\Omega \subseteq [0,\infty) \times [0,\infty)$. The random time of disturbance t_Δ is distributed over $[0,\infty)$. The shock affects the stock variables in a multiplicative way, such that they instantaneously shift from the current state $(x(t_\Delta), w(t_\Delta))$ to another, disturbed state $(x(t_\Delta + dt), w(t_\Delta + dt)) = (\Delta_x \cdot x(t_\Delta), \Delta_w \cdot w(t_\Delta))$ an infinitesimal time increment dt later. Thus, the shock decreases or increases the stocks x and w by a fraction of Δ_x and Δ_w, respectively. The same equations of motion, (1) and (2), govern the dynamic development of the system before and after the disturbance, but due to the one-time and instantaneous shift in the stock variables at time t_Δ the dynamic regime of the system may be altered.

4.3 Basic dynamics

In this model, the deterministic dynamics of the ecological-economic system that is at all times t in a general market equilibrium where profit-maximizing harvesting firms have open access to ecosystems are described by the following system of coupled differential equations (see Appendix):

$$\dot{x} = r_x \, (x - v_x) \left(1 - \frac{x}{k_x}\right) - n \, \alpha \, \frac{(q_x x)^\sigma}{(q_x x)^{\sigma-1} + (q_w w)^{\sigma-1}} \ , \tag{7}$$

$$\dot{w} = r_w \, (w - v_w) \left(1 - \frac{w}{k_w}\right) - n \, \alpha \, \frac{(q_w w)^\sigma}{(q_x x)^{\sigma-1} + (q_w w)^{\sigma-1}} \ . \tag{8}$$

These dynamics are represented by the state-space diagram in Figure 1 for parameter values $r_x = r_w = 0.5$, $k_x = k_w = 1$, $v_x = v_w = 0.08$, $q_x = q_w = 1$, $\lambda = 1$, $\alpha = 0.5$, $\sigma = 0.5$ and $n = 1$. The green line is the isocline for $\dot{w} = 0$, the red line is the isocline for $\dot{x} = 0$. Left (right) of the $\dot{w} = 0$-isocline the dynamics are characterized by $\dot{w} > 0$ (< 0). Likewise, below (above) the $\dot{x} = 0$-isocline the dynamics are characterized by $\dot{x} > 0$ (< 0). In each segment of the state space, the green and red arrows indicate this direction of dynamics. With these dynamics, B is an unstable steady-state equilibrium; D and D$'$ as well as F and F$'$ are locally saddlepoint-stable steady-state equilibria; and A, C, E and E$'$ are locally stable steady-state equilibria[34].

[34] For an explicit analysis of transitional dynamics and sustainability, see Martinet et al. (2007), Bretschger & Pittel (2008).

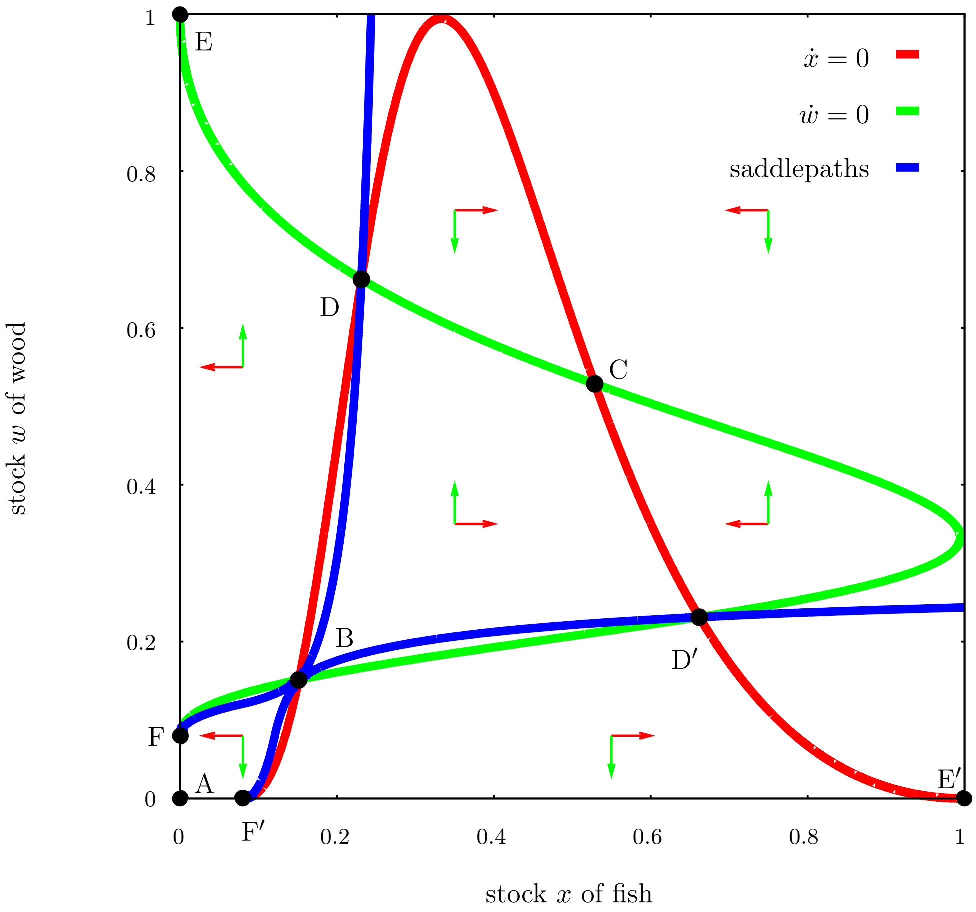

Figure 1: Phase diagram illustrating Basic Dynamics: $\dot{w} > 0$ left of the green line ($\dot{w} < 0$ right of the green line) and $\dot{x} > 0$ below the red line ($\dot{x} < 0$ above the red line). B is an unstable steady-state equilibrium; D, D′, F, F′ are locally saddlepoint-stable steady-state equilibria; A, C, E, E′ are locally stable steady-state equilibria. The corresponding domains of attraction are the area in between the two saddlepaths southwest of B for equilibrium A, the area in between the two saddlepaths northeast of B for equilibrium C, the area northwest of the upper saddlepath for equilibrium E (southeast of the lower saddlepath for E′), the upper (lower) saddlepath northeast of B for equilibrium D (D′), and the upper (lower) saddlepath southwest of B for equilibrium F (F′). Parameter values: $r_x = r_w = 0.5$, $k_x = k_w = 1$, $v_x = v_w = 0.08$, $q_x = q_w = 1$, $\lambda = 1$, $\alpha = 0.4$, $\sigma = 0.5$, $n = 1$.

Let $(x_i^\star, w_i^\star)$ denote the stock levels in steady-state equilibrium $i \in$ {A, B, C, D, D′, E, E′, F, F′}. The *domain of attraction* $\mathcal{A}_i$ *of equilibrium* i is the set of all initial states for which the system converges towards equilibrium i:

$$\mathcal{A}_i = \left\{ (x_0, w_0) \in \mathbb{R}_+ \times \mathbb{R}_+ \middle| \lim_{t \to \infty} (x(t), w(t)) = (x_i^\star, w_i^\star) \right\} \ , \tag{9}$$

where $(x(t), w(t))$ denotes the solution of Eqs. (7) and (8) for initial state (x_0, w_0).

The model features eight domains of attraction: the area in between the two saddlepaths southwest of B for equilibrium A, the area in between the two saddlepaths northeast of B for equilibrium C, the area northwest of the upper saddlepath for equilibrium E (southeast of the lower saddlepath for E′), the upper (lower) saddlepath northeast of B for equilibrium D (D′), and the upper (lower) saddlepath southwest of B for equilibrium F (F′). Each domain of attraction comprises only a limited part of the state space, so that the system may flip from one domain of attraction to another one as a result of exogenous disturbance. Thus, if the system was initially, for example, on the saddlepath converging towards equilibrium D it may be disturbed such that it no longer converges towards equilibrium D, but flips into the domain of attraction of another equilibrium, e.g. C or E.

4.4 Formal definitions of resilience and sustainability

As we are considering a setting under uncertainty where the system may be subject to an exogenous random shock, we shall briefly discuss different meanings of sustainability and resilience with respect to uncertainty, before giving the exact definitions of resilience and sustainability that we will use in the analysis.

To guide today's decision-making in a world of uncertainty, sustainability has to be a meaningful and operational concept *ex ante*, i.e. given today's (incomplete) information. Such an ex-ante concept of sustainability makes an ex-ante assessment of the future consequences of today's actions with respect to some normative sustainability criterion which refers to the *actual future* state of the world and given *today's* information about the uncertain future consequences of today's action.

If, for example, a notion of strong sustainability is adopted, the normative sustainability criterion is that, in the actual future development, critical stocks and services of (natural) capital are maintained above some minimum levels. A corresponding ex-ante concept is then ecological-economic viability (Baumgärtner & Quaas 2009), which demands that, roughly speaking, the critical stocks and services of (natural) capital are maintained above some minimum levels *with some minimum probability.* If, as another example, a notion of weak sustainability is adopted, the normative sustainability criterion is that, in the actual future development, a given level of aggregate wealth or welfare is maintained. A corresponding ex-ante concept is then that *the certainty equivalent of* the next generation's aggregate wealth is not smaller than the current generation's aggregate wealth (Asheim & Brekke 2002).

Ex ante and under uncertainty, the normative sustainability criterion – whatever it may be – cannot be met for sure, but only with a certain probability. Even if some action is found to be sustainable ex-ante, i.e. the action meets the ex-ante concept of sustainability, it may *ex post* turn out to actually not be sustainable with respect to the normative sustainability criterion. While meeting the ex-ante concept of sustainability is, of course, the best that today's actors can do under uncertainty, our interest here is to study the question of what is the role of resilience to exogenous disturbances for actually meeting the normative sustainability criterion in some ecological-economic system. To study this question, we have to consider the normative sustainability criterion rather than an ex-ante concept of sustainability. In this paper, we therefore use the term "sustainability" synonymous to the statement that the normative sustainability criterion has been actually met.

We define the resilience of the ecological-economic system based on Holling (1973) and Carpenter et al. (2001): The ecological-economic system in a given state is resilient to an exogenous disturbance if it does not flip to another domain of attraction.

Definition 1 (Resilience)
The ecological-economic system in state $(x(t_\Delta), w(t_\Delta))$ is called *resilient* to disturbance by an actual shock Δ at time t_Δ if and only if the disturbed system is in the same domain of attraction in which the system has been at the time of disturbance[35]:

$$(x(t_\Delta), w(t_\Delta)) \in \mathcal{A}_i \quad \Rightarrow \quad (x(t_\Delta + dt), w(t_\Delta + dt)) \in \mathcal{A}_i \; . \tag{10}$$

Thus, resilience is defined relative to the system state at the time of disturbance, $(x(t_\Delta), w(t_\Delta))$, and the actual extent of disturbance, Δ[36]. Whether the system is actually resilient (or not) in this sense can therefore only be assessed for sure *ex post*, after the disturbance has actually occurred. This means, the notion of resilience employed here is one of ex-post. In particular, it does not express any *ex-ante* expectation or assessment of resilience to all possibly occurring disturbances.

[35]In slight notational sloppiness, we denote the realization of the random variable by the same symbol as the random variable.

[36]In the following, we will still simply speak of the system as being "resilient" for short, but this implicitly means "resilient in state $(x(t_\Delta), w(t_\Delta))$ to the actual disturbance Δ".

The normative criterion for *sustainability* applied here is that utility shall at no time fall below a specified level, the so-called *sustainability threshold* $\bar{u}$.

Definition 2 (Sustainability)
A consumption path $(y(t), c(t), h(t))$ is called *sustainable* with respect to *sustainability threshold* $\bar{u}$ if and only if utility at no time falls below the specified threshold level $\bar{u}$:

$$u(y(t), c(t), h(t)) \geq \bar{u} \quad \text{for all } t\ . \tag{11}$$

Thus, sustainability is defined relative to a sustainability threshold $\bar{u}$[37]. This captures intergenerational equity in the sense that an equal minimum level of well-being, $\bar{u}$, is sustained for all generations, with ecosystem services and manufactured goods being substitutes in generating well-being. Obviously, the choice of a particular sustainability threshold $\bar{u}$ requires a normative judgment.

As with the property of resilience (Definition 1), whether a consumption path is actually sustainable or not (*sensu* Definition 2) can only be assessed for sure *ex post*, after the disturbance has actually occurred. Therefore, the notion of sustainability (as the notion of resilience) employed here is one of *ex-post*. In particular, it does not express any ex-ante expectation or assessment of sustainability under all possibly occurring disturbances[38].

Although utility $u(y, c, h)$ directly depends only on the manufactured good and the two ecosystem services, and not on the two stocks of natural capital *per se*, the two natural capital stocks are nevertheless important to meet the sustainability criterion (11) because the ecosystem services are to be obtained from these stocks. In other words, sustainability criterion (11) can only be met if the levels of both natural capital stocks, fish and wood, meet a related criterion. A necessary and sufficient condition, in terms of stock levels $x(t)$ and $w(t)$, for sustainability criterion (11) to be met in a system that is at all times t in a general market equilibrium where households maximize their individual utility and profit-maximizing harvesting firms have open access to ecosystems, is that[39]

$$\left[(q_x x(t))^{\sigma-1} + (q_w w(t))^{\sigma-1}\right]^{\frac{1}{\sigma-1}} \geq \frac{\bar{u}^{1/\alpha}}{\alpha\left[(1-\alpha)\lambda\right]^{\frac{1-\alpha}{\alpha}}} \quad \text{for all } t\ . \tag{12}$$

Note that with this condition, the open-access market equilibrium studied here, which is well known to be inefficient, may nevertheless be sustainable.

[37] In the following, we will still simply speak of "sustainability" for short, but this implicitly means "sustainability of consumption path $(y(t), c(t), h(t))$ with respect to sustainability threshold $\bar{u}$".

[38] As we are not interested here in the issue of management or (optimal) control of the ecological-economic system, we do not need to employ ex-ante notions. Ex-ante notions would be needed, though, for studying forward-looking decision-making under uncertainty. We have elaborated elsewhere on an ex-ante criterion of ecological-economic sustainability under uncertainty (Baumgärtner & Quaas 2009).

[39] Proof: Insert Eqs. (21), (22) and (23) in Eqs. (11) and rearrange.

We call the set of all $x(t)$ and $w(t)$ that fulfill condition (12) for a given sustainability threshold $\bar{u}$ the *sustainability set* $\mathcal{S}_{\bar{u}}$. Figure 2 illustrates the sustainability set in state space for different values of the sustainability threshold $\bar{u}$ with parameter values $q_x = q_w = 1$, $\lambda = 1$, $\alpha = 0.5$, $\sigma = 0.5$. For each threshold level $\bar{u}$, the sustainability set $\mathcal{S}_{\bar{u}}$ comprises all dynamic paths $x(t)$ and $w(t)$ that are contained for all times in the area northeast of the magenta-colored curve.

4.5 Possible relationships between resilience and sustainability

In the following, we show that in the model described above different relationships between resilience and sustainability may hold, depending on the initial state (x_0, w_0) of the ecological-economic system, the sustainability threshold level $\bar{u}$, and the disturbance Δ at t_Δ to which the system is (or is not) resilient. Each case is illustrated by an example. For that sake, we consider the parameterization of the model that also underlies Figure 1 ($r_x = r_w = 0.5$, $k_x = k_w = 1$, $v_x = v_w = 0.08$, $q_x = q_w = 1$, $\lambda = 1$, $\alpha = 0.4$, $\sigma = 0.5$, $n = 1$). This figure thus depicts the different domains of attraction to which our argument refers. To start with, assume that the support of the random disturbance Δ is $\Omega = [0, \infty) \times [0, \infty)$, that is, the actual disturbance may either increase or decrease the two state variables x and w, and the states can potentially assume any non-negative values after the disturbance.

Proposition 1
There exists an initial state (x_0, w_0) of the system and a sustainability threshold level $\bar{u}$, so that under any actual disturbance Δ at any time t_Δ *resilience is necessary for sustainability.* In particular, this is the case for all $(x_0, w_0) \in \mathcal{S}_{\bar{u}}$ if there exists an equilibrium i with $\mathcal{S}_{\bar{u}} \subseteq \mathcal{A}_i$.

The proposition is proven by giving an example for (x_0, w_0) and $\bar{u}$, so that under any actual disturbance Δ at any time t_Δ resilience is necessary for sustainability. The example is illustrated by Figure 3. Consider the initial state ($x_0 = 0.7$, $w_0 = 0.5$) (marked by a cyan-colored dot) and sustainability threshold level $\bar{u} = 0.22$ (marked by a magenta-colored curve). Suppose that sustainability holds, that is, the entire dynamic path $(x(t), w(t))$ remains within the sustainability set $\mathcal{S}_{\bar{u}}$ before and after the disturbance. Then, as the sustainability set is a subset of the domain of attraction $\mathcal{A}_\mathrm{C}$ of equilibrium C, the entire dynamic path $(x(t), w(t))$ necessarily remains within this domain before and after the disturbance. This means, the system is necessarily resilient to the disturbance. Thus, resilience is necessary for sustainability. Evidently, this is true independently of the particular disturbance Δ or time of disturbance t_Δ. The crucial property here is that the

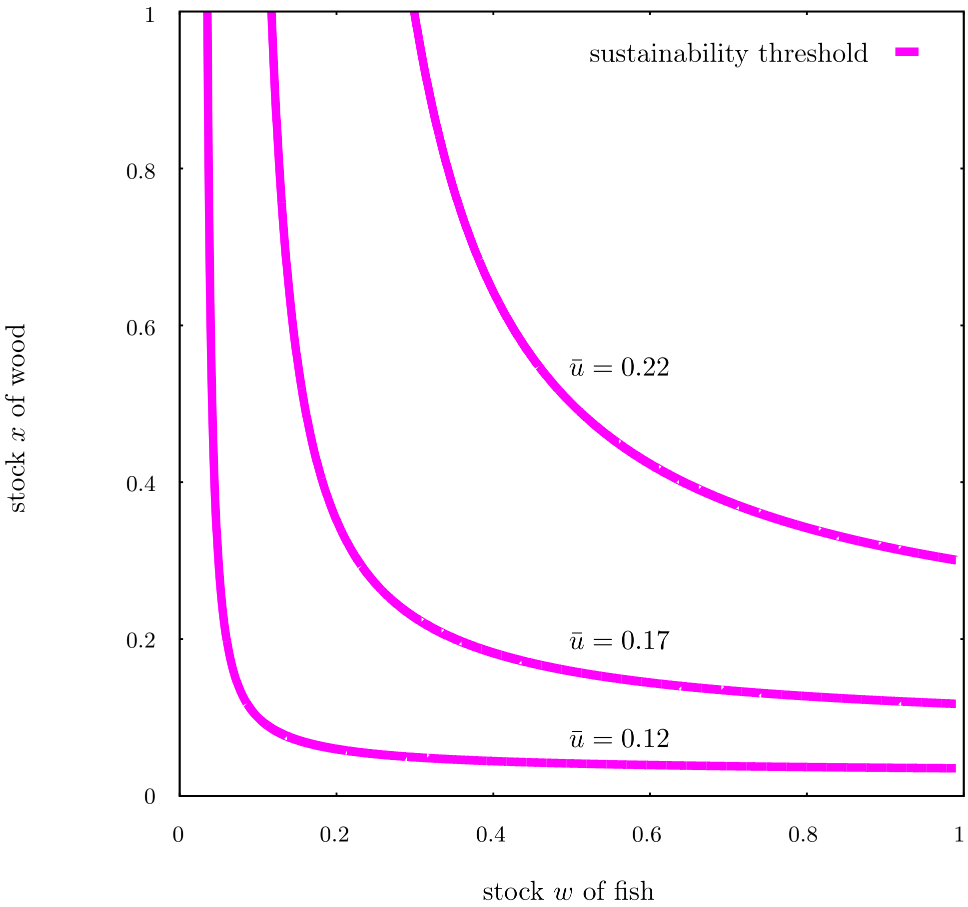

Figure 2: Sustainability set in state space: For $\bar{u} = 0.12$, 0.17 and 0.22. For each threshold level $\bar{u}$, the sustainability set $\mathcal{S}_{\bar{u}}$ comprises all dynamic paths $x(t)$ and $w(t)$ that are contained for all times in the area northeast of the magenta-colored curve. Parameter values: $q_x = q_w = 1$, $\lambda = 1$, $\alpha = 0.5$, $\sigma = 0.5$.

