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PhD Dissertation 01/2009

Ecological-Economic Assessment of Biological Invasions – A Conceptual Contribution on the Basis of the Concept of Ecosystem Services

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00343954

ISSN 1860-0387

„Ecological-Economic Assessment of Biological Invasions“

A Conceptual Contribution on the Basis of the Concept of Ecosystem Services

Dissertation
zur Erlangung des akademischen Grades
doctor agriculturarum (Dr. agr.)

am Institut für Agrar- und Ernährungswissenschaften
der Naturwissenschaftlichen Fakultät III
der Martin-Luther-Universität Halle-Wittenberg
(Dekan: Prof. Dr. Peter Wyscisk)

vorgelegt von

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geb. am 10.03.1976 in Marbach am Neckar

Archiv

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Verteidigung am: 1. Dezember 2008

Acknowledgements

This thesis has been supported by several people, whom I would like to thank for a variety of reasons. First of all there are my supervisors Prof. Bernd Hansjürgens and Prof. Heinz Ahrens, who gave me the opportunity of writing my PhD under their supervision. Prof Hansjürgens gave me very valuable comments on the draft version of the thesis. In addition I would like to thank Dr. Stefan Klotz who agreed on becoming my third supervisor.

I want to thank also many of my colleagues at the UFZ: Among others Katja Sigel for good advice in the huge and misty field of uncertainty; Florian Eppink for providing me very honest and constructive suggestions and comments; but also Christoph Schröter-Schlaack, Felix Rauschmayer, Johannes Schiller, Irene Ring and Martin Quaas supported my thesis by reading different chapters very thoroughly. They all helped with valuable comments and good feed back on the economic aspects and the overall structure of the work.

I am also grateful to Ingolf Kühn, Daniel Kissling and Carsten Nesshoever, who discussed the ecological side of the topic with me, and thus brought forward an important part of the thesis.

Furthermore my work benefited from support of the following persons: Maren Mortensen, Colette deRoo, Sebastian Unger, Kathleen Schwerdtner, Marie Hanusch and Ina Gittel. Thanks for your help with proofreading and the specific assistance of each of you.

Finally, I would like to thank Norgard and Rhoda Born. I am deeply grateful for your moral support and the time spent on all the non-economic issues around the thesis. And last but not least the biggest thanks goes to my family Ingo Bräuer and Clara Born. Clara always provided me an endless source of alternatives to the thesis and with her birth opportunity costs became a new dimension. Ingo's patience, the countless discussions and his everlasting support and endless motivation made this thesis possible. Without him I would never have finished this work.

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1 Introduction

1.1 Problem background

The phenomenon of biological invasions has become widely known as a result of the damages that alien species cause in new habitats. Such damages do not only affect conservation issues, but are also of concern for human health and land use. Impacts of biological invasions include changes in nearly untouched ecosystems and their species composition, but also changes in the recreational value of landscapes. Effects on human health and damages to human-made infrastructure such as railroads or water pumps are further problems of the ecological issue. Due to the manifold impacts they generate, predominantly within environmental systems, biological invasions are a high-ranking issue in environmental policy. Very frequently they are mentioned as one of the key drivers of one of the most pressing environmental problems of the globalised world: the loss of biodiversity (Oecd 1996: 90; McNeely 2001: pp. 5).

To date, the Convention on Biological Diversity (CBD), the international framework convention from the UN Summit in Rio de Janeiro in 1992, has been forcefully calling for political action to counter this phenomenon (Cop 2002). The convention acknowledges the major involvement of humans in the invasion problem. Humans are main drivers for the dispersal of alien species. Public transport and travel are decisive factors that determine the number of species introduced unintentionally (Dalmazzone *et al.* 2000). Ballast water or package material is responsible for bringing foreign species unintentionally into new environments (Nentwig 2007: 15). As a deliberate driver, trade with exotic ornamental plants enlarges the pool of potentially invasive species (Kowarik 2003a). At the same time the CBD recognises that humans can combat the invasion problem by implementing well-aimed strategies. It provides a catalogue of possible management strategies, namely prevention, eradication and control of alien invasive species, and also demands a hierarchical application approach. This hierarchical application approach means that prevention ought to be preferred before eradication, and eradication before control. Each strategy covers several measures. Prevention as a strategy, for instance, can comprise measures of border control or the application of black lists for trade imports. Each strategy is suitable for a certain phase of the species invasion and promises an effective approach to reduce the invasive species' impacts on the environment.