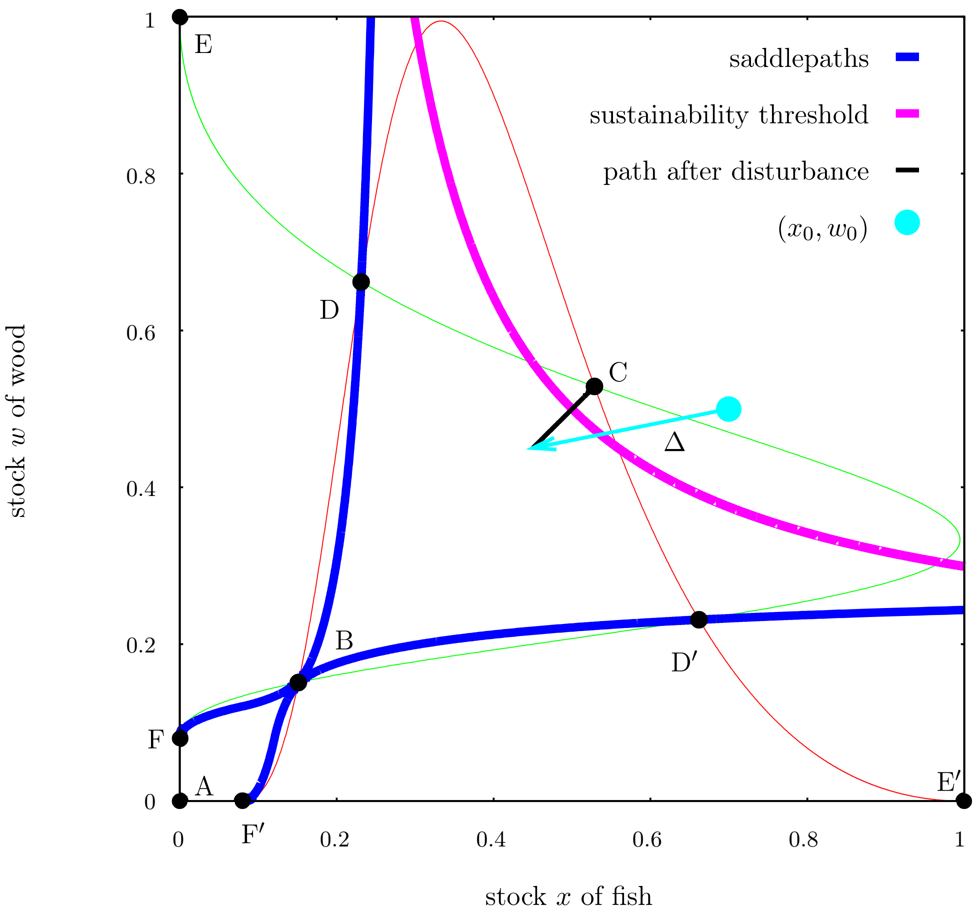

Figure 3: Phase diagram illustrating Propositions 1 and 4: Resilience of the system in state $(x_0 = 0.7, w_0 = 0.5)$ to any disturbance at any time is necessary for sustainability at threshold level $\bar{u} = 0.22$. Resilience of the system in state $(x_0 = 0.7, w_0 = 0.5)$ to disturbance $\Delta = (0.643, 0.9)$ at time $t_\Delta = 0$ is not sufficient for sustainability at threshold level $\bar{u} = 0.22$. Other parameter values as in Figure 1.

sustainability set $\mathcal{S}_{\bar{u}}$ is fully contained within the domain of attraction $\mathcal{A}_\mathrm{C}$ of equilibrium C[40].

Proposition 2
There exists an initial state (x_0, w_0) of the system, an actual disturbance Δ at time t_Δ, and a sustainability threshold level $\bar{u}$, so that *resilience is not necessary for sustainability.*

The proposition is proven by giving an example for (x_0, w_0), Δ, t_Δ and $\bar{u}$, so that resilience is not necessary for sustainability. The example is illustrated by Figure 4. In this example, the domains of attraction $\mathcal{A}_\mathrm{C}$ and $\mathcal{A}_\mathrm{D}$ of equilibria C and D are all fully contained in the sustainability set $\mathcal{S}_{\bar{u}}$ for threshold level $\bar{u} = 0.12$ (marked by a magenta-colored curve). The initial state of the system, $(x_0 = 0.212,\ w_0 = 0.401)$ (marked by a cyan-colored dot), is assumed to be in equilibrium D's domain of attraction $\mathcal{A}_\mathrm{D}$. Without disturbance, the system would therefore remain within the sustainability set $\mathcal{S}_{\bar{u}}$ for all times. Yet, sustainability is not compromised if the system flips into equilibrium C's domain of attraction $\mathcal{A}_\mathrm{C}$ as a consequence of disturbance $\Delta = (1.552,\ 1)$ at time $t_\Delta = 0$, because this domain of attraction is also fully contained in the sustainability set $\mathcal{S}_{\bar{u}}$. That is, resilience is not necessary for sustainability.

Proposition 3
There exists an initial state (x_0, w_0) of the system and a sustainability threshold level $\bar{u}$, so that under any actual disturbance Δ at any time t_Δ *resilience is sufficient for sustainability.* In particular, this is the case for all $(x_0, w_0) \in \mathcal{A}_i$ if there exists an equilibrium i with $\mathcal{A}_i \subseteq \mathcal{S}_{\bar{u}}$.

The proposition is proven by giving an example for (x_0, w_0) and $\bar{u}$, so that under any actual disturbance Δ at any time t_Δ resilience is sufficient for sustainability. The example is illustrated, again, by Figure 4. The initial state of the system, $(x_0 = 0.212,\ w_0 = 0.401)$ (marked by a cyan-colored dot), is assumed to be in equilibrium D's domain of attraction $\mathcal{A}_\mathrm{D}$, which is fully contained in the sustainability set $\mathcal{S}_{\bar{u}}$ for sustainability threshold level $\bar{u} = 0.12$ (marked by a magenta-colored curve). Suppose that resilience holds, that is, the entire dynamic path $(x(t), w(t))$ remains within equilibrium D's domain of attraction $\mathcal{A}_\mathrm{D}$ also after the disturbance. Then, as this domain is a subset of the sustainability set $\mathcal{S}_{\bar{u}}$, the entire dynamic path $(x(t), w(t))$ necessarily remains within the sustainability set. Thus, resilience is sufficient for sustainability. Evidently, this is true independently of the particular disturbance Δ or time of disturbance t_Δ. The crucial property here is that

[40] The proof generalizes as follows. Suppose that $(x(t), w(t)) \in \mathcal{S}_{\bar{u}}$ for all $t \geq 0$ (sustainability). If $\mathcal{S}_{\bar{u}} \subseteq \mathcal{A}_i$, as long as $(x(t), w(t)) \in \mathcal{S}_{\bar{u}}$ it also holds that $(x(t), w(t)) \in \mathcal{A}_i$. Thus, $(x(t), w(t)) \in \mathcal{A}_i$ for all $t \geq 0$ (resilience).

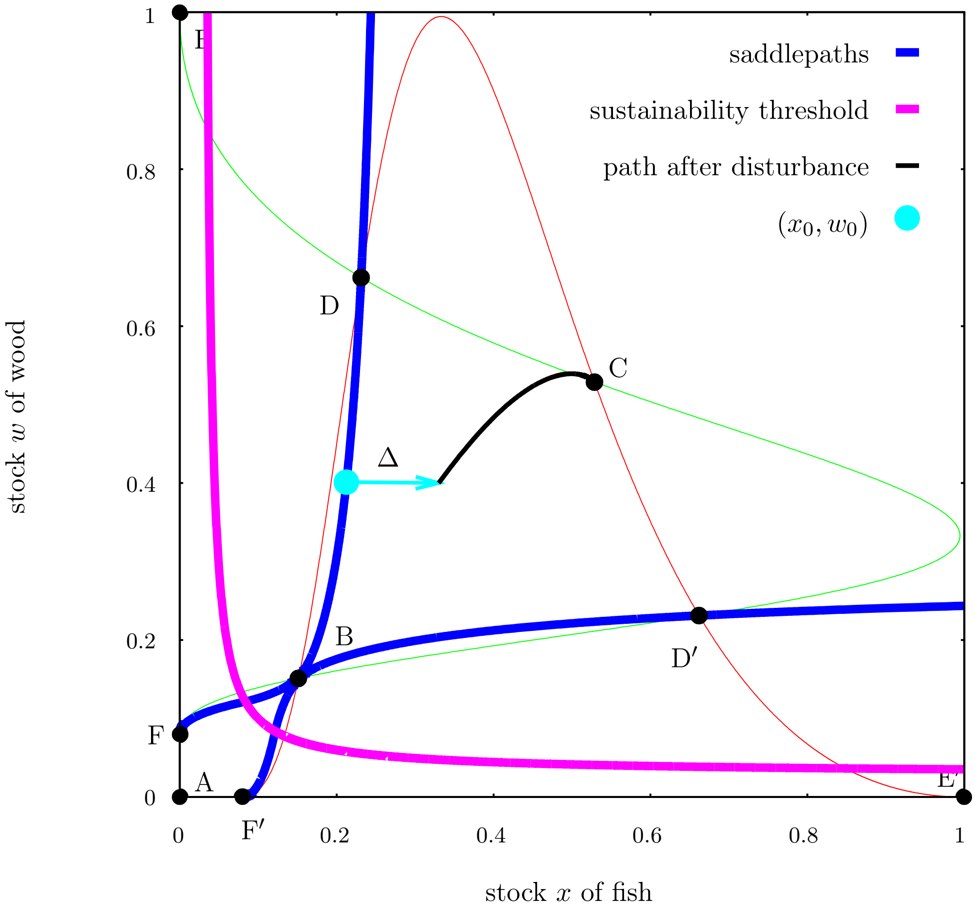

Figure 4: Phase diagram illustrating Propositions 2 and 3: Resilience of the system in state $(x_0 = 0.212,\ w_0 = 0.401)$ to disturbance $\Delta = (1.552,\ 1)$ at time $t_\Delta = 0$ is not necessary for sustainability at threshold level $\bar{u} = 0.12$. Resilience of the system in state $(x_0 = 0.212,\ w_0 = 0.401)$ to any disturbance at any time is sufficient for sustainability at threshold level $\bar{u} = 0.12$. Other parameter values as in Figure 1.

equilibrium D's domain of attraction $\mathcal{A}_\text{D}$ is fully contained within the sustainability set $\mathcal{S}_{\bar{u}}$[41].

Proposition 4
There exists an initial state (x_0, w_0) of the system, an actual disturbance Δ at time t_Δ, and a sustainability threshold level $\bar{u}$, so that *resilience is not sufficient for sustainability.*

The proposition is proven by giving an example for (x_0, w_0), Δ, t_Δ and $\bar{u}$, so that resilience is not sufficient for sustainability. The example is illustrated, again, by Figure 3. Consider the initial state of the system, $(x_0 = 0.7,\ w_0 = 0.5)$ (marked by a cyan-colored dot), which is in equilibrium C's domain of attraction $\mathcal{A}_\text{C}$ and also in the sustainability set $\mathcal{S}_{\bar{u}}$ for sustainability threshold level $\bar{u} = 0.22$ (marked by a magenta-colored curve). The system is resilient to disturbance $\Delta = (0.643,\ 0.9)$ at time $t_\Delta = 0$, as the system is still in equilibrium C's domain of attraction after this disturbance. Yet, the disturbed system is no longer in the sustainability set, and will, at least for some period of time, remain outside the sustainability set. Thus, resilience is not sufficient for sustainability.

With these four elementary propositions, combined statements about the relationship between resilience and sustainability are possible. Figure 3 provides an example where *resilience is necessary, but not sufficient, for sustainability.* Figure 4 provides an example where *resilience is sufficient, but not necessary, for sustainability.*

Propositions 2 and 4 may hold for the same initial state and sustainability threshold, but not the same actual disturbance. Loosely speaking, this means that *resilience is neither necessary nor sufficient for sustainability.* Strictly speaking, this means that there exists an initial state (x_0, w_0) of the system and a sustainability threshold $\bar{u}$ such that the following holds: there is an actual disturbance Δ at any time t_Δ such that resilience is not necessary for sustainability, and there is another actual disturbance Δ' at the same or another time t'_Δ such that resilience is not sufficient for sustainability. Figure 5 provides an example. In this example, the sustainability set $\mathcal{S}_{\bar{u}}$ for sustainability threshold level $\bar{u} = 0.17$ (marked by a magenta-colored curve) partially contains, inter alia, the domains of attraction $\mathcal{A}_\text{C}$ and $\mathcal{A}_{\text{E}'}$ of equilibria C and E′. The initial state of the system, $(x_0 = 0.85,\ w_0 = 0.18)$ (marked by a cyan-colored dot), is in that part of equilibrium E′'s domain of attraction $\mathcal{A}_{\text{E}'}$ that is also part of the sustainability set $\mathcal{S}_{\bar{u}}$. Disturbance $\Delta = (0.882,\ 1.833)$ at time $t_\Delta = 0$ would flip the system into the domain of attraction of equilibrium C, i.e. the system is not resilient against such disturbance. Yet, the system remains in the sustainability set

[41] The proof generalizes as follows. Suppose the system in domain $\mathcal{A}_i$ is resilient to disturbance Δ (resilience). Then, if $(x_0, w_0) \in \mathcal{A}_i$ it also holds that $(x(t), w(t)) \in \mathcal{A}_i$ for all $t \geq 0$. If $\mathcal{A}_i \subseteq \mathcal{S}_{\bar{u}}$, it follows that $(x(t), w(t)) \in \mathcal{S}_{\bar{u}}$ for all $t \geq 0$ (sustainability).

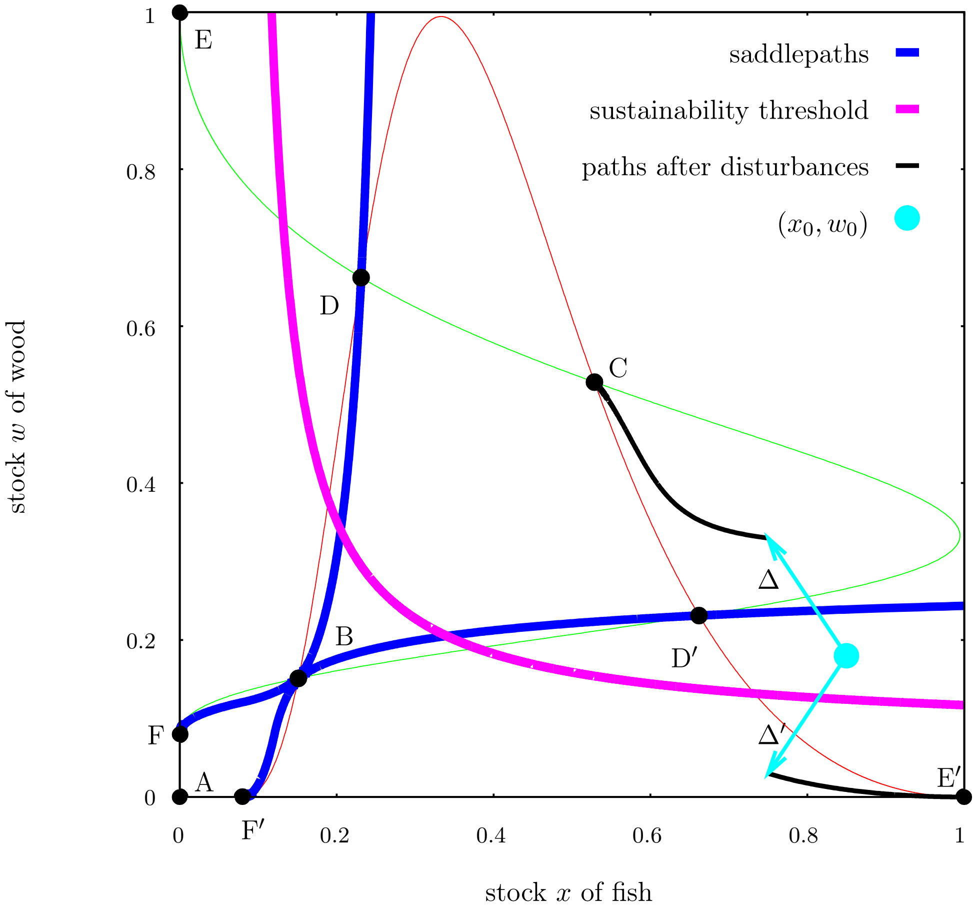

Figure 5: Phase diagram illustrating resilience is neither necessary nor sufficient for sustainability: Resilience of the system in state ($x_0 = 0.85$, $w_0 = 0.18$) to disturbance $\Delta = (0.882, 1.833)$ at time $t_\Delta = 0$ is not necessary for sustainability at threshold level $\bar{u} = 0.17$; resilience to disturbance $\Delta' = (0.882, 0.167)$ at time $t'_\Delta = 0$ is not sufficient for sustainability. Other parameter values as in Figure 1.

$\mathcal{S}_{\bar{u}}$ for all times. That is, resilience of the system in state (x_0, w_0) to disturbance Δ at time t_Δ is not necessary for sustainability at threshold level $\bar{u}$. Now, consider a disturbance $\Delta' = (0.882,\, 0.167)$ at time $t'_\Delta = 0$ which would leave the system in the same domain of attraction $\mathcal{A}_{\mathrm{E}'}$ but move it out of the sustainability domain $\mathcal{S}_{\bar{u}}$. That is, resilience of the system in state (x_0, w_0) to disturbance Δ' at time t'_Δ is not sufficient for sustainability at threshold level $\bar{u}$.

As long as the support of the random disturbance Δ is unbounded, *resilience is necessary and sufficient for sustainability* if and only if the domain of attraction, in which the system initially is, and the sustainability set coincide. In the model studied here, this possibility does not exist, and it is very unlikely to exist in any model. However, resilience may be necessary and sufficient for sustainability if the support of the random disturbance Δ is small enough, that is, if only small disturbances may occur. Formally, there exists an initial state (x_0, w_0) of the system, a sustainability threshold level $\bar{u}$, and a support $\Omega \subset [0, \infty) \times [0, \infty)$ of the random disturbance, so that under any actual disturbance $\Delta \in \Omega$ at any time t_Δ *resilience is necessary and sufficient for sustainability.*

This is proven by giving an example, which is illustrated by Figure 6. The initial state of the system, $(x_0 = 0.528,\, w_0 = 0.528)$ (marked by a cyan-colored dot), is assumed to be in equilibrium C, so that the system remains in this equilibrium until the time of disturbance t_Δ. With a support $\Omega = [0.621, 1.379] \times [0.621, 1.379]$ of the random disturbance, whatever disturbance $\Delta \in \Omega$ actually occurs at time t_Δ will not move the system outside of the cyan-colored rectangle. Given the system dynamics (depicted in Figure 1), the system will return to equilibrium C along a dynamic path that is entirely within both the sustainability set $\mathcal{S}_{\bar{u}}$ for sustainability threshold level $\bar{u} = 0.17$ and equilibrium C's domain of attraction $\mathcal{A}_{\mathrm{C}}$, because the cyan-colored rectangle is fully contained within both the sustainability set $\mathcal{S}_{\bar{u}}$ and equilibrium C's domain of attraction $\mathcal{A}_{\mathrm{C}}$. Thus, both sustainability and resilience hold, so that resilience is necessary and sufficient for sustainability.

4.6 Discussion and conclusion

In this paper, we have studied how, in general, resilience is related to sustainability in the development of an ecological-economic system. Resilience is in the first place a purely descriptive concept of system dynamics. In contrast, sustainability is a normative concept capturing basic ideas of intergenerational justice when human well-being depends on natural capital and services. Thus, resilience and sustainability are independent concepts characterizing the dynamics of ecological-economics systems.

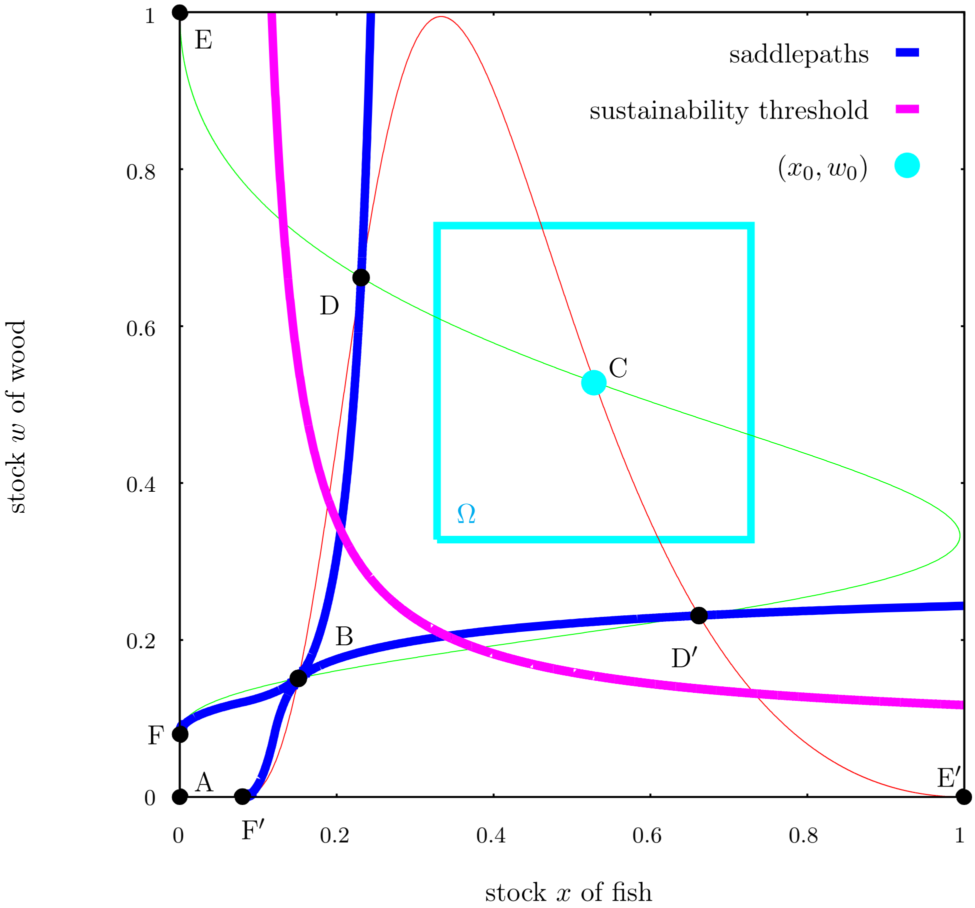

Figure 6: Phase diagram illustrating resilience is necessary and sufficient for sustainability: Resilience of the system in state ($x_0 = 0.528$, $w_0 = 0.528$) to any disturbance Δ from the support $\Omega = [0.621, 1.379] \times [0.621, 1.379]$ at any time t_Δ is necessary and sufficient for sustainability at threshold level $\bar{u} = 0.17$. Other parameter values as in Figure 1.

We have distinguished and specified four possible relationships between resilience and sustainability: a) resilience of the system is necessary, but not sufficient, for sustainability; b) resilience of the system is sufficient, but not necessary, for sustainability; c) resilience of the system is neither necessary nor sufficient for sustainability; and d) resilience of the system is both necessary and sufficient for sustainability. All of those are logically possible, and any may hold in the simple dynamic ecological-economic model that we have presented here, depending on the initial state of the system, the normative sustainability threshold, and the uncertain disturbance to the system.

The result that there are four potential relationships between resilience and sustainability has a much broader validity and generally holds for all systems with more than two domains of attraction. Generalizing from our particular model, we conjecture that the following holds. If the sustainability set is a subset of the domain of attraction and the system initially is in the sustainability set, resilience of the system is *necessary* for sustainability. Resilience is *sufficient* for sustainability, on the other hand, if the entire domain of attraction in which the system initially is, is contained in the sustainability set. Finally, resilience and sustainability *are equivalent* if the domain of attraction in which the system initially is coincides with the sustainability set.

All taken together, in general the deduction from sustainability to resilience, or vice versa, is not possible. This has implications for the sustainable management of ecological-economic systems. In particular, the property of resilience should not be confused with the positive normative connotations of sustainability, and, vice versa, more criteria than just resilience have to be taken into account when designing policies for the sustainable development of ecological-economic systems. Rather, for the sustainable management of an ecological-economic system it is decisive to know (1) the current state of the system, (2) the domains of attraction of the system, (3) the sustainability norm and the associated sustainability set in state space, and (4) the potential extent of disturbance.

Acknowledgments

We are grateful to two anonymous reviewers for critical discussion and to the German Federal Ministry of Education and Research for financial support under grant 01UN0607.

Appendix

With the harvesting functions (3) and (4), assuming that the unit costs of harvesting effort are simply given by the wage rate, and given that the wage rate equals the marginal product λ of labor in the manufactured-goods sector, the aggregate profits of firms harvesting fish and timber are given by:

$$\pi_x = p_x\,C - \lambda e_x = (p_x q_x x - \lambda) e_x \ , \tag{13}$$

$$\pi_w = p_w\,H - \lambda e_w = (p_w q_w w - \lambda) e_w \ . \tag{14}$$

If firms can freely enter and exit the industry, an open-access equilibrium is characterized by complete dissipation of resource rents, i.e. zero profits for all firms:

$$\pi_x = 0 \quad \text{and} \quad \pi_w = 0 \ . \tag{15}$$

With Eqs. (13) and (14), this condition implies that equilibrium market prices p_x for fish and p_w for timber are related to resource stocks x and w as follows:

$$p_x = \frac{\lambda}{q_x} x^{-1} \ , \tag{16}$$

$$p_w = \frac{\lambda}{q_w} w^{-1} \ . \tag{17}$$

The Marshallian demand functions of a representative household can be obtained from solving the utility maximization problem

$$\max_{y,c,h} u(y,c,h) \quad \text{subject to} \quad (6) \ , \tag{18}$$

where $u(y,c,h)$ is the utility function (5), and (6) is the budget constraint. The first-order conditions lead to the Marshallian demand functions for fish and timber,

$$c(p_x, p_w, \lambda) = \alpha\lambda \frac{p_x^{-\sigma}}{p_x^{1-\sigma} + p_w^{1-\sigma}} \ , \tag{19}$$

$$h(p_x, p_w, \lambda) = \alpha\lambda \frac{p_w^{-\sigma}}{p_x^{1-\sigma} + p_w^{1-\sigma}} \ , \tag{20}$$

as well as to the demand for the manufactured good,

$$y(p_x, p_w, \lambda) = (1-\alpha)\lambda \ . \tag{21}$$

With labor income λ and the open-access equilibrium prices of the two resources, Eqs. (16) and (17), this gives consumption of an individual household as a function of the two resource stocks:

$$c(x,w) = \alpha \frac{(q_x x)^{\sigma}}{(q_x x)^{\sigma-1} + (q_w w)^{\sigma-1}} \ , \tag{22}$$

$$h(x,w) = \alpha \frac{(q_w w)^{\sigma}}{(q_x x)^{\sigma-1} + (q_w w)^{\sigma-1}} \ . \tag{23}$$

Using the market-clearing conditions $C = nc$ and $H = nh$ in the equations of motion (1 and 2) for the two resource stocks, these become

$$\dot{x} = f(x) - n\,c(x, w) \; , \tag{24}$$

$$\dot{w} = g(w) - n\,h(x, w) \; . \tag{25}$$

Inserting $c(x, w)$ and $h(x, w)$ from Equations (22) and (23), and $f(x)$ and $g(w)$ from Equations (1) and (2), yields Equations (7) and (8).

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Chapter 5

Consumer Preferences Determine Resilience of Ecological-Economic Systems

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September 2011

This article was published in *Ecology and Society* (2011) **16**(4), 9.
URL: http://www.ecologyandsociety.org/vol16/iss4/art9/.
http://dx.doi.org/10.5751/ES-04392-160409

Consumer preferences determine resilience of ecological-economic systems

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 firms, 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.

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

JEL-Classification: Q01, Q20, Q57

5.1 Introduction

Natural systems that are used and managed by humans for the ecosystem services they provide may exhibit nontrivial 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, semiarid rangelands, wetlands, shallow lakes, coral reefs, or high-seas fisheries (Gunderson & 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 that exist prior to any human interference with the system, such as ecological properties of shallow lakes or the interaction between grass and shrub species in semiarid 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 & 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, i.e., "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 because of 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 multispecies 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.

5.2 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

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

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 $\sigma \rightarrow \infty$, 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

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

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

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., Appendix A4 in Scheffer 2009):

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

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

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

$$c^{prod} = v_x \, x \, e_x, \tag{6}$$

$$h^{prod} = v_w \, w \, e_w, \tag{7}$$

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

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

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

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

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

5.3 Analysis

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.

5.3.1 Natural dynamics

In the absence of any resource harvesting by society, the system's dynamics are completely determined by the natural dynamics of the two resources stocks of fish and wood, described by Equations (2) to (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 interspecies competition ($\gamma_x = \gamma_w = 0$), both stocks converge to their respective carrying capacities. The isoclines $dx/dt = 0$ and $\dot{\omega} = 0$ thus are the straight lines with $w = \kappa_w$ and $x = \kappa_x$, respectively. This dynamics is represented by the upper phase diagram in Figure 1 for parameter values $\rho_x = \rho_w = 0.5$ and $\kappa_x = \kappa_w = 1$. 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 are characterized by $dx/dt > 0 (< 0)$. Likewise, left

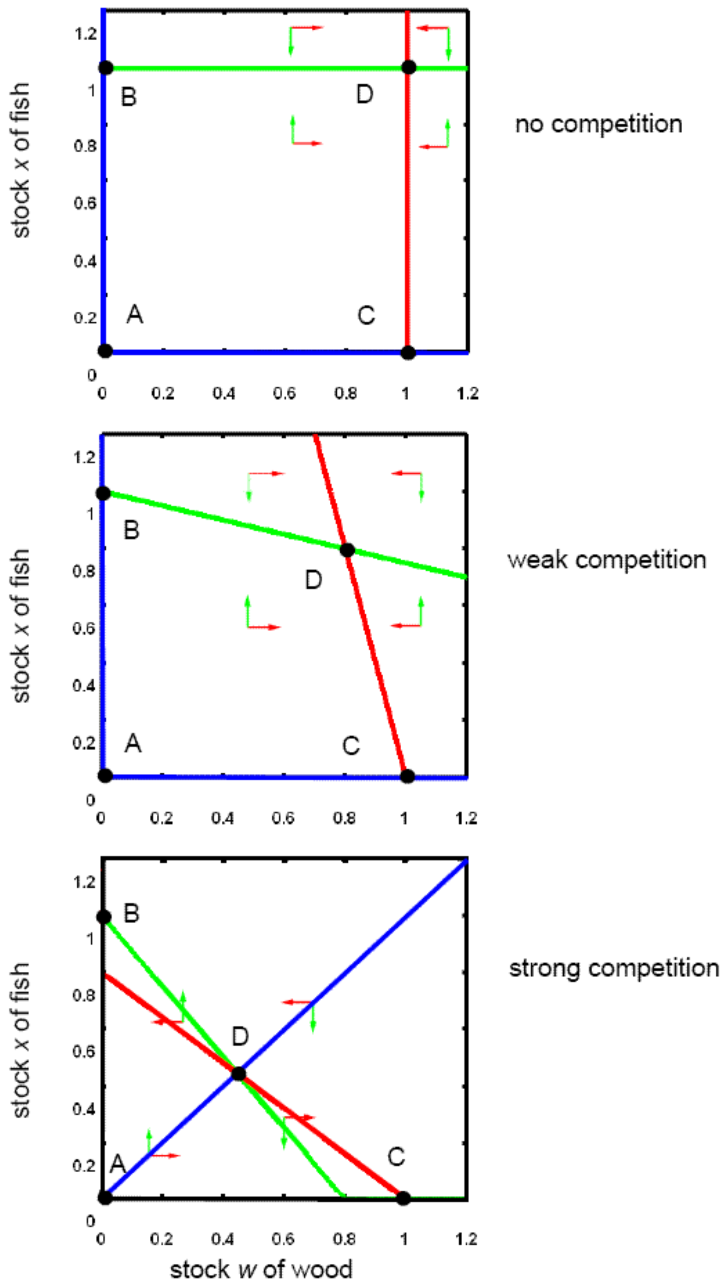

Figure 1: Phase diagrams in state space for the ecosystem's natural dynamics without any harvesting ($C = H = 0$). Dynamics is characterized by $dx/dt > 0 (< 0)$ below (above) the green line, and $dw/dt > 0 (< 0)$ left (right) of the red line. Blue lines indicate saddlepaths. The upper diagram displays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

(right) of the $dw/dt = 0$-isocline the dynamics are 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.

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 interspecies competition ($\gamma_x = \gamma_w = 0$), it is obvious from the state-space representation (Fig. 1, upper diagram) that A is an unstable equilibrium, whereas 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 ($x = 0, w \geq 0$) and ($x \geq 0, w = 0$). From any system state in this domain the system will automatically converge towards equilibrium D. Therefore, 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 interspecies competition, neither stock reaches its full carrying capacity because of competition from the other species (Fig. 1, middle and lower diagrams). As long as interspecies competition is weaker than intra-species competition ($\gamma_i < 1$), however, the ecosystem still exhibits one (almost) globally stable equilibrium at point D (Fig. 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 interspecies competition is stronger than intra-species competition ($\gamma_i > 1$, Fig. 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. Because of an exogenous disturbance, the system may flip from one basin of attraction to another. This means, ecological interaction in the form of strong interspecies competition has a destabilizing effect on the ecosystem.

5.3.2 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, 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 interspecies competition. In the next section, we then systematically study variations in these two parameters: complementarity and interspecies 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 interspecies competition (cf. Fig. 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 $\rho_x = \rho = 0.5, \kappa_x = \kappa_w = 1, \gamma_x = \gamma_w = 0, v_x = v_w = 1, \alpha = 0.6, \sigma = 0.4$ and $n = 1$.

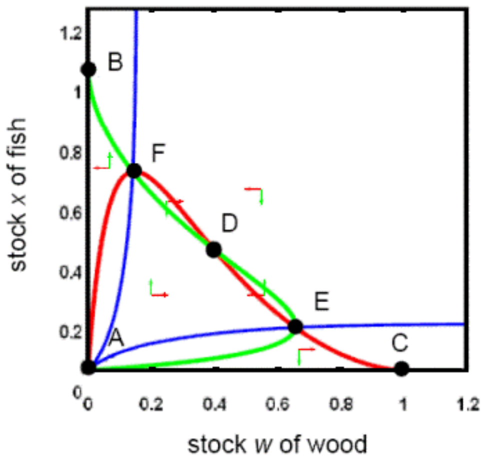

Figure 2: Phase diagram for the ecosystem's dynamics under open access and profit-maximizing harvesting. Dynamics are characterized by $dx/dt > 0 (< 0)$ left (right) of the green line, and $dw/dt > 0 (< 0)$ below (above) the red line. A is an unstable equilibrium; E and F are locally saddlepoint-stable 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 are characterized by $dx/dt > 0 (< 0)$. Likewise, below (above) the $dw/dt = 0$-isocline the dynamics are 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, that 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, i.e., open access to ecosystems of profit-maximizing harvesting firms, has fundamentally altered the resilience properties of the ecosystem. Although in the absence of resource harvesting and not too-strong interspecies 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 that 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.

5.3.3 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, i.e., 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 $\sigma = 0.4$, which reflects a mild complementarity (cf. Fig. 2). Figure 3 illustrates the resilience properties of the ecosystem when, everything else being equal, the elasticity of substitution changes to $\sigma = 0.95$ (low complementarity) and $\sigma = 0.05$ (high complementarity).

From Figure 3 (left diagram) it is apparent that even for open access and profit-maximizing

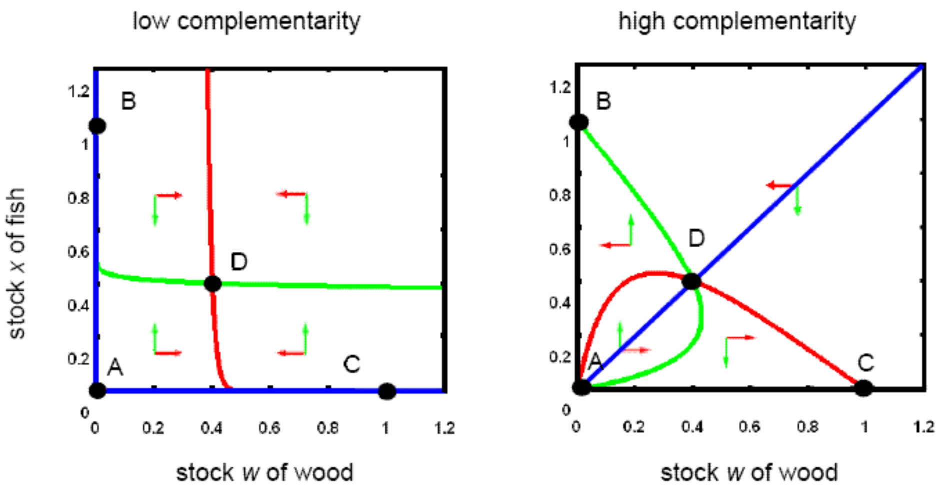

Figure 3: Phase diagrams for the ecosystem's dynamics under open access and profit-maximizing harvesting for low complementarity ($\sigma = 0.95$, left diagram) and high complementarity ($\sigma = 0.05$, right diagram) between ecosystem services in consumption. Dynamics are characterized by $dx/dt > 0(< 0)$ 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 ($x = 0, w \geq 0$) and ($x \geq 0, w = 0$). 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: $\rho_x = \rho_w = 0.5, \kappa_x = \kappa_w = 1, \gamma_x = \gamma_w = 0, v_x = v_w = 1, \alpha = 0.6, n = 1$.

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 interspecies 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. Fig. 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. Because 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-reenforcing dynamics. At a certain threshold value of σ ($\sigma = \frac{1}{3}$ 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 (Fig. 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 (Fig. 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 (Fig. 4, right diagram).

In passing we note that increasing the productivity of the harvest technology, v_x and v_w,

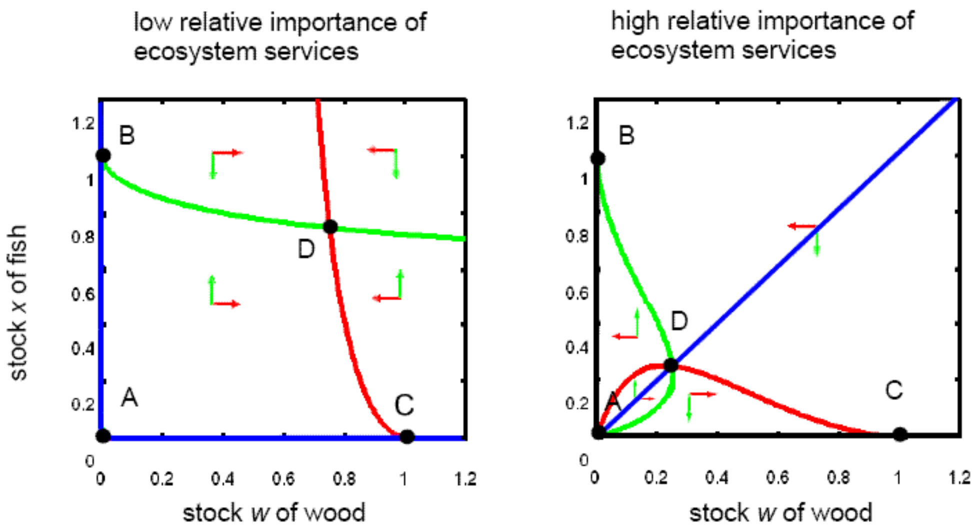

Figure 4: Phase diagrams for the ecosystem's dynamics under open access and profit-maximizing harvesting for different levels of relative importance of ecosystem services, a. Dynamics are characterized by $dx/dt > 0(< 0)$ left (right) of the green line, and $dw/dt > 0(< 0)$ below (above) the red line.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

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, although 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 interspecies competition. This is shown in the remainder of the section.

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

In all three phase diagrams of Figure 5, equilibrium A, where both species are extinct, is unstable. In the case of low complementarity ($\sigma = 0.95$; Fig. 5, upper diagram), 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 coexistence of both species is likely. At a certain threshold value of σ (about $\sigma = 0.62$ 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 ($\sigma = 0.4$, middle and $\sigma = 0.05$; Fig. 5, lower diagram). Note that compared to Figures 2 to 4, the threshold value of σ in Figure 5 is higher (i.e., threshold-complementarity is lower) because of the additional destabilizing effect of species competition.

The destabilizing effect of increasing interspecies competition also occurs under resource harvesting. This is shown in Figure 6 for a given level of resource complementarity. Without interspecies competition ($\gamma_x = \gamma_w = 0$; Fig. 6, upper diagram), the interior equilibrium D with both resource species in existence is locally stable, but exhibits limited resilience because of open-access resource harvesting. The resilience of this interior equilibrium sharply decreases with the introduction of species competition ($\gamma_x = \gamma_w = 0.25$; Fig. 6, middle diagram): 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 interspecies competition ($\gamma_x = \gamma_w = 1.25$; Fig. 6, lower diagram) entails lower abundances of both species at the saddlepoint-equilibrium D.

Comparing Figure 6 to Figure 1 shows that the effects on resilience of increasing interspecies 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

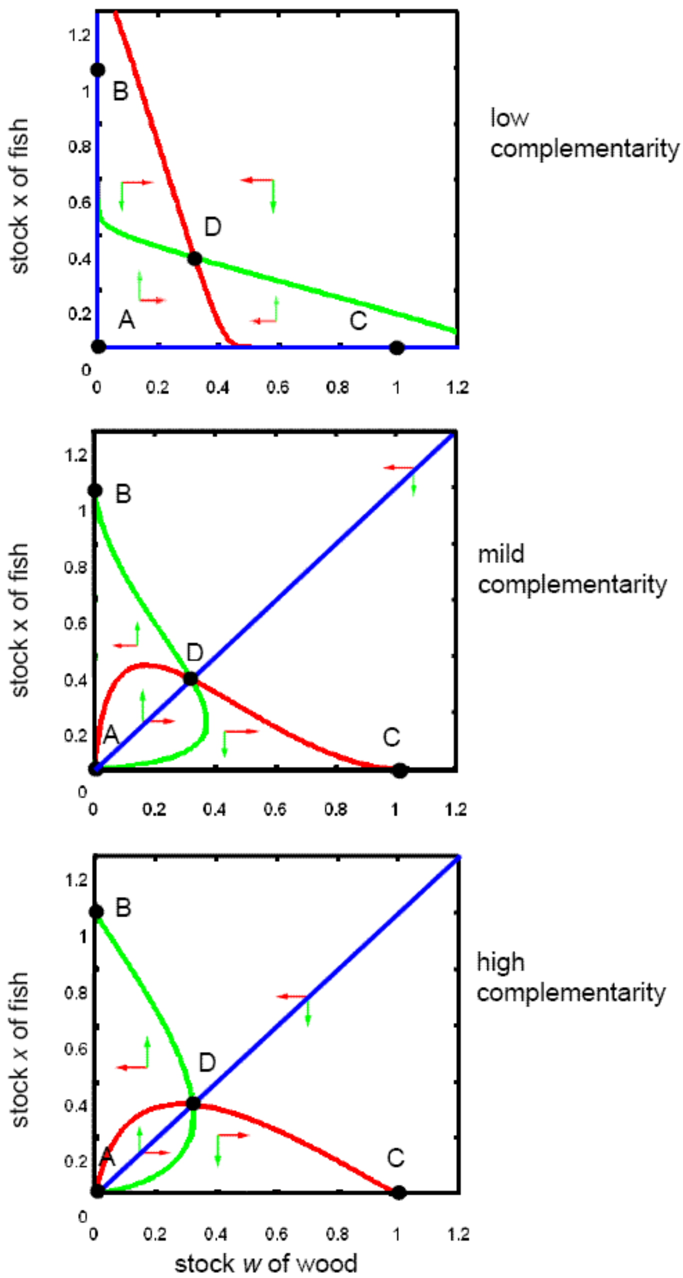

Figure 5: Phase diagrams for the ecosystem's dynamics with interspecies competition for different levels of complementarity between ecosystem services in consumption, s. Dynamics in each diagram are characterized by $dx/dt > 0(< 0)$ left (right) of the green line, and $dw/dt > 0(< 0)$ below (above) the 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: $\rho_x = \rho_w = 0.5, \kappa_x = \kappa_w = 1, \gamma_x = \gamma_w = 0.25, v_x = v_w = 1, \alpha = 0.6, n = 1$.

preferences, low levels of species competition are sufficient to significantly further decrease the resilience 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 that may be even larger with stronger species competition.