The CBD is a political document with great relevance. It points the way for future political decisions: the issue of species invasion cannot be neglected on the political agenda, and appropriate decision-making is required. Although the convention regards the suitability of strategies, it does not provide the frame to answer the question of the appropriateness of the different management options, which is additionally demanded in Art. 8(h) of the CBD[1]. Appropriateness of a measure, however, is important for decision makers. They often face the problem of very cost-intensive management options of invasive species, in which the efficiency of the measures applied is important.

Economics deals with scarcity and choice (Perman *et al.* 1999: 104). It considers situations where resources or money is constrained and where costs and benefits of one action have to be weighed up against other alternatives. In fact such budget constraints dominate public decision-making every day, especially in the context of biological invasions´ management. By regarding costs and benefits of both, impacts of biological invasions and of management measures, economics can support decision-making concerning the appropriateness of a measure by revealing the ratio of costs and benefits. This decision-supporting role of economic assessments has been of growing concern for the last years (Scalera and Zaghi 2004: 15).

There are two fundamentally different economic approaches in the field of biological invasions. The first approach accounts for gross costs of biological invasions and records the overall costs. The study of Pimentel (2002), which estimates a global economic loss of $ 31 trillion due to invasive species' damages, is just one prominent example for this approach. Such studies aim to raise public awareness and to select the species with the highest gross-damage costs for initial action. The second approach provides information about the net benefits of conducted management options (for instance Zavaleta 2000; Barbier 2001). To this effect it considers the ratio of benefits of management measures and their costs, therefore delivering information about the appropriateness of a measure. This second approach corresponds to the demand of appropriateness of the CBD as it weighs up costs and benefits of biological invasions.

For this second approach of economic assessments there are several challenges (Touza *et al.* 2007: 354; Binimelis *et al.* 2007: 331). Firstly, it is difficult to cover all impacts

[1] Article 8(h) requires its parties to "prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats and species" (CBD 2001:5).

occasioned by an invasive species. If economic assessments aim to provide information in order to support decision-making concerning the appropriateness of measures, it has to cover all impacts due to invasion of relevance for society. Therefore, economic assessments face the problem of covering the multiple effects of biological invasions on human well-being. This has been often insufficiently considered in economic assessment studies, as they tend to only focus on single aspects of impacts while neglecting the not immediately apparent side-effects. Secondly, all impacts have to be economically valuated, which means to transfer damages of biological invasions and benefits of the management in monetary units. This is challenging due to the lack of market prices for biological invasions´ impacts. Thus, surrogate markets have to be established to conduct such an economic valuation which covers several challenges (see for instance Perman *et al.* 1999: pp. 388; Garrod and Willis 1999: pp. 4). Thirdly, the invasion problem is characterised by high levels of uncertainty. Damaging effects are always complex and the knowledge about ecological mechanisms is restricted (Kolar and Lodge 2001). Due to the complex structure of ecosystems and their processes, one major impact of an invasive species normally causes cascading effects and triggers changes in dependent ecosystem processes. The loss of biodiversity is the most prominent example for a serious impact of biological invasions, because the extinction of a certain species can cause a lot of cascading effects in the trophical and abiotic conditions of its former habitat. Therefore, decisions regarding the management of biological invasions are taken under uncertainty. Uncertainty, as synonym for incomplete knowledge, occurs in the context of predicting the outcome of the invasion process, i.e. whether, when, how and to what degree invasive species will damage the environment and human well-being. In addition, uncertainty also occurs when determining the success of management strategies. The success of a strategy is dependent on a conglomeration of factors, such as the susceptibility of an ecosystem, internal traits of the invasive species (e.g. the degree to which it is superior to the native species in habitat competition), and the applied measure.