5.4 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-reenforcing 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).

Although our model analysis was based on specific functional forms and certain proper-

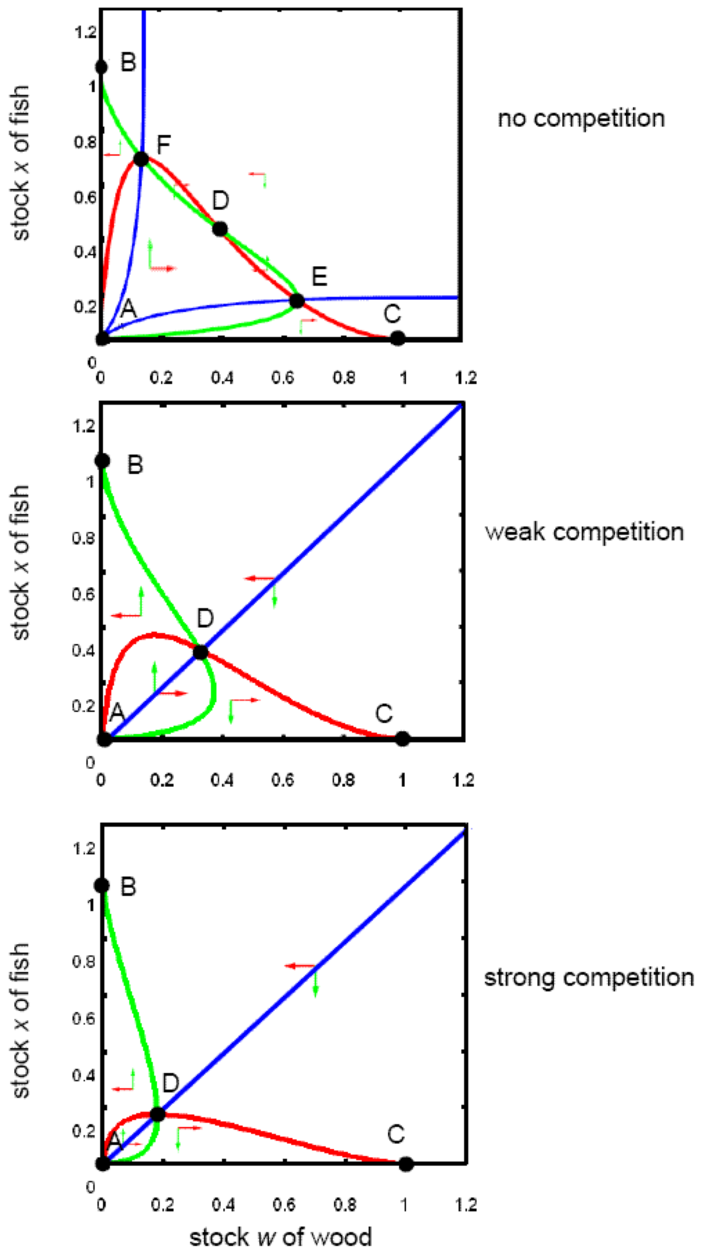

Figure 6: Phase diagrams for the ecosystem's dynamics at a given level of resource complementarity and increasing interspecies competition, γ_i. Dynamics in each diagram are characterized by $dx/dt > 0(< 0)$ left (right) of the green line, and $dw/dt > 0(< 0)$ below (above) the red line. Blue lines indicate 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: $\rho_x = \rho_w = 0.5, \kappa_x = \kappa_w = 1, v_x = v_w = 1, \alpha = 0.6, \sigma = 0.4, n = 1$.

ties 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 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

We are grateful to two anonymous reviewers and the subject editor for critical and constructive discussion, and to the German Federal Ministry of Education and Research for financial support under grant no. 01UN0607.

Appendix

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

$$\max_{y,c,h} u(y,c,h) \quad \text{subject to} \quad \omega = y + p_x c + p_w h \ , \tag{11}$$

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:

$$c(p_x, p_w, \omega) = \alpha\omega \frac{p_x^{-\sigma}}{p_x^{1-\sigma} + p_w^{1-\sigma}} \quad \text{and} \tag{12}$$

$$h(p_x, p_w, \omega) = \alpha\omega \frac{p_w^{-\sigma}}{p_x^{1-\sigma} + p_w^{1-\sigma}} \ . \tag{13}$$

Profits of representative firms harvesting fish and timber are given by

$$\pi_x = p_x \, c^{prod} - \omega e_x = (p_x v_x x - \omega) \ e_x \quad \text{and} \tag{14}$$

$$\pi_w = p_w \, h^{prod} - \omega e_w = (p_w v_w w - \omega) \ e_w \ , \tag{15}$$

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:

$$p_x = \frac{\omega}{v_x} x^{-1} \quad \text{and} \tag{16}$$

$$p_w = \frac{\omega}{v_w} w^{-1} \ . \tag{17}$$

Inserting these expressions into demand functions (12) and (13), we obtain open- access per-capita resource demands of fish and timber as functions of the respective resource stocks:

$$c(x,w) = \alpha \, \frac{(v_x x)^{\sigma}}{(v_x x)^{\sigma-1} + (v_w w)^{\sigma-1}} \text{ and} \tag{18}$$

$$h(x,w) = \alpha \, \frac{(v_w w)^{\sigma}}{(v_x x)^{\sigma-1} + (v_w w)^{\sigma-1}} \ . \tag{19}$$

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Chapter 6

Combining performance-based and action-based payments to provide environmental goods under uncertainty

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November 2012

This paper was published in *Ecological Economics* **85** (2013): 77–84.
http://dx.doi.org/10.1016/j.ecolecon.2012.11.001

Combining performance-based and action-based payments to provide environmental goods under uncertainty

Abstract: Payments for environmental services (PES) are widely adopted to support the conservation of biodiversity and other environmental goods. Challenges that PES schemes have to tackle are (i) environmental uncertainty and (ii) information asymmetry between the provider of the service (typically a farmer) and the regulator. Environmental uncertainty calls for action-based payment schemes, because of the more favorable risk allocation if the farmer is risk-averse. Information asymmetry, on the other hand, calls for a performance-based payment, because of the more direct incentives for the farmer. Based on a principal-agent model, we study the optimal combination of both, performance-based and action-based payments under conditions of environmental uncertainty and asymmetric information. We find that for a risk-neutral regulator a combination is optimal in the majority of cases and that the welfare gain of the combined scheme over a pure action-based (performance-based) payment increases with information asymmetry (environmental uncertainty). We further show that for a regulator who is risk-averse against fluctuations in environmental goods provision the optimal performance-based payment is lower than for a risk-neutral regulator. We quantitatively illustrate our findings in a case study on the enhancement of the butterfly Scarce Large Blue (*Maculinea teleius*) in Landau/Germany.

Keywords: Conservation Contracts, Payments for Ecosystem Services, Payments for Environmental Services, Biodiversity, Uncertainty

JEL-Classification: Q28; Q18; H41

6.1 Introduction

The protection and enhancement of environmental assets are objectives shared by many governments around the globe. Often these assets depend on how farmers manage their private land, but as they typically have characteristics of public goods, farmers have little incentives to make socially optimal decisions (Bardsley and Burfund 2008). For this reason policy instruments such as payments for environmental or ecosystem services (PES) have been advocated to create incentives similar to those that would be provided by market prices, if markets for environmental services would exist (e.g. Bulte et al. 2008, Corbera et al. 2007, Vatn 2009).

Two types of payment schemes are used in practice: Action-based payments are bound to a predefined action or measure, whereas performance-based payments are directly bound to the outcome of a desired ecosystem good or service[42]. Performance-based payments have the advantage that they set the direct incentive to provide ecosystem services efficiently (Matzdorf 2004, Zabel & Roe 2009). A drawback of performance-based payment schemes is that the risk of producing an ecosystem good comes at the expense of the farmer, since the quantity of environmental service also depends on external influences beyond the farmer's control. If the farmer is risk-averse, and the regulator is risk-neutral, a pure performance-based payment scheme thus leads to an inefficient risk allocation[43]. As a result, most existing schemes are action-based, although performance-based payments are sometimes applied for the conservation of an already given state or of existing biodiversity (Osterburg 2006, Hampicke 2001). Action-based payments may be a cost-effective alternative if there is a clear action that is required to provide the environmental good, known and observable by the regulator (Gibbons et al. 2011). If there is informational asymmetry between farmer and regulator, however, a pure action-based payment is likely to lead to an inefficient outcome.

In this paper, we consider payment schemes that combine performance-based and action-based payments. We set up a principal-agent model to study what combination of both is optimal when there is both environmental uncertainty affecting the provision of the environmental good and asymmetric information about how productive a management

[42] Many labels for these payment schemes can be found within the literature. Other common names for action-based payments are e.g. input- or measure-based payments, for performance-based payments the terms output-oriented, outcome- or result-based payments are also common.

[43] An efficiency improvement in the risk-allocation could be obtained by shifting risk from the risk-averse farmer to the risk-neutral regulator. One way of doing this (the one considered in this paper) is to combine a pure performance-based payment to some extent with an action-based payment.

	asymmetric information between farmer and regulator	both perfectly informed
environmental risk	combination	action-based
no environmental risk	performance-based	any

Table 1: Table showing the optimal PES scheme for a risk-neutral regulator.

action is for providing the environmental good.

We find that the optimal payment typically will be a combination of performance-based and action-based payments (see Table 1). A pure performance-based payment is optimal for a risk-neutral regulator (the principal) only if either there is no environmental risk or if the farmer (the agent) is risk-neutral. A pure action-based payment is optimal only if the regulator has full information about the marginal productivity of the actions for providing the environmental good. The performance-based fraction of the optimal payment increases with environmental uncertainty, while the action-based fraction increases with information asymmetry. These findings are also reflected in the welfare gains of the combined scheme over the pure performance-based or action-based schemes: the welfare gain, measured as the payoff of a risk-neutral regulator, of the optimally combined scheme over an optimally chosen, pure action-based (performance-based) payment increases with information asymmetry (environmental uncertainty).

The assumption of a risk-neutral regulator may be inappropriate, because society's marginal willingness to pay for the environmental asset may increase if an environmental asset becomes increasingly scarce. For this reason we also consider a regulator who is risk-averse against fluctuations in environmental goods provision. As the argument for an action-based payment scheme is the more favorable allocation of risk if the farmer is risk-averse but the regulator is risk-neutral, one might expect that the performance-based fraction of the optimal payment might be relatively higher when the regulator is risk-averse. We find, however, that the optimal performance-based payment actually decreases with the regulator's degree of risk aversion.

We apply our analysis to the case study on the enhancement of the butterfly Scarce Large Blue (*Maculinea teleius*) in Landau/Germany, based on data from the literature (Drechsler et al. 2007, Wätzold et al. 2008). Results indicate that the optimal combination of the performance-based and action-based payments may lead to a welfare gain of several thousand euros per hectare.

6.2 Principal-agent model of environmental good provision under uncertainty

We consider a principal-agent setting where a regulator (the principal) offers a PES to a single, representative farmer (the agent), who chooses an action that contributes to the production of an environmental good. This means, we assume that all farmers share the same characteristics with respect to preferences and production technology[44]. We thereby extend the approach of Zabel & Roe (2009), allowing for a combination of a performance-based payment with an action-based payment, and risk aversion on the regulator's side.

The temporal structure of the problem is that, first, the principal announces the payment scheme. Second, the agent decides on whether or not he would like to participate in the program. If he participates, he receives (or pays) a base-payment. Third, the agent chooses his action, and fourth, nature adds stochastic disturbance. Finally, the agent receives performance-based and action-based payments from the principal, and society enjoys the environmental good.

The quantity y of the environmental good is produced according to

$$y = \phi x + \varepsilon. \tag{1}$$

The provision of the environmental good can be increased by the farmer's action x with a constant marginal productivity ϕ. For example, x can be thought of as the area of farmland set aside for biodiversity protection. We consider y to be the *additional* environmental goods provided, i.e. y is the (net) growth of the environmental good. This growth is also affected by a stochastic disturbance ε, capturing environmental noise, which is independent and identically normally distributed with zero mean and standard deviation σ_ε[45].

Marginal productivity ϕ of the action x is known to the farmer, but not to the regulator. This information asymmetry arises, because the farmer knows the peculiarities of his farmland while the regulator does not. The regulator only knows a prior probability distribution over ϕ. We assume that this is any probability distribution with a mean $\bar{\phi}$ and variance σ_ϕ^2. The quantity x of the action exerted by the farmer is common knowledge of both farmer and regulator.

Some important and restrictive assumptions about the production of the environmental good are embodied in Eq. (1), which we shall discuss in the following.

[44] The question how to deal with heterogenous farmers (for example by designing adequate auction schemes such as, e.g. Latacz-Lohmann & van der Hamsvoort (1997)) is beyond the scope of this paper.

[45] Note that the net growth of the environmental good may be negative even with a positive effort x, due to environmental uncertainty.

(i) Taking asymmetry with regard to the observability of the farmer's action into account has similar effects as the information asymmetry with regard to marginal productivity and could be included in the model in a straightforward way. In either case the essential assumption is that the farmer may have more information about his contribution to the provision of the environmental good than the regulator.

(ii) Assuming *perfect* information on ϕ on the farmer's side is rather strong. However, the crucial aspect of this assumption is that the farmer has *better* information about what he is doing than the regulator. Assuming perfect information only simplifies the analysis. Environmental uncertainty, captured by ε in equation Eq. (1), is an aspect of incomplete information about the production of y faced by the farmer and the regulator to an equal extent.

(iii) As we are considering a representative farmer, assuming production of the environmental good according to Eq. (1) means that either there are no external effects between farmers, or that all external effects are internalized, for example by a farmer's association that negotiates about the PES contract with the regulator.

We consider a payment ω for the provision of the environmental good that is composed of a base payment b, a payment for the action, $a\,x$, and of a payment for the performance, i.e. the provision of the environmental good, $p\,y$,[46]

$$\omega = b + a\,x + p\,y. \tag{2}$$

Because of environmental uncertainty, y may be negative. In such a case the performance-based payment $p\,y$ will be negative as well, although typically the *expected* performance-based payment will be positive. The base payment b is chosen such that the farmer nevertheless has an incentive to participate in the PES scheme. Using the base payment to meet the participation constraint is in line with the recent literature on PES that has adopted this approach from labor economics (Zabel & Roe 2009)[47].

The participation constraint could also be met in other ways, for example by increasing the action-based component of the overall payment to compensate for expected losses in the performance-based component, or by restricting the performance-based payment to be

[46] We restrict our analysis to linear combinations of the three payment parts here. An analysis of more general payment structures is left for future research.

[47] One example for combined payment schemes can be found in Switzerland: Here the schemes contain site-specific direct payments, similar to the base payment considered here. Additional payments are bound to the condition that 7% of the farm area are managed in line with specific ecological standards, corresponding to the action-based payment considered here. Finally, the Swiss authority adds a performance-based payment when biodiversity is sufficiently high.

nonnegative. These approaches will lead to lower welfare, however, either because they induce distortions in the payment structure (as for the case of the adapted action-based payment), or because the expected public expenditures associated with the PES scheme are unnecessarily high (when performance-based payments are restricted to be nonnegative). For these reasons, a negative base payment may well be part of the optimal payment scheme.

If the farmer participates in the program, his payoff Y is given by

$$Y = \omega - \frac{c}{2} x^2. \tag{3}$$

Here we use c to denote the cost parameter of the action, with linearly increasing marginal costs. If the farmer does not participate, both payment and costs are zero, and thus the net payoff is zero.

The farmer maximizes expected utility. We assume a risk-averse farmer with preferences that exhibit constant absolute risk aversion (CARA):[48]

$$U = E_\varepsilon \left[-\exp\left(-\eta Y\right)\right], \tag{4}$$

where η is the coefficient of constant absolute risk aversion and E_ε denotes the expectation with respect to environmental uncertainty ε.

As participation is voluntary, expected utility of participation in the program must at least equal utility from not participating in the program and receiving a zero net payoff for sure. With $U(0) = -\exp(0) = -1$ denoting the reservation utility level of the farmer, the participation constraint is

$$E_\varepsilon \left[-\exp\left(-\eta Y\right)\right] \geq -1. \tag{5}$$

The regulator receives a benefit

$$v(y) = y - \frac{\rho}{2} y^2 \tag{6}$$

from the provision of the environmental good. The quadratic benefit function (6) captures in a simple form that the regulator may be averse against uncertainty in the provision of the environmental good, with ρ being the regulator's coefficient of risk aversion. The benefit $v(y)$ is measured in monetary terms, such that the regulator's net benefit is given by $v(y) - \omega$. This assumption may actually be quite restrictive. It presupposes that the

[48] We make this assumption to be able to solve the model analytically. It is in line with most models of this type. More realistic is the case of decreasing absolute risk aversion (Gollier 2001). A deeper exploration of this case is, again, left for future research.

environmental good desired by the regulator is well-defined and that further its benefit can be measured in monetary terms. For the case study, we use published results from a contingent valuation study for this purpose. We make this assumption here, as we are not interested in studying the effects of ill-defined objectives[49]. The quadratic benefit function (6) implies that the *marginal* environmental benefit is linear in y. As the marginal environmental benefit can be identified with the willingness to pay (WTP) for the provision of the environmental good, the specification Eq. (6) is in line with a common empirical specification of the WTP for environmental goods.

The optimal payment scheme (a, b, p) is derived by solving the regulator's optimization problem to maximize expected net benefit

$$\max_{a,p,b} E_\phi \left[E_\varepsilon[v(y) - \omega]\right] \tag{7}$$

subject to the constraints that the action x is chosen by the farmer such as to maximize the farmer's expected utility and the participation constraint Eq. (5).

6.3 Analytical results: Optimal combination of performance-based and action-based payments

The problem is solved backwards by first considering the farmer's optimization for given payment levels (b, a, p). Inserting Eq. (1) and (2) into Eq. (4), the farmer's expected utility is

$$E_\varepsilon \left[-\exp\left(-\eta \left(b + (a + p\,\phi)\,x + p\,\varepsilon - \frac{c}{2}\,x^2\right)\right)\right]. \tag{8}$$

Taking the expectation over environmental uncertainty we obtain

$$\tilde{U} = -\frac{1}{\eta}\ln(-U) = b + (a + p\,\phi)\,x - \frac{c}{2}\,x^2 - \frac{\eta}{2}\,p^2\,\sigma_\varepsilon^2, \tag{9}$$

which is the certainty equivalent of the income lottery generated by participating in the PES with uncertain provision of the environmental good.

Using the first-order conditions of utility maximization with respect to x, we find that the farmer's optimal choice of action is (see appendix 6.5)

$$x^\star = \begin{cases} \frac{a+p\,\phi}{c} & \text{if } \tilde{U} \geq 0 \\ 0 & \text{if } \tilde{U} < 0. \end{cases} \tag{10}$$

Assuming that the farmer participates in the PES program (which is the case if $\tilde{U} \geq 0$), the optimal action is increasing in both action-based and performance-based payments, and

[49] Ill-defined objectives may favor action-based payments compared to performance-payments (Zabel & Roe 2009).

decreasing in the cost parameter c, which is in line with intuition. For a given performance-based payment p, it is also increasing in the marginal productivity ϕ of the action (cf. appendix 6.5).

Using the result of Eq. (10) in Eq.(1), we find that the (uncertain) provision of the environmental good under payment scheme (b, a, p) is given by

$$y^* = \phi \, \frac{a + p\,\phi}{c} + \varepsilon. \tag{11}$$

For a given marginal productivity ϕ of the farmer's action, and a given payment scheme (b, a, p), the expected benefit of environmental good provision (conditional on ϕ) thus is

$$\begin{aligned} E_\varepsilon \left[v(y^*) \right] &= E_\varepsilon \left[\phi \, \frac{a + p\,\phi}{c} + \varepsilon - \frac{\rho}{2} \left(\phi \, \frac{a + p\,\phi}{c} + \varepsilon \right)^2 \right] \\ &= \phi \, \frac{a + p\,\phi}{c} - \frac{\rho}{2} \left(\phi \, \frac{a + p\,\phi}{c} \right)^2 - \frac{\rho}{2} \, \sigma_\varepsilon^2. \end{aligned} \tag{12}$$

To obtain this result, we have used that the expected value of ε is zero. Environmental uncertainty thus decreases the benefit of a risk-averse regulator, but this effect is independent of the payment scheme. This means that, in the absence of asymmetric information, risk aversion on the regulator's side has no influence on the optimal payment scheme. Put differently, we have the following lemma (which we need to derive result 3):

Lemma 1. *Environmental uncertainty does not directly affect the optimal payment scheme for a risk-averse regulator.*

Environmental uncertainty affects the optimal payment scheme indirectly, however, because the farmer is risk-averse, as we will show below. To meet the participation constraint, the regulator has to set the base payment b such that reservation utility is reached. The certainty equivalent of participating in the PES is obtained by using Eq. (10) in Eq. (9). Equating this to the certainty equivalent of not participating, which is equivalent to an income of zero, we obtain the minimal base payment b of

$$b^\star = \frac{\eta}{2} \, p^2 \, \sigma_\varepsilon^2 - \frac{(a + p\,\phi)^2}{2\,c}. \tag{13}$$

With a risk-averse farmer, environmental uncertainty increases the base payment. As the performance-based payment $p\,y$ may either be positive (in case of favorable environmental conditions) or negative (in case of very unfavorable environmental conditions), the effect of p on the based payment is ambiguous. The action-based payment a, by contrast, unambiguously decreases the base payment.

Overall, the base payment may well be negative, because the expected value of action-based and performance-based payments is positive. In such a case the regulator can use the base payment to reduce expected public expenditures associated with the PES scheme. Such a negative base payment then means that a farmer will have to make a payment to the regulator in order to benefit from the participation in the PES scheme. Depending on the parameter constellation, in particular if environmental uncertainty is large and the farmer is very risk-averse, the base payment may of course also be positive. In this case the participating farmer receives a payment just for agreeing to participate in the PES program.

Using Eq. (10), (11), and (13) in Eq. (2), the expected payment can be expressed as

$$E_\varepsilon[\omega] = \frac{(a + p\,\phi)^2}{2\,c} + \frac{\eta}{2}\,p^2\,\sigma_\varepsilon^2. \tag{14}$$

As the farmer is risk-averse, and the performance-based payment is affected by environmental uncertainty, the optimal payment rate p depends on environmental uncertainty. Using Eq. (12), (13), and (14) in Eq.(7), and employing Lemma 1, the regulator's optimization problem can be written as

$$\max_{a,p} E_\phi\left\{\phi\,\frac{a + p\,\phi}{c} - \frac{\rho}{2}\left(\phi\,\frac{a + p\,\phi}{c}\right)^2 - \frac{(a + p\,\phi)^2}{2\,c} - \frac{\eta}{2}\,p^2\,\sigma_\varepsilon^2\right\}. \tag{15}$$

In appendix 6.5 we show that the optimal action-based and performance-based payments are

$$a^\star = \frac{\bar\phi}{\Omega}\left(c\,\eta\,\sigma_\varepsilon^2 + 2\,\frac{\rho}{c}\,\bar\phi^2\,\sigma_\phi^2\right) \tag{16a}$$

$$p^\star = \frac{\sigma_\phi^2}{\Omega}\left(1 - \frac{\rho}{c}\,\left(\bar\phi^2 - \sigma_\phi^2\right)\right) \tag{16b}$$

with

$$\Omega \equiv \sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2 + \frac{\rho}{c}\,\left(2\,\sigma_\phi^2\,\left(\bar\phi^2 + 2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar\phi^2 + \sigma_\phi^2\right)\,\sigma_\varepsilon^2\right) + \frac{\rho^2}{c^2}\,\sigma_\phi^2\,\left(\bar\phi^4 + 3\,\sigma_\phi^4\right) > 0. \tag{17}$$

To analyze the optimal payment scheme, we first focus on the case of a risk-neutral regulator, assuming $\rho = 0$. In this case, the optimal payments are given by the following, much simpler expressions.

$$a^\star|_{\rho=0} = \bar\phi\,\frac{c\,\eta\,\sigma_\varepsilon^2}{\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2} \tag{18a}$$

$$p^\star|_{\rho=0} = \frac{\sigma_\phi^2}{\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2} \tag{18b}$$

The following result is obtained immediately.

Result 1. *For a risk-neutral regulator ($\rho = 0$), the optimal PES scheme includes both an action-based and a performance-based component, except for the following cases*

- *If the farmer is risk-neutral ($\eta = 0$), or if there is no environmental uncertainty ($\sigma_\varepsilon^2 = 0$), the action-based component of the optimal payment scheme is zero.*
- *If there is no asymmetric information, $\sigma_\phi^2 = 0$, the performance-based component of the optimal payment scheme is zero.*

This result shows that an optimal PES scheme should combine both an action-based and a performance-based payment. The only exceptions are extreme cases where either environmental uncertainty plays no role or the regulator has perfect information about the productivity of the farmer's actions. Furthermore, the relative shares of the action-based and the performance-based component of the optimal PES scheme depends on environmental uncertainty and information asymmetry in a very intuitive way, as stated in the following result.

Result 2. *a) For a risk-neutral regulator ($\rho = 0$), the optimal action-based payment increases and the optimal performance-based payment decreases with the farmer's degree of risk aversion, η, and with environmental uncertainty, σ_ε.*

b) For a risk-neutral regulator ($\rho = 0$), the optimal action-based payment decreases and the optimal performance-based payment increases with the degree of asymmetric information, σ_ϕ.

We now turn to the case of a risk-averse regulator, assuming $\rho > 0$, and focus on the question how the optimal payment scheme compares to the case of a risk-neutral regulator. The optimal payment scheme is much more complicated than in the case of a risk-neutral regulator. We show in appendix 6.5 that the optimal action-based payment depends in an ambiguous way on the degree of risk aversion. For low levels of risk aversion for both the regulator and the farmer, the optimal action-based payment increases with the regulator's degree of risk aversion. We find that even for a risk-neutral farmer the optimal action-based payment for a risk-averse regulator is positive, which is different from the case of a risk-neutral regulator. For very high levels of risk aversion of regulator and farmer, however, the optimal action-based payment will decrease again with the regulator's risk aversion. The performance-based payment, by contrast, decreases with the regulator's degree of risk aversion whenever it is positive at all.

Result 3. *a) For a risk-averse regulator, the optimal action-based payment is positive even when the farmer is risk-neutral.*

b) The optimal performance-based payment decreases with the regulator's degree of risk aversion,

$$\frac{\partial p^\star}{\partial \rho} < 0 \quad \textit{for all } p^\star > 0. \tag{19}$$

The intuition behind this result is as follows. The farmer will choose his action $x^\star$ in response to the payment according to Eq. (10). An increase in the action-based payment will increase the farmer's action independently of the marginal productivity, while an increase in the performance-based payment will lead to a lower (higher) increase in the level of his action if marginal productivity is low (high). Relative to the action-based payment, the performance-based payment thus amplifies the effect of the regulator's uncertainty on marginal productivity on the provision of the environmental good. The more risk-averse the regulator, the relatively less attractive becomes the performance-based payment compared to the action-based payment[50].

In the extreme, the performance-based payment may even become negative. This is the case if

$$\rho > \frac{c}{\bar{\phi}^2 - \sigma_\phi^2}. \tag{20}$$

A second effect is that the presence of informational asymmetry and the associated risk premium make the payment for environmental services overall less attractive for the risk-averse regulator. If this effect is sufficiently strong – which is the case for high environmental uncertainty – also the optimal action-based payment is lower for a risk-averse compared to a risk-neutral regulator.

As a final step of the analysis we study how high is the welfare gain, measured by the regulator's objective function, for the combined payment scheme compared to either a pure action-based or a pure performance-based payment scheme. The pure action-based, or performance-based, schemes are obtained by setting $p \equiv 0$, or $a \equiv 0$, in the regulator's optimization problem Eq. (15).

In appendix 6.5 we derive the welfare levels for all three payment schemes, assuming a risk-averse regulator. We find that the combined payment scheme outperforms the pure action-based scheme except for the case when the regulator has full information, i.e. $\sigma_\phi^2 = 0$. The combined scheme outperforms the pure performance-based scheme except for the case when there is no environmental uncertainty, $\sigma_\varepsilon^2 = 0$, *and* when the regulator is risk-neutral, $\rho = 0$. For a risk-averse regulator, the pure performance-based scheme is worse than the combined scheme even in the absence of environmental uncertainty (see also result 3a).

For a risk-neutral regulator, the comparisons for the welfare levels is as follows. The welfare gain of the combined payment scheme over the pure action-based scheme is given

[50] This result depends on the assumption that the informational asymmetry between the farmer and the regulator is with regard to the marginal productivity of the action, but that there is no hidden action.

by

$$E\left[v(y)-\omega\right] - E\left[v(y)-\omega\right]\big|_{p\equiv 0}\Big|_{\rho=0} = \frac{1}{2\,c}\,\frac{\sigma_\phi^4}{\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2}, \tag{21a}$$

and the welfare gain of the combined payment scheme over the pure performance-based scheme is given by

$$E\left[v(y)-\omega\right] - E\left[v(y)-\omega\right]\big|_{a\equiv 0}\big|_{\rho=0} = \frac{1}{2\,c}\,\frac{\left(\bar\phi\,c\,\eta\,\sigma_\varepsilon^2\right)^2}{\left(\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2\right)\,\left(\bar\phi^2 + \sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2\right)}. \tag{21b}$$

Finally, the welfare difference between the pure performance-based PES scheme and the pure action-based one is given by

$$E\left[v(y)-\omega\right]\big|_{a\equiv 0} - E\left[v(y)-\omega\right]\big|_{p\equiv 0}\Big|_{\rho=0} = \frac{1}{2\,c}\,\frac{\sigma_\phi^4 + \bar\phi^2\,\left(\sigma_\phi^2 - c\,\eta\,\sigma_\varepsilon^2\right)}{\bar\phi^2 + \sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2}. \tag{21c}$$

Using these relationships, we obtain the following result:

Result 4. *For a risk-neutral regulator ($\rho = 0$),*

a) *The welfare gain of the combined PES scheme over the pure action-based scheme increases with information asymmetry σ_ϕ^2 and decreases with both the farmer's degree of risk aversion η and environmental uncertainty, σ_ε^2.*

b) *The welfare gain of the combined PES scheme over the pure performance-based scheme decreases with information asymmetry σ_ϕ^2 and increases with both the farmer's degree of risk aversion η and environmental uncertainty, σ_ε^2.*

c) *The pure performance-based PES scheme is better than the pure action-based one if and only if*

$$c\,\eta\,\sigma_\varepsilon^2 < \sigma_\phi^2\,\left(1 + \frac{\sigma_\phi^2}{\bar\phi^2}\right). \tag{22}$$

Inequality (22) gives an explicit condition which instrument to use if only pure performance-based or action-based payments are available. As found in Result 2, the performance-based component of the optimal payment tends to be low when the farmer's coefficient of risk-aversion, η, and environmental uncertainty, σ_ε are high, while the action-based component of the optimal payment tends to be low when the degree of informational asymmetry is high. These effects are also captured in condition (22). A "high" coefficient of risk-aversion thereby means that $1/\eta$ is low relative to the cost parameter c, and a "high" informational asymmetry means that σ_ϕ is high relative to $\bar\phi$.

In order to quantify these effects, we apply our analysis to the case of butterfly protection.

6.4 Quantitative application: Optimal payment scheme for butterfly protection

We base our quantitative application on published ecological-economic studies on the conservation of the Scarce Large Blue (*Maculinea teleius*) in the region of Landau, Germany (Drechsler et al. 2007, Wätzold et al. 2008). Within European nature conservation, butterflies of the *Maculinea* genus are considered as important flagship species (Dierks & Fischer 2009, Thomas & Settele 2004) and have suffered substantial population declines with local extinctions in recent years (Wynhoff 1998). *M. teleius* is therefore considered as a threatened species in Europe (Swaay & Warren 1999). The butterfly is characterized by a complex life cycle whereby the early instars of this species first feed on the blossoms of *Sanguisorba officinalis* (Great Burnet). Late instars are carried by ants (e.g. *Myrmica rubra*) into their nests where the larvae actively prey on ant brood. Especially the blooming of *S. officinalis* and therefore the egg deposition and stage of development of the early instars of *M. teleius* are determined by the mowing regime (Dierks & Fischer 2009). Conservation measures for different mowing regimes have been applied for the enlargement and enhancement of *M. teleius*.

Before turning to the application of our model, we would like to emphasize the limitations with regard to the direct applicability of our analysis for the conservation of Scarce Large Blue in the Landau region. To obtain effective conservation, many farmers have to be included in the protection scheme, and external effects of management arise due to metapopulation dynamics (Drechsler et al. 2007). Thus, the assumption of a single, representative farmer is a strong abstraction from reality. It should therefore be kept in mind that further sophistication of payment schemes may be needed to actually implement the optimal conservation scheme for butterfly conservation in Landau. Given these caveats, the quantitative results derived in the following still give some feeling how large the potential welfare gains of combining performance-based and action-based payments could be.

Environmental benefit y is measured in monetary terms. Wätzold et al. (2008, Table 1) provide modeling results of how many butterflies can be conserved by applying conservation measures (alternative mowing regimes) to a certain area of farmland. Specifically, the authors consider three projects that correspond to 4, 16, and 64 ha on which conservation measures are applied. The results of a Contingent Valuation study published in Wätzold et al. (2008, Table 3), indicate societal conservation benefits of approximately 260, 297, and 426 thousand euros for the three projects. Taking the number of hectares with conservation measures as the farmer's action x, a simple OLS regression shows that the expected marginal productivity of applying conservation measures is $\bar{\phi} = 2.74$ thousand euros per hectare[51].

According to Drechsler et al. (2007, page 183), the coefficient of variation of marginal productivity is about $\sigma_\phi/\bar{\phi} = 0.25$. Hence, $\sigma_\phi^2 = 0.47$. Wätzold et al. (2008) use a linear cost function with constant marginal costs of 0.123 thousand euros per hectare. However, Figure 7b in Drechsler et al. (2007) suggests that it is equally plausible to assume an overall convex cost function. A quadratic cost function with cost parameter $c = 0.015$ [1000 euros/ha] gives the best fit for the range of hectares (4, 16 and 64) considered by Wätzold et al. (2008), assuming log-normal errors.

We assume a coefficient of relative risk aversion for farmers of $\eta = 0.74$, which is consistent with experimental evidence for Western Europeans (Andersen et al. 2008). The mean income for German farmers in 2011 was 24.6 thousand euros per year. This leads to an estimate for the degree of absolute risk aversion of about $\eta = 0.74/24.6 = 0.03$/thousand euros.

Since we have no information on the degree of environmental uncertainty, we vary the standard deviation of environmental noise, σ_ε. Furthermore, we also vary the regulator's coefficient of risk aversion to obtain an insight of how this parameter influences the quantitative results. These are shown in Figure 1. The left-hand panel in this figure shows the optimal performance-based payment, and the right-hand side the optimal action-based payment as functions of environmental stochasticity σ_ε. In line with *Result 2*, the optimal performance-based payment decreases, and the action-based payment increases, with environmental uncertainty. With a risk-averse regulator ($\rho > 0$), the effect corresponds to *Result 4*: With increasing risk aversion, the performance-based fraction of the combined scheme becomes less attractive for the regulator. The optimal action-based fraction is even higher than the performance-based fraction under circumstances with low environmental stochasticity and increases slightly with increasing σ_ε while the performance-based fraction decreases respectively.

Turning to a risk-neutral regulator again we quantify the overall welfare gains of the different payment schemes for the case of the protection of the Scarce Large Blue (*M. teleius*) in Landau. The results shown in Figure 2 correspond to the theoretical findings of *Result 1*:

[51] The estimated equation is $y_1 = y_0 + \phi x + \epsilon$, which results in the estimates $y_0 = 250.8$ (standard error 2.9) and $\phi = 2.74$ (standard error 0.075), with an $R^2 = 0.999$. Allowing for $y_0 > 0$ we assume that there might be some willingness to pay also when $x = 0$, and use $y = y_1 - y_0$ as a measure for the additional environmental good provided. Thus we interpret the results of the contingent valuation study differently than Wätzold et al. (2008), who assume that the estimated willingness to pay must be zero for $x = 0$.

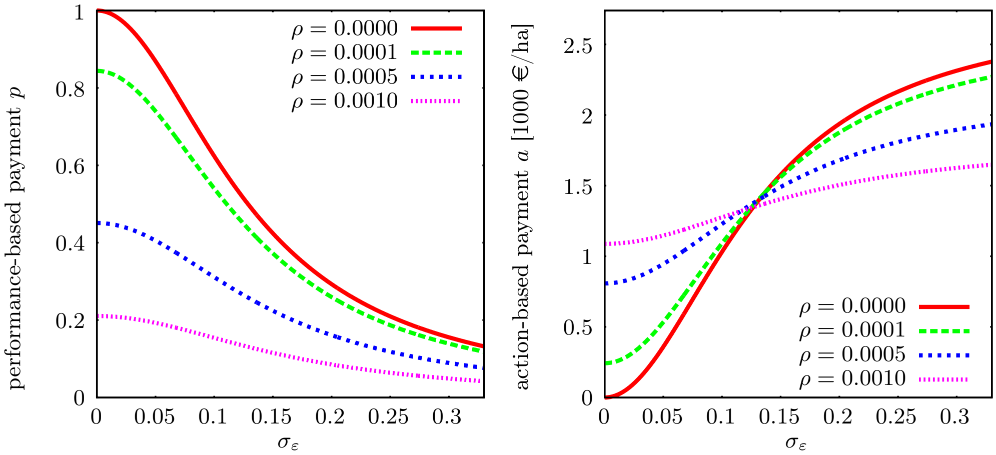

Figure 1: Optimal performance and action-based payments for protection of the Scarce Large Blue (*M. teleius*) in Landau, Germany. The model is calibrated using data from Drechsler et al. (2007) and Wätzold et al. (2008).

For low environmental uncertainty the pure performance-based PES may do substantially better than the pure action-based PES, while for high environmental uncertainty, the pure action-based PES would be preferred. However, the combined payment scheme leads always to a higher welfare than either the pure action-based or the pure performance-based scheme. The welfare gain of the combined payment over the pure action-based PES decreases with environmental uncertainty. Furthermore, the welfare gain of the combined scheme over the pure performance-based PES is zero in the absence of environmental uncertainty, in which case the pure performance-based payment is optimal, as shown in *Result 1*, but is positive when environmental uncertainty matters. Overall the welfare gain of the combined scheme over the pure schemes may sum up to several thousand euros per hectare.

6.5 Conclusion

In the ongoing discussion on new policy instruments for the provision of environmental goods such as PES, performance-based payments gain significant support. In contrast to action-based payment schemes, which are bound to a predefined action or measure, performance-based payments are directly bound to the outcome of the desired environmental good. Even though in the literature performance-based payments occur as the preferred concept in many ways, examples of action-based payments still predominate in practical application. Although it is acknowledged that an action-based payment scheme is not optimal under information asymmetry, a performance-based payment scheme may not

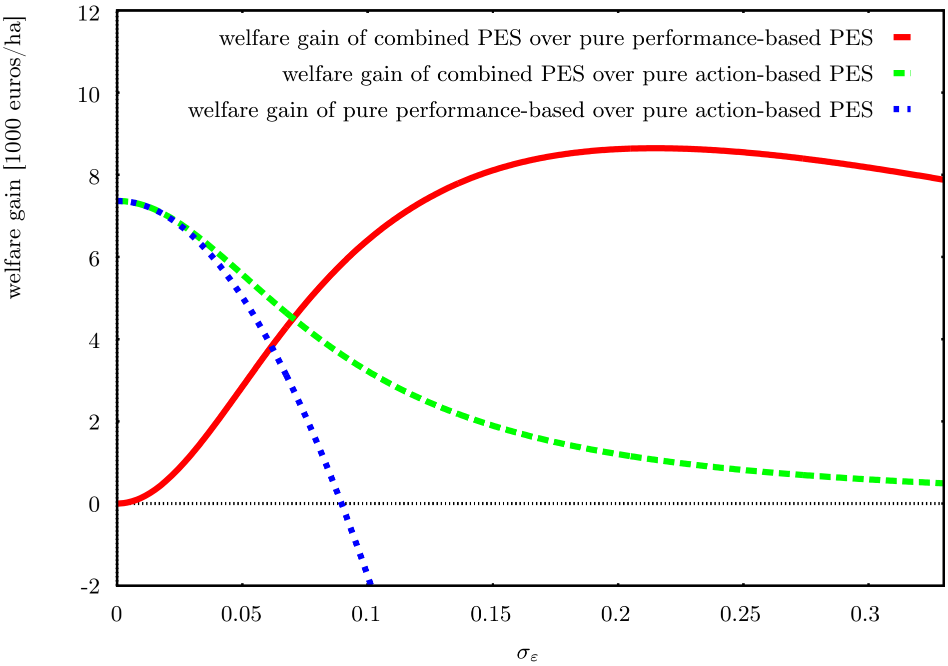

Figure 2: Welfare gains of combined payment scheme over pure performance-based or pure action-based payment schemes, and of pure performance-based payment scheme over pure action-based payment scheme for a risk-neutral regulator ($\rho = 0$) aiming to protect the Scarce Large Blue (*M. teleius*) in Landau, Germany. Data as in figure 1.

be the preferred option either if the performance is risky. Therefore, in this paper we have studied how both types of payment schemes could be optimally combined, because typically both information asymmetry and environmental uncertainty matter in a real-world context.

Based on a principal-agent model we have shown that an exclusively performance-based payment is optimal only if there is no environmental uncertainty or if both the farmer and the regulator are risk-neutral. An exclusively action-based payment is optimal only, if the regulator has full information about the productivity of the action i.e. if there is no information asymmetry. In every other case a combination of performance-based and action-based payments (with different weighting) may increase welfare. Accordingly, the welfare gain of the combined scheme over the pure action-based scheme increases with information asymmetry, while the welfare gain of the combined scheme over the pure performance-based scheme increases with environmental uncertainty.

With a risk-averse regulator the situation changes as follows. The regulator is not directly affected through environmental uncertainty, but indirectly through the farmer's choice of action, since the farmer chooses his actions considering environmental uncertainty. Under information asymmetry a performance-based payment would amplify the effect of the regulator's uncertainty about the action's productivity compared to the action-based payment. Thus, for a risk-averse regulator, the performance-based payment tends to pay out less favorable compared to the action-based payment.

Our quantitative application to the case study of butterfly conservation indicates that the welfare gain of the combined scheme over the pure action-based or performance-based schemes may be substantial, reaching several thousand euros per hectare.

Acknowledgements

We thank Stefan Baumgärtner, Uwe Latacz-Lohmann, and Frank Wätzold for their helpful discussion and comments. The study was supported financially with a PhD scholarship from the Heinrich-Böll foundation.

Appendix

Farmer's optimal choice of action

The first-order condition of maximizing Eq. (9) with respect to x is

$$(a + p\,\phi) - c\,x = 0. \tag{E.23}$$

Rearranging leads to Eq. (10).

Assuming that the farmer participates in the PES program, the comparative statics of the farmer's optimal action with respect to a, p, c, and ϕ are obtained as follows

$$\frac{\partial x^\star}{\partial a} = \frac{1}{c} > 0 \tag{E.24}$$

$$\frac{\partial x^\star}{\partial p} = \frac{\phi}{c} > 0 \tag{E.25}$$

$$\frac{\partial x^\star}{\partial c} = -\frac{a + p\,\phi}{c^2} < 0 \tag{E.26}$$

$$\frac{\partial x^\star}{\partial \phi} = \frac{p}{c} > 0 \tag{E.27}$$

Optimal payment scheme

Taking expectation over ϕ according to the regulator's assumed distribution with mean $\bar{\phi}$ and variance σ_ϕ^2, the optimization problem can be written as

$$\max_{a,p} \Bigg\{ \frac{a\,\bar{\phi} + p\,(\bar{\phi}^2 + \sigma_\phi^2)}{c} - \frac{\rho}{2}\,\frac{a^2\,(\bar{\phi}^2 + \sigma_\phi^2) + 2\,a\,p\,(\bar{\phi}^3 + 3\,\bar{\phi}\,\sigma_\phi^2) + p^2\,(\bar{\phi}^4 + 6\,\bar{\phi}^2\,\sigma_\phi^2 + 3\,\sigma_\phi^4)}{c^2} - \frac{a^2 + 2\,a\,p\,\bar{\phi} + p^2\,\bar{\phi}^2 + p^2\,\sigma_\phi^2}{2\,c} - \frac{\eta}{2}\,p^2\,\sigma_\varepsilon^2 \Bigg\}. \tag{E.28}$$

After few steps of simplification, the first-order conditions with respect to a and p can be written as

$$c\,\left(a^\star + \bar{\phi}\,p^\star - \bar{\phi}\right) + \rho\,\left(a^\star\,(\bar{\phi}^2 + \sigma_\phi^2) + \bar{\phi}\,p^\star\,(\bar{\phi}^2 + 3\,\sigma_\phi^2)\right) = 0 \tag{E.29}$$

$$\frac{\bar{\phi}^2 + \sigma_\phi^2}{c}\,(1 - p^\star) - \frac{a^\star\,\bar{\phi}}{c} - \eta\,p^\star\,\sigma_\varepsilon^2 - \rho\,\frac{a^\star\,(\bar{\phi}^3 + 3\,\bar{\phi}\,\sigma_\phi^2) + p^\star\,(\bar{\phi}^4 + 6\,\bar{\phi}^2\,\sigma_\phi^2 + 3\,\sigma_\phi^4)}{c^2} = 0. \tag{E.30}$$

Solving for $a^\star$ and $p^\star$, we obtain

$$a^\star = \frac{2c\bar{\phi}^3\rho\sigma_\phi^2 + c^3\eta\bar{\phi}\sigma_\varepsilon^2}{\left(c+\bar{\phi}^2\rho\right)^2\sigma_\phi^2 + 4c\rho\sigma_\phi^4 + 3\rho^2\sigma_\phi^6 + c^2\eta\left(c+\rho\left(\bar{\phi}^2+\sigma_\phi^2\right)\right)\sigma_\varepsilon^2} \tag{E.31a}$$

$$p^\star = \frac{c\,\sigma_\phi^2\left(c-\rho\left(\bar{\phi}^2-\sigma_\phi^2\right)\right)}{\left(c+\bar{\phi}^2\rho\right)^2\sigma_\phi^2 + 4c\rho\sigma_\phi^4 + 3\rho^2\sigma_\phi^6 + c^2\eta\left(c+\rho\left(\bar{\phi}^2+\sigma_\phi^2\right)\right)\sigma_\varepsilon^2}. \tag{E.31b}$$

The denominator of these expressions is

$$\begin{aligned}
&\left(c+\bar{\phi}^2\rho\right)^2\sigma_\phi^2 + 4c\rho\sigma_\phi^4 + 3\rho^2\sigma_\phi^6 + c^2\eta\left(c+\rho\left(\bar{\phi}^2+\sigma_\phi^2\right)\right)\sigma_\varepsilon^2 \\
&\quad = \left(c^2 + 2\,c\,\bar{\phi}^2\rho + \bar{\phi}^4\rho^2\right)\sigma_\phi^2 + 4c\rho\sigma_\phi^4 + 3\rho^2\sigma_\phi^6 + c^2\eta\left(c+\rho\left(\bar{\phi}^2+\sigma_\phi^2\right)\right)\sigma_\varepsilon^2 \\
&\quad = c^2\,\left(\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2\right) \\
&\qquad + \rho\,c\,\left(2\,\sigma_\phi^2\,\left(\bar{\phi}^2 + 2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2+\sigma_\phi^2\right)\,\sigma_\varepsilon^2\right) \\
&\qquad + \rho^2\,\sigma_\phi^2\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right).
\end{aligned} \tag{E.32}$$

Simplifying and dividing by c^2 leads to Eq. (17). Plugging into Eqs. (B.31a) and (B.31b) , we obtain Eqs. (16a) and (16b). (16).

Proof of result 3

Differentiating Eq. (16b) with respect to ρ yields

$$\begin{aligned}
\frac{\partial p^\star}{\partial\rho} &= \frac{\sigma_\phi^2}{\Omega^2}\Bigg[-\frac{1}{c}\,\left(\bar{\phi}^2-\sigma_\phi^2\right)\,\Bigg(\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2 \\
&\quad + \frac{\rho}{c}\,\left(2\,\sigma_\phi^2\,\left(\bar{\phi}^2+2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2+\sigma_\phi^2\right)\,\sigma_\varepsilon^2\right) + \frac{\rho^2}{c^2}\,\sigma_\phi^2\,\left(\bar{\phi}^4+3\,\sigma_\phi^4\right)\Bigg) \\
&\quad - \left(1-\frac{\rho}{c}\,\left(\bar{\phi}^2-\sigma_\phi^2\right)\right)\,\Bigg(\frac{1}{c}\,\left(2\,\sigma_\phi^2\,\left(\bar{\phi}^2+2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2+\sigma_\phi^2\right)\,\sigma_\varepsilon^2\right) \\
&\quad + \frac{2\,\rho}{c^2}\,\sigma_\phi^2\,\left(\bar{\phi}^4+3\,\sigma_\phi^4\right)\Bigg)\Bigg] \\
&= \frac{\sigma_\phi^2}{c\,\Omega^2}\Bigg[-\left(\bar{\phi}^2-\sigma_\phi^2\right)\,\left(\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2 - \frac{\rho^2}{c^2}\,\sigma_\phi^2\,\left(\bar{\phi}^4+3\,\sigma_\phi^4\right)\right) \\
&\quad - \left(2\,\sigma_\phi^2\,\left(\bar{\phi}^2+2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2+\sigma_\phi^2\right)\,\sigma_\varepsilon^2 + \frac{2\,\rho}{c}\,\sigma_\phi^2\,\left(\bar{\phi}^4+3\,\sigma_\phi^4\right)\right)\Bigg] \\
&= -\frac{\sigma_\phi^2}{c\,\Omega^2}\left[3\,\sigma_\phi^2\,\left(\bar{\phi}^2+2\,\sigma_\phi^2\right) + 2\,c\,\eta\,\bar{\phi}^2\,\sigma_\varepsilon^2 + \left(2-\frac{\rho}{c}\,\left(\bar{\phi}^2-\sigma_\phi^2\right)\right)\,\frac{\rho}{c}\,\sigma_\phi^2\,\left(\bar{\phi}^4+3\,\sigma_\phi^4\right)\right]
\end{aligned} \tag{E.33}$$

Differentiating Eq. (16b) with respect to ρ yields

$$\begin{aligned}\frac{\partial a^\star}{\partial \rho} = \frac{\bar{\phi}}{c\,\Omega^2}\Bigg[2\,\bar{\phi}^2\,\sigma_\phi^2\Bigg(\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2 \\ + \frac{\rho}{c}\,\left(2\,\sigma_\phi^2\,\left(\bar{\phi}^2 + 2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2 + \sigma_\phi^2\right)\,\sigma_\varepsilon^2\right) + \frac{\rho^2}{c^2}\,\sigma_\phi^2\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right)\Bigg) \\ - \left(c\,\eta\,\sigma_\varepsilon^2 + 2\,\frac{\rho}{c}\,\bar{\phi}^2\,\sigma_\phi^2\right)\Bigg(2\,\sigma_\phi^2\,\left(\bar{\phi}^2 + 2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2 + \sigma_\phi^2\right)\,\sigma_\varepsilon^2 \\ + \frac{2\,\rho}{c}\,\sigma_\phi^2\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right)\Bigg)\Bigg] \\ = \frac{\bar{\phi}}{c\,\Omega^2}\Bigg[2\,\bar{\phi}^2\,\sigma_\phi^2\left(\sigma_\phi^2 + c\,\eta\,\sigma_\varepsilon^2 - \frac{\rho^2}{c^2}\,\sigma_\phi^2\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right)\right) \\ - c\,\eta\,\sigma_\varepsilon^2\left(2\,\sigma_\phi^2\,\left(\bar{\phi}^2 + 2\,\sigma_\phi^2\right) + c\,\eta\,\left(\bar{\phi}^2 + \sigma_\phi^2\right)\,\sigma_\varepsilon^2 + \frac{2\,\rho}{c}\,\sigma_\phi^2\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right)\right)\Bigg] \\ = \frac{\bar{\phi}}{c\,\Omega^2}\Bigg[2\,\bar{\phi}^2\,\sigma_\phi^4\left(1 - \frac{\rho^2}{c^2}\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right)\right) \\ - c\,\eta\,\sigma_\varepsilon^2\left(4\,\sigma_\phi^4 + c\,\eta\,\left(\bar{\phi}^2 + \sigma_\phi^2\right)\,\sigma_\varepsilon^2 + \frac{2\,\rho}{c}\,\sigma_\phi^2\,\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right)\right)\Bigg]\end{aligned} \tag{E.34}$$

Proof of result 4

Using Eq. (16a) and (16b) in the regulator's objective function that is given in Eq. (E.28), we obtain after few steps of rearrangement

$$\begin{aligned}&E\left[v(y) - \omega\right] \\ &= \frac{1}{2\,c}\,\frac{\sigma_\phi^2\left(\bar{\phi}^2 + \sigma_\phi^2\right) + \frac{\rho}{c}\,\sigma_\phi^2\left(\bar{\phi}^4 + \sigma_\phi^4\right) + c\,\eta\,\bar{\phi}^2\,\sigma_\varepsilon^2}{\sigma_\phi^2 + 2\,\frac{\rho}{c}\,\sigma_\phi^2\left(\bar{\phi}^2 + 2\,\sigma_\phi^2\right) + \frac{\rho^2}{c^2}\,\sigma_\phi^2\left(\bar{\phi}^4 + 3\,\sigma_\phi^4\right) + c\,\eta\,\sigma_\varepsilon^2\left(1 + \frac{\rho}{c}\,\sigma_\phi^2\left(\bar{\phi}^2 + \sigma_\phi^2\right)\right)}\end{aligned} \tag{E.35}$$

for the combined PES.

It is straightforward to verify that the optimal pure action-based payment would be

$$a^\star|_{p\equiv 0} = \frac{\bar{\phi}}{1 + \frac{\rho}{c}\left(\bar{\phi}^2 + \sigma_\phi^2\right)}. \tag{E.36}$$

Using this, together with $p \equiv 0$, in the regulator's objective function yields a welfare level of

$$E\left[v(y) - \omega\right]|_{p\equiv 0} = \frac{1}{2\,c}\,\frac{\bar{\phi}^2}{1 + \frac{\rho}{c}\left(\bar{\phi}^2 + \sigma_\phi^2\right)} \tag{E.37}$$

for the pure action-based PES. Finally, the optimal pure performance-based payment would be

$$p^\star|_{p\equiv 0} = \frac{\bar{\phi}^2 + \sigma_\phi^2}{\bar{\phi}^2 + \sigma_\phi^2 + \frac{\rho}{c}\left(\bar{\phi}^4 + 6\,\bar{\phi}^2\,\sigma_\phi^2 + 3\,\sigma_\phi^4\right) + c\,\eta\,\sigma_\varepsilon^2}. \tag{E.38}$$

With this, the welfare level is

$$E\left[v(y)-\omega\right]\big|_{a\equiv 0}=\frac{1}{2\,c}\,\frac{(\bar{\phi}^2+\sigma_\phi^2)^2}{\bar{\phi}^2+\sigma_\phi^2+\frac{\rho}{c}\,(\bar{\phi}^4+6\,\bar{\phi}^2\,\sigma_\phi^2+3\,\sigma_\phi^4)+c\,\eta\,\sigma_\varepsilon^2} \tag{E.39}$$

for the pure performance-based PES.

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Chapter 7

Discussion and Conclusion

Within the research field of environmental and resource economics, resilience is increasingly believed to be a key concept for the management of ecological-economic systems. However in many cases it is not taken into account that the paradigm of resilience as a descriptive concept is not sufficient to provide management directives for these systems. Policy Instruments such as conservation contracts are a possible management strategy to incentivize behavior that attempts to maintain or to reach a specific system state which corresponds to a desired level of ecosystem services. Up to this point characteristics of systems, such as the level of resilience or the possibility of non-linear reactions, receive no consideration within the design of conservation policy. As a consequence, the characteristics of ecological-economic systems also receive no consideration for the management of these systems. This dissertation concluded that these characteristics should be regarded for contract design to define proper management guidelines.

In the following section the insights of the dissertation thesis are reviewed, contributing to the overall objective of the dissertation: The design of policy instruments for the management of ecological-economic systems with respect to system dynamics. In addition the integration and contribution to the current state of the literature is emphasized. A second section addresses a critical appraisal of the applied methodology and the choice of model assumptions. A section proposing and recommending further research completes the chapter.

7.1 Discussion of results

During the research for the dissertation the paradigm of resilience has been revealed as applicable and beneficial for the design of conservation contracts and for the definition of management directives to obtain desired ecosystem services in a number of ways:

Payments, which feature to prevent or achieve a special environmental service, are for their success heavily dependent on the functioning and interdependencies within ecological-economic systems, e.g. species interactions, growth rates, the access to resources and consumer preferences for the environmental service. Additional economical influences also change the stability landscape of the system as shown within the dissertation. Since the performance of conservation contracts is heavily dependent on the reaction and feedback of the underlying ecological-economic system, deep knowledge about system dynamics is important for the design of policy instruments. However, up to this point system dynamics have seldom been connected to the design of conservation contracts and to questions concerning the management of ecological-economic systems, which shall be guided by such policy instruments.

As argued within the introduction part of the dissertation thesis “Payments for environmental“ or “ecosystem services“ (PES) are a common instrument for the provision of public goods such as biodiversity. Considering this the dissertation discussed the current state of the literature regarding the PES approach. A common definition of “Ecosystem services” is, as the Millennium Ecosystem Assessment (MA) outlined it, the “[...] provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits” (MA 2005: 27). This is also a definition many authors draw upon to as their key reference e.g. Corbera et al. (2007), Pascual & Perrings (2007), Engel et al. (2008), Chen et al. (2009), Carpenter et al. (2009), Norgaard (2010) and Swallow et al. (2009). On the other hand it was revealed that the term “environmental services” is less clearly defined and there are barely two authors sharing the same definition. Furthermore some authors also use environmental services as synonymous to ecosystem services e.g. Myers (1996), Engel et al. (2008) and the FAO (2012).

As a consequence, the dissertation proposed a consistent meaning for both terms, connecting the definition to management strategies as proposed trough a debate in Germany where landowners receive money for conducting or omitting a specific land management. Usually such efforts of landowners are not regarded in definitions of environmental and ecosystem goods or services. In contrast “services” are usually defined as services which humans receive from nature but not as services by humans.

Therefore in contrast to the current definitions of PES, the dissertation gave center stage to the actor’s role terminologically, taking the term environmental service as a service by the landowner who provides an ecosystem service. Since humans can be rewarded for their actions and services, whereas “nature” cannot, the connotation of “Payments for ecosystem services” was found to be redundant.

The case study of Namibia’s semi-arid rangelands motivated the further research approach of the dissertation thesis. This ecological-economic system reveals multiple-stable states and faces an eventual regime shift and as a consequence the question arose about the optimal management strategies. The prevailing view in the literature is that a regime shift from a grass-dominated to a bush dominated system state - or basin of attraction - should be prevented in semi-arid grasslands (de Klerk 2004, Moleele et al. 2002, Smit 2004 and

Gil-Romera et al. 2010). Authors have proposed that a given state should be conserved to meet the needs of the present and future generations (Arrow 1995, Lebel et al. 2006). In a second step however, and with great implications on upcoming management strategies, the question arose if an alternative state might be sufficient as well.

It became apparent that the crucial assumption in the discussion regarding the resilience of ecological-economic system so far was that the system has exactly two different states: One which is desirable and one which is not. A system therefore is either in the desired state of the system and a society has to maintain it or it is not, and, given a high resilience of the system, society is locked in (e.g. Hanna et al. 1996). It is this scenario which often leads to the conclusion, that resilience is a precondition for sustainability (which is true, assuming, that the society is in a desired system state and there is no other state which fulfils the requirement of sustainability). This is also in line with other sources such as the shallow lake model (e.g. Scheffer 1997, Mäler et al. 2003, Martin 2004) or rangeland models (e.g. Perrings & Stern 2000, Anderies et al. 2002 and Janssen et al. 2004). In general here one simple management strategy is that a society might seek to maintain its actual desired state and prevent a regime shift. Or, in cases of an undesired initial state, tries to reach the desired state. Here the dissertation reason not much different. However, the most authors do not take into account the following possibilities: i) not only one basin of attraction might yield the desired level of utility and ii) due to the assumed system properties a certain, desired amount of natural capital can fall below a given sustainability threshold even within a basin of attraction.

To study the resilience of ecological-economic systems and the services they provide an ecological-economic model was designed, which features more than two domains of attraction. The occurrence of systems with multiple stable states has been widely discussed; e.g. Petraitis & Dudgeon (2004), Feng et al. (2012) and Nao & Akihiko (2014) and especially for marine and coral reef ecosystems e.g. Knowlton (2004), Takashina & Mougi (2014) and Cruz et al. (2014). A connection regarding the relationship to sustainability has not yet been conducted up to this point. Contributing to this research gap the dissertation analyzed the relationship of resilience as a property of ecological-economic systems and sustainability as a normative claim on the basis of an ecological-economic model. As a result in Chapter 4 the insight was revealed that the deduction from sustainability to resilience, or vice versa, is not possible in general.

This has implications for the management of ecological-economic systems with more than two domains of attraction, since the claim for resilience of a single state is not decisive for the management of ecological-economic systems but the properties of the whole system. In particular: (1) the current state of the system, (2) the domains of attraction of the system,

(3) the sustainability norm and the associated sustainability set in state space, and (4) the potential extent of disturbance.

Taking these insights as a basis, the dissertation derived that two following questions for management advice are possible to decide whether an actual system state should be prevented or if a system shift can be allowed or should be facilitated:

1. Do any other basins of attraction exist which can provide the desired ecosystem service in the same quantity?

2. Can the ecosystem service be substituted by another service within another basin of attraction?

If one question can be answered positively, a system change is possible and in line with a claim of sustainability of a society. However, the question remains in case of maintaining the actual state: Does the cost of maintaining outperform the necessary adaptation costs within a new basin of attraction? If both questions 1) and 2) are answered negatively the actual system state should be maintained. Clearly these questions are not to be answered easily but might be leading questions for an application of these theoretical insights within further research (see Section 7.3, Further Research).

Since the maintenance of a whole ecosystem requires super-ordinated actions in many cases, policy instruments might be implemented. As for their success they are heavily dependent on the reaction and feedback of the ecological-economic system. Therefore the dissertation studied impacts of external factors which change the stability landscape of ecological-economic systems. Based on an analytical model it has been derived that the stability landscape of ecological-economic systems is by no means immutable: People are affected by changes in ecosystem services, but also do humans modify these services, the corresponding ecosystem and its dynamic through the decisions they make and the management actions they take. Thus, the stability landscape of the systems depends on socio-economical circumstances together, but with distinction to ecological feedbacks. Up to this point the literature has primarily focused on changes in systems' stability, which are initiated by changes of ecosystem interdependencies and biotic factors. There is substantial debate on the relationship between biodiversity and stability, with a focus on how biodiversity affects the stability of systems, see e.g. May (1973), Naeem & Li (1997), Tilman et al. (1998), Tilman & Downing (1994), McCann (2000), Ives & Carpenter (2007), Scheffer (2009) and Naeem et al. (2012), to name a few. Although the discussion is broad it is generally agreed upon not only that species' diversity is crucial for system stability (Naeem et al. 1994), but primarily for their composition and connectivity.

The relationship between the resilience of a system and its biodiversity is discussed as another aspect. A higher level of biodiversity is usually thought of as equivalent with higher resilience (Mageau et al. 1995). Connected to this debate is the theory of the co-evolution of systems. Norgaard (1994) describes how the change of ecological and social systems can be understood in their connectivity, and how the change in one system affects other systems and subsystems. Also Levin et al. (1998) point out that the resilience of a system is affected by institutional circumstances since: "[...] the resilience of social systems in turn depends on a range of institutional and other properties" (Levin et al. 1998: 221). Another pioneer of this perspective has been Holling (1978) who emphasizes the impact of feedbacks and changing dynamics of ecosystems in his approach of "adaptive management". Here, control mechanisms and instruments have to be seen as merely part of an experiment and have to be adjusted necessarily and continuously to meet the properties of the changing system (Gunderson et al. 1995). The answer to the question of how one might exactly influence systems stability and make such behavior beneficial for an efficient conservation contracting requires further research (see Section 7.3, Further Research).

As already discussed instruments such as regulatory or market-oriented measures can be used to provide ecosystem services by promoting special management strategies - especially if this is required in terms of the revealed preconditions of Chapter 4. Many advantages and disadvantages regarding different payment schemes and incentives have been sufficiently discussed within the literature (Ferraro & Kiss 2002, Matzdorf 2004, Gibbons et al. 2011 and Hampicke 2013). Within the literature those payment schemes are examined primarily within case studies considering the conservation of single species, such as birds, butterflies or considering habitats such as certain types of forests, fens or heath (Wunder 2005, Pagiola 2008, Asquith et al. 2008, Corbera et al. 2007 and Manzo-Delgado et al. 2014). Also considerations about policy implications and this new way to incentivize and reward environmental services are discussed (Engel et al. 2008, Bulte et al. 2008, Clements et al. 2010, Fauzi & Anna 2013, and Greiner & Stanley 2013). General studies at the model and conceptual level on the other hand are scarce.

As an exception Zabel & Roe (2009) analyzed the concept of performance-based contracts under uncertainty with the help of a principal-agent model. They concluded that a base-payment is necessary to conduct a performance-based payment under uncertainty. Although this has been sufficiently discussed on a theoretical basis (e.g. Hampicke 2006 and Gibbons et al. 2011) this is one of the first examples which considered conservation contracts on a conceptual level.

The dissertation built on this current state of the literature with an extension of the model of Zabel & Roe (2009) i.e. with the assumption of information asymmetries between principal and agent, environmental uncertainty and a risk-averse regulator. The applied model originates from the general contract theory of Bolton & Dewatripont (2005) who discuss several forms of contracting, e.g. with or without uncertainty or information asymmetries (see Methodological Annex). While Zabel & Roe (2009) examined only a single payment scheme, i.e., a performance-based payment, the dissertation combined a performance-based and action based-payment for one payment scheme.

As a result it has been concluded within the dissertation that an exclusively performance-based payment is optimal only if there is no environmental uncertainty or if both the farmer and the regulator are risk-neutral. An exclusively action-based payment is optimal only if the regulator has full information about the productivity of the action i.e. if there is no information asymmetry. A main contribution of the dissertation was the insight that in every other case a combination of performance-based and action-based payments (with different weightings) may increase welfare. This also gives way for further management recommendations, especially with respect to the dynamics of ecological-economic systems and the desired services derived from these systems.

Taken together, the results of the dissertation lead to the following management implications: If the conservation objective is dependent on its basin of attraction, as discussed, it is also dependent on the resilience of that state. High resilience corresponds to low environmental uncertainty and it is therefore, taken the other way around, nearly certain that a system does not flip into another basin of attraction i.e. an undesired stable state.

Since uncertainty matters for the design of conservation contracts the corresponding resilience of the ecological-economic system also does. With respect to a management perspective, given an initial state and desirability of a system state, four possibilities arise for a system with two basins of attraction, each in need of its own management strategy and contracting:

- Given a first situation a system is in a state of “high” resilience, i.e. it has a low probability to change into another domain of attraction after a disturbance, and the given state of the system is supporting the conservation objective. As a consequence, the uncertainty considering the probability of maintaining an already existent conservation objective is low. With low uncertainty of maintaining or achieving the conservation objective, expected utility respecting the payoff rises, the base payment decreases. As the relative shares of the conservation contract depend on environmental uncertainty, with decreasing uncertainty the share of an action-based payment decreases and the share of the performance-based payment increases.

- Secondly, the situation might occur that a system is within a state of low resilience, and the given state of the system is supporting the conservation objective. As a consequence it is unlikely that the existing conservation objective will be maintained. With high uncertainty of maintaining or achieving the conservation objective, expected utility respecting the payoff rises, the base payment decreases. As the relative shares of the conservation contract depend on environmental uncertainty, with increasing uncertainty the share of an action-based payment increases and the share of the performance-based payment decreases.

- Thirdly, a system might be within a state of high resilience and the given state of the system does not support the conservation objective. As a consequence it is almost certain that the desired state cannot be reached. As the relative shares of the conservation contract depends on environmental uncertainty, the share of an action-based payment increases and the share of the performance-based payment decreases in this case.

- In the fourth and final case, a system might be within a state of low resilience, i.e. it has a high probability to change into another domain of attraction after a disturbance. The given state of the system does not support the conservation objective. As a consequence the probability to change the given state of the system and to reach the desired state of the system is high, uncertainty is low. With increasing certainty the share of an action-based payment decreases and the share of the performance-based payment increases for an optimal conservation contract.

Since those connections are not regarded sufficiently for the design of policy instruments current conservation contracts might lead to welfare losses.

To sum up the discussion it can be stated: Resilience was identified as a leading criterion for the design of instruments and the derivation of management strategies featuring ecosystem goods and services, but not as a goal in itself. Ecology relations such as ecological interdependencies and economic relations such as preferences and management regimes, change the systems' stability and therefore the resilience and the corresponding probability of a regime shift. Since uncertainty matters for the design of conservation contracts so does the resilience of the corresponding system state whereby each situation of combined risk and desirability of the initial state asks for its own management strategy. As this holds true within a framework for a system with two basins of attraction further research is needed for cases of systems with multiple stable states (see Further research).

7.2 A critical appraisal of methods

Some essential insights of the dissertation are derived through analytical models. Within the general field of analytical modeling the concern is often raised that there is a tendency to use models as a means of legitimizing rather than informing policy decisions. The same holds true regarding the subject of ecological-economic models. For example Robinson states that "[b]y cloaking a policy decision in the ostensibly neutral aura of scientific forecasting, policy makers can deflect attention from the normative nature of that decision [...]" (Robinson 1992: 148). It is needless to say that those tendencies would be difficult to prove; however this cannot be a general statement on the methodology of ecological-economic modeling. Instead it should be the responsibility of science and a claim of best practice to make assumptions explicit and indicate the applicability of the results. In this sense it should be indicated once more that also within this dissertation the model assumptions constitute the obtained results.

Complementing the previous chapters of the dissertation the applied methods and model assumptions are reflected and commented here. The central focus of the dissertation was neither a methodological discussion nor a qualitative or quantitative inquiry and although some of the case studies of the thesis comprise empirical considerations and results, the main research was on a theoretical and conceptual level. Therefore different methods and model types have been used, depending on the research question of each Chapter as introduced in the following. Given the broad range of methods the interdisciplinary character of the dissertation becomes apparent. For a further detailed explanation of limitations and assumptions there will be a focus on the applied analytical ecological-economic model and the principal-agent model.

By means of creating analytical models science aims at simplicity and reduction of the real world while looking for boundaries that minimize interaction. However as Costanza et al. (1993) indicate: "The interactions between ecological and economic systems are many and strong. So, splitting the world into separate systems is a poor choice of boundary" (Costanza et al. 1993: 545). Thus, the focus within the field of environmental and resource economics is on investigating the interdependencies of economy and ecology. Therefore "ecological-economic" models are used here which are characterized through assumptions on both economic relations e.g. the management regime such as profit-maximizing harvesting firms, and determinants of the ecosystem such as the intrinsic growth rates of the resources in question.

Evidently, no model is "right" within an entire range of uses but only in the context of a specific purpose. Model usefulness might be judged best by its ability to help to solve a certain problem, in this case to clarify the relationship between resilience and sustainability in ecological-economic systems. To address the research question and clarify the argument of Chapter 4 it was the explicit aim of the model to create a system with the characteristic of multiple stable states. Therefore model assumptions which feature especially those model characteristics have been chosen despite other characteristics.

The aim of the analytical ecological-economic model here was to achieve not high realism or precision but to address basic questions about the functioning of systems with multiple stable states. Therefore the ecological-economic models of Chapters 4 and 5 can be characterized as conceptual models, aiming at high generality but less realism or precision. The relationships of the management regime are simplified and highly stylized by the assumptions of e.g. open access and without extraction cost, perfect substitutability of the ecosystem services and where households' utility only depends on the given goods and services. Also with Chapter 4 the center of attention was not how the resilience of a basin of attraction exactly has been or to what extent a system has to be disturbed to change its basin of attraction. Therefore those parameters have not been quantified within the model.

As intended, the system featured multiple basins of attraction which enabled a broader discussion and generated a comprehensive picture of the relationship between resilience of ecological-economic systems and sustainability. The crucial assumptions for the dynamic of the system have been that there was no interaction between the natural capital stocks and that the resources have been compliments. These assumptions have then been widened and partly relaxed within Chapter 5, to reveal the dependencies of those assumptions and their impact on systems' stability. As for the logistic growth function for both biological resources the crucial property was that the intrinsic growth rate was bounded as the stock declines to zero. The crucial property for the utility function was the complementarity between the two ecosystem services and the substitutability of aggregate ecosystem services by manufactured goods. With a variation of the complementarity assumption the systems' stability changes, i.e. with complementarity between the ecosystems services the resilience of the basin of attraction, where both species are in existence, decreases.

Taken together the dynamics of this special model setting have been discussed thoroughly. With Chapter 4 general insights about the probabilities of systems with multiple basins of attraction and its relationship to sustainability have been achieved. With the variation of the different assumptions as debated in Chapter 5 the systems dynamic has been debated in detail, giving the results a much broader validity.

Despite the assumptions and discussion of well established functions other assumptions direct the results of the model: The definitions of resilience and sustainability. As for resilience, the definition of Holling & Gunderson (2002) on the basis of Holling (1973) was essential throughout the whole dissertation. For the definition of resilience in Chapter 4, the resilience of a specific basin of attraction could only be declared as resilient ex post, i.e. after a disturbance took place. If a system remains in its basin of attraction a system's state can be declared as resilient, if not, the system has been not resilient due to this disturbance. However generalized statements for other disturbances are not possible. With respect to this approach no management advice is possible ex ante, but also not intended within Chapter 4. Therefore the given definition of resilience served its purpose. Consequently, for a management perspective within Chapter 5, additional assumptions are necessary to imply management recommendations and estimate the resilience of a basin of attraction before a disturbance took place and independent of a specific disturbance. For this purpose it is necessary to quantify resilience to assess the probability of a given system state to change into another basin of attraction, a task which is not easily conducted. As described within the Introduction resilience is not measured directly and quantitatively here. Instead the resilience of the ecological-economic system is said to decrease as the distance between corresponding stability basins decreases, which increases the systems' number of alternative basins of attraction.

The paradigm of resilience was an interesting but intricate research field. Many ideas and principals are based on mathematical findings which have been adopted to real ecosystems in retrospect. Different definitions of resilience have applications within different research areas. At this point it might be indicated that resilience has been understood as a property of ecological-economic systems and not, as it might be within other approaches, as a normative aspect. Therefore for that reason alone it would be not possible to conclude from a specific characteristic whether it is desirable or not, this would be a natural fallacy. As this is a classical examination within philosophical theory one can reflect upon the additional value of an ecological-economic model, but it might be bridging disciplines.

The terms sustainability and sustainable development as used within the dissertation are closely related to the approach of sustainability based on and in analysis of the concept of Ott & Döring (2004) and modified for the purpose of the research question. Regarding the discussion about weak and strong sustainability which the dissertation calls upon, the model of Chapter 4 "only" assumed weak sustainability. Fulfilling the claim of sustainability a given level of aggregate wealth or welfare has to be maintained within the model setting, a claim which is usually associated with weak sustainability. Here, a minimum level of well-being is sustained for all generations, with ecosystem services and manufactured goods as substitutes (Asheim & Brekke 2002). "Strong" sustainability however would require the maintenance of a minimum level of critical natural capital for a future development.

However, the minimum-level of well being as demanded within the dissertation will be only achieved if both resources are maintained, since complementarity of the resources has been assumed. Therefore in this case "weak'" sustainability comprises the claim for "strong" sustainability i.e. that critical natural capital has to be maintained.

As to study the design of conservation contracts the dissertation drawed upon the methodology of the so called principal-agent models (see Methodological Annex for further details). As usual two contract parties have been distinguished for the principal-agent model in Chapter 6: One principal who offers a contract and one agent, to whom the contract is offered. As usual the two contract parties have been assumed to be rational individuals i.e. they estimate the given possible alternatives regarding the highest expected utility and each aim to achieve the highest possible payoff. Since principal-agent models are quite common within labor economics or contract design of insurance, contracts for providing a special ecosystem service have not been studied in such a formal context in the majority of cases. With the help of contract theory and principal-agent models it was possible to formalize the relationships and requirements of conservation contracts which have been scarcely addressed on a conceptual level. Since it was first necessary to transfer these established model types to a context of conservation contracts the model started rather simply. Therefore many variations and enhancements are possible (see Section 7.3, Further research).

For a typical conservation contract the agent was assumed to be better informed about the best action to choose to reach a given conservation objective. Therefore as one core concept, asymmetric information was assumed within the contract model. Two types of asymmetric information are usually distinguished, while only the second has been regarded within the dissertation: First the principal has no perfect knowledge about the characteristics of the agent and secondly the principal is not perfectly informed about the actions of the agent. Asymmetric information with regard to the type of the agent might lead to adverse section. This problem is often addressed within a context of insurance contracts and labor economics (Siemens & Kosfeld 2014, Chiappori & Salanié 2000 and Cohen & Siegelman 2010) but has not been regarded here; since the agent was assumed to be a representative farmer i.e. all farmers share the same characteristics. Asymmetric information regarding the information of the agent might lead to problems of moral hazard if the agent has different objectives than the principal. Thus, to improve efficiency the literature has considered how to deal with cheating by the agent and ensure compliance, so contracts to fight moral hazard are designed. Some authors discuss windfall effects, arguing that e.g. actions which already have been performed before the payment took place should not be rewarded. On the other hand, those payments might ensure that actions which provide a certain ecosystem service will also be conducted in the future. Other voices also appreciate those effects as producer surpluses (Hampicke 2006). However in the dissertations setting it was not necessary that the agent conducted a certain action but chose the actions himself and is rewarded afterwards due to expenditures. Therefore compliance has not been considered. There is another aspect which differentiates the contract model to other current approaches: Since the focus was on utility maximization on the part of the principal in contrast to cost efficiency, no budget restrictions have been assumed. For the discussion of efficient contracting the comprehensive studies of e.g. Drechsler et al. (2007) and Wätzold et al. (2008) might be recommended.

It was further assumed that the principal has no preferences about a specific action alternative but is exclusively interested in the outcome i.e. the performance of the conservation objective, which might not reflect reality. Notice also that in case of a risk-averse regulator this aversion is against fluctuations in environmental goods provision. Thus, his utility is solely dependent on the performance of the ecosystem state minus the payment for the agent. Within a first instance only one principal and one landowner have been assumed for model simplicity acting in a bilateral contracting situation and without other parties as externalities. An extension of the model setting which considers more than one farmer or groups of actors will be interesting and is recommended for further research. Also bargaining was not assumed within the thesis. Another shortcoming of the model is that the model assumed perfect knowledge of marginal productivity of the conducted action for the part of the agent. This assumption might be rather strong but simplifies the analysis. Besides, the crucial aspect of this assumption is that the farmer has better information about his contribution to the provision of the environmental good, i.e. marginal productivity, than the regulator. Due to environmental uncertainty, incomplete information about the production is faced by the farmer and the regulator to an equal extent.

As within the literature concerning payments for ecosystem services the best way to cope with information asymmetries has been found within performance-based payments. However, as considered within the dissertation, the risk burden of the agent often is opposed to a pure performance-based payment scheme. As a main achievement and innovation of the dissertation, a combination of different payment schemes was applied. Therefore a model with both environmental uncertainty for the agent and the principal and, regarding the principal, hidden information concerning the best possible action of the agent was chosen. Here a payment scheme was defined which on the one hand pays the agent for the actions he attended on a basis of his expenses. Additionally to that, a payment on the basis of the performance was offered. As a third component a base-payment was provided which compensates the agent's opportunity cost.

The implementation of a base-payment has been a much discussed point. In comparison with the current state of the literature a base-payment is not uncommon. Within their model Zabel & Roe (2009) assumed that a performance-payment is safeguarded by means of a base-payment. Otherwise with high uncertainty a contract on a voluntary basis would obviously be unlikely to be negotiated. Additionally, the combination of performance-based payments with a base-payment is also well established within practical applications, e.g. with the well known example of the “Ökoqualitätsverordnung” (ÖQV) in Switzerland. Within a first attempt for a combination of performance-based and action-based payments the first idea was to exclude a base-payment, since base-payments bear the suspicion of inefficient payout and improper welfare gains for the actor (Drechsler et al. 2007 and Hampicke 2006). There are also other ways to meet the participation constraint of the actor. One way might be to increase the action-based component of the contract to compensate for expected losses in the performance-based component. Another possibility might be to restrict the performance-based payment to be non-negative. However it is plausible to assume that these assumptions will lead to lower welfare: If the action-based payment would counterbalance expected losses they induce distortions in the payment structure since actions might be performed which gain a high payoff but are unrewarding. In addition, net growth of the environmental good may be negative even with a positive effort, which is an important impact regarding the assumption of a base payment as discussed below. On the other hand if performance-based payments are expected to be nonnegative the expected public expenditures associated with the PES scheme might be unnecessarily high.

Taken together the logic behind the assumption of a base-payment is the following here: A base-payment was assumed since it increases the flexibility and efficiency of the contract design. However, a base payment might be not necessary. In this case it will be zero. In every other case it will on the one hand counterbalance welfare gains or on the other hand compensate for performance losses. Therefore the base payment will be negative or positive, respectively.

7.3 Further Research

The dissertation is concerned with the management of ecological-economic systems and the desired ecosystem services they provide. The problem which arises, is as Berkes (2007) states, that “[...] systems are sufficiently complex that our knowledge of them and, our ability to predict their future dynamics, will never be complete” (Berkes 2007: 284). The resilience of a system’s state can be a guideline for the probability of how the system behaves while facing disturbances and offers significant insights of how environmental uncertainty can be overcome (Mäler 2008). Ecosystem services depend on such system dynamics thus their maintenance cannot be treated independently of these. Therefore for the efficient design of policy instruments such as conservation contracts, it is highly advisable to regard those dynamics. Although this position has been underlined and made plausible during the research of the dissertation, some questions remain for further research.

To begin with, the question of how the resilience of a system state should be linked exactly to the design of conservation contracts is not finally answered. The question is how the level of resilience affects the choice for an optimal combination of payment schemes. Some results have been given to this subject within Section 7.2 for a system with two basins of attraction and with the assumption, that one state of the system is considered to be desired because of its ecosystem services provision, and one is not. Obviously, for systems with two basins of attraction with either of the states being desirable or undesirable, there would be different management implications. For the first case with two desirable basins of attraction, resilience of one state would obviously not be decisive for sustainability since both cases would be beneficial. However society might decide that one state is more preferable. Potentially for the latter case in economical terms, the given resources could be used and exploited, since no desired state can be reached with a management intervention.

However, the interesting point for further research will be to broaden the previous assumptions and consider systems with multiple stable states. Drawing on the insights of Chapters 4 and 5 of the dissertation, the question arises which consequences occur if more than one desired state of a system is possible i.e. a system with more than one basin of attraction which supports the conservation objective. Obviously management advice here would also be based upon the knowledge of the probability that a system would flip into another desired basin of attraction, or, with a higher probability, would flip into an undesired state. Also here it would be necessary to take more than just the knowledge about the actual given system state into account and management advice has to be based on the knowledge about the system as a whole.

Regarding the design of conservation contracts, many extensions would be interesting. Within the model of Chapter 6 perfect knowledge of the actions' impact was assumed. As a modification it would be interesting to design a model with imperfect knowledge about the marginal productivity of actions. Furthermore a contract could be designed with more than one time period. In this case the actor would have the chance to learn how his management actions influence and direct the conservation objective. Thus, the increase (and decrease) of a conservation objective within a first period gives a hint for further necessary actions and reduces uncertainty, in the case of imperfect knowledge.

Another extension concerns the possible negotiation of contracts. Up to this point a contract on a take-it or leave basis which is offered by the principal was assumed. Bargaining might be a possible and interesting option if a specific area has to be involved to reach a conservation objective and a principal might not be sure about the offer he has to make.

Also, up to this point only one actor and one principal have been assumed as a necessary simplification to gain basic insights of the main driving forces of the model. Naturally it would be interesting and necessary to conceptualize a framework of contract designs with more than one agent, as it is widely done in the field of network analysis. Here first of all the interdependencies between single actors or groups are analyzed. Also the spatial dimension in compensation schemes, particularly of mobile ecosystem goods such as birds or butterflies, have been widely discussed e.g. based on the idea of an agglomeration bonus where land-owners only receive payments if managed patches are arranged in a specific spatial configuration (Drechsler et al. 2007 and Wätzold et al. 2008). However a study which discusses group behavior regarding the specific dynamics of the relevant ecological-economic system has not yet been conducted. Also, assuming a contract with bargaining, an agent could negotiate new and adjusted contract contents based on the gained insights for a second period. With these insights adjustments concerning environmental uncertainty and feedbacks of the ecosystem would be possible and might be regarded within a new contract design.

If more than one actor is assumed, the field of auctions for conservation contracts gains high importance. While auction and auction theory is a renowned and well explored field of research, some aspects regarding conservation contracts remain for further investigation. First of all within auction theory it is usually assumed that the bidder has perfect information about their costs and can therefore align his bid in an appropriate way. However the costs of maintaining or enhancing a conservation objective are usually uncertain given the assumption of imperfect knowledge about marginal productivity. An interesting research question for auction theory will be to analyze how this uncertainty influences the bidding behavior of the landowners: On the one hand they might choose to submit a lower bid to be awarded the contract; on the other hand uncertainty about the production costs will promote higher bid submission.

In addition to this, a last research recommendation regards the insight that a management regime significantly changes the system's resilience. Here the question remains: How to influence a system in a way such that a state which is desired, and at the same time resilient, could be reached or maintained in the long run? For this purpose the model of Chapter 6 might be adjusted and extended by considering that the agent's expected utility changes with respect to the resilience of the system and the desired conservation objective. As for the production function an interesting adjustment might be that the conservation objective would be directly dependent on the system's resilience, which then affects the actor's expected utility and therefore her behavior and choices.

Furthermore, it might be interesting to transfer and apply the obtained model results of Chapter 6 and its extensions to a real landscape and research area. However this will probably require long term studies and monitoring of both ecological development and management of the ecological-economic system, followed by a comparison between systems with different stability landscapes. This last research question is likely to be the most extensive and eventually the most important, and will be sufficient to provide interesting research work and decisive implications for the design of conservation contracts and the management of ecological-economic systems for a long time to come.

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Chapter 8

Methodological Annex - Contract Theory

8.1 Introduction

For more than three decades the theory of contracts and incentives has received vast attention within labor economics, organization theory, and corporate finance. In 1973 Ross introduced the study of agency in terms of problems of compensation contracting. Here, in essence, agency was seen as an incentive problem originating in society and not, as the existing stream of research considered, as a merely theory of the firm (e.g. Williamson 1964, Alchian & Demsetz 1972). Simultaneously Mitnick (1975) introduced the insight that institutions form around agency, originating the institutional theory of agency. Today, the most applied principal-agent models and the contract theory provide "[...] a rich research basis for selecting an appropriate contractual form, with an emphasis on the effects of uncertainty and asymmetric information" as Hooper (2008) points out. The assumption of information asymmetries as an extension and improvement of general equilibrium models has been crucial for the emergence of the principle-agent theory and the economics of contracts, as Salanié (1997: 2) indicates: "It is fairly straightforward to extend the general equilibrium model to cover uncertainty as long as information stays symmetric. Unfortunately, asymmetries of information are pervasive in economic relationships [...]. The theory of contracts originates in these failures of general equilibrium theory".

As explained within Chapter 7, a principal-agent model was chosen to investigate general mechanisms of contract relations between principal and agent. This course of action was chosen based on the insight, that contracts plays a major not only within firms and employment relationships but also considering policy instruments regarding the provision of ecosystem services. Due to spatial limitations it has not been possible within each Chapter to go into detail for all applied model types and methodology. Therefore additional information is provided here for the broad field of contract theory with a special emphasis on principal-agent models.

8.2 Theoretical background and some basic models

The fundamental activity of every production or allocation process in economics is the exchange of goods and services. As Ross (1973: 134) noted "agency relationship has arisen between two (or more) parties when one, designated as the agent, acts for, on behalf of, or as representative for the other, designated the principal, in a particular domain of decision problems [. . .]".

The starting point of incentive theory therefore corresponds to the problem of delegating a task to an agent with private information. This raises the problem of managing information flows. As Arrow (1964) points out, the agent has been selected for his better knowledge concerning a task and the principal can never hope to completely check the agent's performance. Incentive theory considers when this private information is a problem for the principal.

The theory of incentives and contracts emerges with the exchange of labor for money, and questions about their optimal allocation. A classic example of Microeconomic Theory for a simple exchange situation is the well known "Edgeworth box": Two actors trade two commodities in a given place and point in time. The idea of "state-contingent" commodities, where the output is dependent on a specific, future state (Arrow 1964 and Debreu 1959), together with the formulation of a theory of "choice under uncertainty" by Neumann & Morgenstern (1944), has enabled more complex exchange situations to be considered, such as trading in different points in time and risk sharing of default and non-delivery.

In general, contract theory and incentive theory can also be addresses as a branch of game theory (Salanié 1997). However, the distribution of information is different in contrast to most games: In the classical Prisoner's Dilemma for example, each actor know as much or as little as the other. On the other hand, a main assumption within principal-agent models is the lack of information on one side about the character or action while this information is crucial for the contracts' outcome.

Principal-agent games can therefore be understood as a special kind of game, assuming one informed and one uninformed party. This disequilibrium of information endowment is mostly described as information asymmetry (see e.g. Salanié 1997). Considering this, in general two forms of information asymmetry can be distinguished:

1. lack of knowledge concerning the type of agent and

2. lack of knowledge concerning the performed actions of the agent.

For the first problem, models of “adverse selection” have emerged, discussing several forms of best practice, efficient contracting and strategies of how to reveal the agents’ type. Here, within so called “signaling games”, the revealed preferences of an agent are interpreted as “signals” to detect her actual type. Within the second group of contract games, where the actions of the agent are not fully known, contracts of “hidden action” are concerned with the problem of “moral hazard” as described below.

To further this basic differentiation, models of contract theory can be expanded in several dimensions: Regarding the related research question, they can be static or dynamic, complete or incomplete, with hidden information or hidden actions, with and without uncertainty, containing two or more agents or parties, and with bilateral or multilateral contracting[52]. To outline the basic ideas of contract theory and principal-agent models, the description has been limited here to a basic model, first without moral hazard followed by an example of adverse selection and an extension of the hidden action problem as applied within the dissertation.

A basic example of optimal contracting

A basic bilateral contract model without uncertainty and moral hazard usually assumes the following situation: An informed party meets an uninformed party whose information is relevant to them. A contract is negotiated either with bargaining or on a take-it-or-leave-it basis. The so called principal-agent models usually assume the latter, given the bargaining power to one party. The principal, usually the uninformed party, offers the contract to an agent. Interaction would stop if the agent rejects the offered contract.

As an example, a standard situation might be a contracting problem between an employer as principal and an employee as an agent. The employee has an initial amount of time which she can either keep for herself or offer to sell to perform a specific task[53]. The employer on the other hand needs some labor to be conducted

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[52]Further information about this branch of research gives Salanié (1997), Laffont & Martimort (2002) and Bolton & Dewatripont (2005).

[53]Within principal-agent models and contract theory the agent is usually denoted as “she” whereas a principal is denoted as “he”.

and might do this by himself or offer others money and thereby delegate his task. So in the initial case and without trade or contracting, the agent has all the time t, whereas the principal has all the money q. This initial endowment can be denoted as $(t_1, q_1) = (0, 1)$ for the principal and $(t_2, q_2) = (1, 0)$ for the agent. Therefore the corresponding utility functions of principal and agent can be denoted as U(t, q) and u(t, q), with utility level $\bar{U} = U(0, 1)$ and $\bar{u} = u(1, 0)$ without trade.

If the employer can make productive use of the agent's time he might offer a contract. This is the case if both the utility function of the agent and the principal are strictly increasing in both arguments and strictly concave. In this case principal and agent might increase their joint payoff by exchanging time for money. The question which arises therefore concerns the optimal rate of exchange between money and time. This leads to the following maximization problem:

$$\max_{t_i, q_i} U(t_1, q_1) + \mu u(t_2, q_2) \quad \text{subject to} \tag{1}$$

$$t_1 + t_2 = 1 \quad \text{and}$$

$$q_1 + q_2 = 1 \ ,$$

since time and money are limited. Here, μ corresponds to the bargaining power of each individual as well as their reservation utility, t_i as the actual time consumed and q_i as the amount of output after trade, respectively. It follows that, without further assumptions joint surpluses are maximized if the marginal rates of substitution between money and time are equalized:

$$\frac{U_t}{U_q} = \frac{u_t}{u_q}.$$

Obviously, there are gains from trade as long as

$$\frac{U_t}{U_q} > \frac{u_t}{u_q}.$$

Therefore principal and agent will trade until their utility level without trade becomes more equal. However, assuming, that the principal offers a take-it-or-leave-it contract, as common within principal-agent models, bargaining is not a part of the game. Since the principal has a strong interest in the agent signing the contract, he has to match at least the agent's utility constraint with his contract offer such as:

Maximization problem of the principal:

$$\max_{t_1,q_1} U(t_1, q_1) \quad \text{subject to} \tag{2}$$

$$u(1 - t_1, 1 - q_1) \geq \bar{u}.$$

Assuming, that the agent will sign the contract and the principal has full knowledge about his utility constraint and the agent's actions, i.e. he can observe all actions or all outcomes, and the outcome directly represents the agent's action, the contract is optimal. Since the principal might now know the agent's utility constraint exactly, the contracts which are offered might not be efficient. For the possibility to reveal the agent's utility constraint, a framework of auctions for conservation contracts might be applicable (see e.g. Latacz-Lohmann & Schilizzi (2005, 2007) for a detailed discussion of conservation contract auctions). This revelation principle is also decisive for contracting under asymmetric information of all kinds: However, it is not necessary to know the exact type of the agent but to make sure that an agent of a specific type "has an incentive to select only the contract that is destined to him/her" (Bolton & Dewatripont 2005: 16). In this sense first contracts of adverse selection are considered in the following sections.

A simple model with adverse selection

Bolton & Dewatripont (2005) consider contracts where the hidden information is the working skills of a prospective future employee, hereafter called agent for simplicity. The skills of the agent might not be known at the time of contracting to the principal. Therefore there might be two types of agents: One who is highly skilled, with a valued time of σ_H and one who is lowly skilled, with a value time hereafter denoted σ_L, with a value of time $\sigma_L < \sigma_H$. The employee knows whether she is skilled or unskilled whereas the principal only knows a probability facing a skilled employee of p_H. The principal now has to make sure that his offered contract is incentive compatible, i.e. each agent picks the contract which is suitable for her. Therefore in general, principal-agent models with imperfect knowledge can be described as a "Stackelberg" game where the principal moves first, but chooses his offer with anticipation to the agent's expected response[54].

Let's assume on the basis of the basic contracting model that the agent's utility function is $u(\sigma t + q)$ and the principal's utility function is $U[\lambda\sigma(1-t)]$,

- with $(1-t)$ as the time sold to the principal, and t as the time she keeps for herself.
- q as the output received
- λ as a positive constant; and
- σ as the skill level of the agent.

For incentive compatibility of an offered contract, the contracts have to be designed, such that type σ_H must prefer a contract (q_H, t_H) over (q_L, t_L) and type L a contract (q_L, t_L) over (q_H, t_H). An optimal menu of contracts under hidden information then can be found by:

$$\max_{t_j, q_j} p_L U[\lambda\sigma_L(1-t_L) - q_L] + p_H U[\lambda\sigma_H(1-t_H) - q_H] \tag{3}$$

$$\text{subject to} u(t_L\sigma_L + q_L) \geq u(\sigma_L) \quad \text{and}$$

$$u(t_H\sigma_H + q_H) \geq u(\sigma_H).$$

Also, two incentive constraints have to be added:

$$u(t_H\sigma_H + q_H) \geq u(t_L\sigma_H + q_L) \quad \text{and} \tag{4}$$

$$u(t_L\sigma_L + q_L) \geq u(t_H\sigma_L + q_H). \tag{5}$$

In general, contracts under incentive constraints will be second-best contracts, which do not achieve optimal allocative and distributive efficiency. The trade of between the extraction of information and an efficient allocation is therefore much discussed (Bolton & Dewatripont 2005).

[54] Stackelberg games denote models of imperfect competition based on a non-cooperative game strategy (Stackelberg 1934).

A model with moral hazard

Turning now to the case that information asymmetry is present, i.e. that the principal is not fully informed about the agent's actions. In the former section information asymmetry arises *before* the contract was signed, i.e. the type of the agent was not revealed. In this instance information asymmetry arises *after* the contract was signed, by the action the agent chooses.

In this setting moral hazard of the agent might occur, since the principal might not be able to observe all the actions of the agent, or observation is too costly. If he can observe the outcome then the case might be interpreted that the outcome represents with some probability the agent's action. A contract therefore can be described as a contract under "hidden action" or "hidden information". Note, however, that in such settings it is implicitly assumed, that the principal knows about the best possible action the agent should perform. However, the difficulty arises in monitoring the act that the agent chooses, especially if agents are numerous. This problem was examined by Spence & Zeckhauser (1971) in the case of insurance. While it might be principally possible to observe all the agent's actions it would not be economically viable to do so (Ross 1973). Therefore within a first line of research regarding this contract setting, the question is how moral hazard can be eliminated or at least reduced by choosing the "right" incentives and contract settings.

To introduce hidden information assume that the amount of time $(1 - t)$, which the agent works is her private information. Then the output for the employer is σ, with a probability function $p(1-t)$ increasing in $1-t$, given that the amount of work is positively correlated with the amount of output given the probability σ. However, "nature" can be good or bad, such that the outcome of the work is state-contingent, i.e. if "nature" is positive, say work creates the output σ_H, whereas if a "bad nature" occurs, output only increases to σ_L.[55] Thus a probability function denoting the outcome might be:

$p_H[1 - t]$ for a "good nature" or as within the former example for a highly skilled agent, and $p_L[1 - t] = 1 - p_H[1 - t]$ in case of a "bad nature" or lowly skilled agent.

Since effort is not observable the contract payoff might be dependent on the realized outcome σ_j. Since the principal must expect that $(1 - t)$ will be chosen by the agent

[55] Note that if output might rise deterministically with effort, the unobservability of effort would not matter.

as to maximize her expected payoff under the outcome-contingent payment scheme, the principal has to make sure that it is the agent's best interest to supply the right level of working time $(1-t)$. Therefore the principal must not only take into account the agent's rationality constraint but also her incentive constraint.

The agent's optimizations problem:

$$(1-t) \in \max_{\tilde{l}} p_L[1-t]u[q(\sigma_L)+t] + p_H[1-t]u[q(\sigma_H)+t]. \tag{6}$$

The principal optimization problem:

$$\max_{q(\sigma_i)}\{p_L[1-t]U[\sigma_L - t[\sigma_L] + p_H[1-t]U[\sigma_H - q(\sigma_H)]]\} \quad \text{subject to} \tag{7}$$

$$p_L[1-t]u[q(\sigma_L)+t)] + p_H[1-t]u[q(\sigma_H)+t] \geq \bar{u} = u(1) \quad \text{and}$$

$$(1-t) \in \max_{\tilde{l}}\{p_L[1-t]u[q(\sigma_L]+t] + p_H[1-t] + u[q(\sigma_H)+t]\}.$$

The compensation scheme typically will comprise a loss since the outcome is only a noisy signal of the agent's effort. Since effort is costly and the outcome is unpredictable, the principal has to insure the agent for her effort and the tradeoff between incentivizing a certain action and insuring against uncertainty remains. In general, the characterization of optimal contracts in the context of moral hazard is still limited (Bolton & Dewatripont 2005).

An assumption which makes the solution much simpler would be to assume a risk-neutral agent, where the agent bears all the risk of the expected outcome. However in most cases a common assumption is a risk-neutral principal. Here, the common explanation is that he is not dependent on the relationship to the agent and that he can diversify his risks. As for the agent on the other hand, it is more difficult to diversify her risks; therefore the usual assumption is risk-aversion on behalf of the agent. As within the chosen principal-agent model of the dissertation, in a first instance the agent was assumed as risk-averse, the principal as risk-neutral. In a second instance both principal and agent have been assumed to be risk-averse. For a risk-averse agent the situation becomes more complicated: Very few general results can be derived about the form of optimal contracting with moral hazard and each asks for more special assumptions. A common, applicable assumption is for one, that it might be assumed that only two outcome states would be possible, or that the performance is normally distributed together with constant absolute risk-averse preferences for the agent and a linear incentive contract.

8.3 Multi-agent games and the problem of contract enforcement

The consideration of multi-agent games and interactions between larger groups of actors is another main focus of game theory and the economics of contracts (Demski & Sappington 1984). As already indicated, this topic, in its simplest form, is related to the model family of adverse selection with multiple but heterogeneous actors. Of course, multi-agent contracts can be also studied as repeated (Abreu et al. 1991) and/or dynamic games (Laffont & Tirole 1988), or e.g. with intertemporal incentives (Holmström & Milgrom 1987), leading to an even more complex contract setting. Also, in relation to the field of multi-agent games, one might referred to the broad field of auction theory, as already mentioned[56]. In a nutshell, auction theory aims to search for the best suitable agent for a specific task, as the agent reveals her characteristics through her bidding.

Further on, if contracts are not negotiated between an institution and a single actor but between many actors, other research branches come into focus, namely the huge research field of Agent-Based Modeling (ABM)[57] and Social Network Analysis, starting e.g. with Kent (1978) and proceeding till today to e.g. Jackson & van den Nouweland (2002) and Jackson (2008)[58]. Contract relationships here are not only investigated regarding the strategies and actions between actors but also between actor groups and institutions. In general, ABMs and Social Network analysis often focus on the dynamics of a specific behavior (Epstein & Axtell 1996). Here, equilibria may be difficult to find or may not exist at all, and outcomes may be complex. However, there might still be patterns of behavior which agents may follow for optimization. Those rules can be searched for and tested empirically (Laver & Sergenti 2011).

As a final remark, it might be noted that contracts, as regarded within the thesis, are assumed to be complete and automatically enforced by a legal system, which is obviously not always given in reality. The enforcement and control of contracts relates to the field of Economic Governance, which recently was mostly coined by

[56]see e.g. Vickrey (1961), Milgrom & Weber (1982), Bulow & Roberts (1989), Milgrom (2004) for technical introductions, Maskin & Riley (2000) for asymmetric auctions, and Klemperer (2002) for the efficiency of auctions design.

[57]For a comprehensive introduction to Computational Sociology and Agent-Based Modeling see e.g. Macy & Willer (2002) and de Marchi & Page (2014).

[58]For a comprehensive introduction to Social Network Analysis see Wassermann & Faust (1994).

Elinor Ostrom and Oliver Wiliamson who shared the Nobel Memorial Prize in Economic Sciences in 2009. Wiliamson is mostly concerned with the theory of the firm and internal enforcements and relationships within a firms' environment (Wiliamson 1985, 1988, 2000), whereas Ostroms' main field of interest is the study of the management of common pool resources and collective action (Ostrom 1990, 1999, 2000, 2005). Both are interested in the contracts' "environment", i.e. under which circumstances and institutional settings contract enforcements and negotiations fail or work out (see also e.g. Milgrom et al. (1990), Aghion & Tirole (1997) or Dixit (1996, 2009)).

Taken as a whole, all this can be only a brief glimpse into the broad field of contract theory and the related fields of research, to which the thesis added a small epistemological gain regarding selected assumptions.

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