Thus, (i) to cover the range of relevant impacts, (ii) to evaluate them economically, and (iii) the include uncertainty linked to the invasion problem are three key challenges when providing economic assessments for the support of decision-making. To date, the explicit consideration of two aspects relating to species invasion, namely to cover the range of impacts and to include uncertainty, is lacking in economic literature. In consequence, the following work addresses both subjects.

Economic assessments are understood as a procedural two-step process, comprising firstly a physical assessment[2], and secondly an economic valuation.

During the first step, the physical assessment, all features of the object, which in this case refer to biological invasions' impacts and suitable management strategies to address these impacts, have to be analysed. This procedure provides the framework for the assessment. The framework will be called "analysis structure of impacts", in the following referred to as "analysis structure"[3]. The physical assessment not only means to quantify impacts, but also to detect, characterise and document them.

In the second step, the economic evaluation, values have to be attributed to the object of assessment, which involves making judgements in which people assign values. Economic values are usually expressed in monetary terms. The economic evaluation is based on the physical assessment. The combination of both steps, the physical assessment and the attribution of values to the object of assessment, constitute the economic assessment.

Thus, to summarise from the above, there are two major problems. They affect:

1. The <u>physical assessment</u>, due to the need for an analysis structure that enables to cover all impacts that occur due to an invasive species,
2. The <u>economic valuation</u>, that has to economically evaluate impacts although no markets exist for the majority of impacts of biological invasions[4].

In addition there is a third problem that affects the whole economic assessment:

3. <u>Uncertainty</u>, as economic assessments have to include this inherent feature of

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[2] The term *physical assessment* used in this thesis has also been referred to as *environmental impact analysis* or *life cycle analysis* (Pearce *et al.* 2006: 56). The physical assessment is also part of cost-benefit analysis (CBA) and refers to impacts of a project or a policy (Turner *et al.* 1998: 2). As this thesis deals with a general understanding of economic assessments of biological invasions' impacts, to refer only to Cost-Benefit Analysis would be too narrow. CBA is only one tool for this approach, next to Cost-Effectiveness Analysis. Hence, the process of analysing impacts will be called physical assessment throughout this thesis.

[3] The physical assessment is defined as the procedure within which impacts are analysed. The framework of this procedure needs a different term. At the moment, however, there are no suitable terms for such a framework, except in German "*Mengengerüst*" (e.g. Hartje *et al.* 2001). As the framework is of central meaning in this thesis, the term *analysis structure* has been chosen because it provides the basic structure along which impacts (of biological invasions) are analysed.

[4] This stage is also referred to as monetarisation which relates to the fact that economics uses money as value format (see chapter 4.1.1). Since other formats of value can be also used, the broader term economic valuation has been chosen.

biological invasions in order to deliver sound assessment results.

A review of the literature on the subject evidences extensive research addressing the second mentioned problems: the economic valuation of biological invasions (for an overview see Born *et al.* (2005)). In these studies the application of, and corresponding problems related with, environmental economic valuation techniques are emphasised.

However, framework conditions as the very base for such an economic valuation have so far been neglected. This means that studies that address extensively the physical assessment of ecological and economic impacts of biological invasions are not at hand. These framework conditions derived from ecological considerations, however, verify the results of the whole economic assessment, as the economic valuation will be determined by the quality of the physical assessment. The overall broadness and the quality of the physical assessment conditions determine how economic assessments can be used for an effective decision aid, as it determines the scope of consideration, i.e. which impacts are to be evaluated. The economic valuation can only show related costs and benefits of those impacts which have been previously selected in the physical assessment to be the object of valuation.

1.2 Objective of the thesis

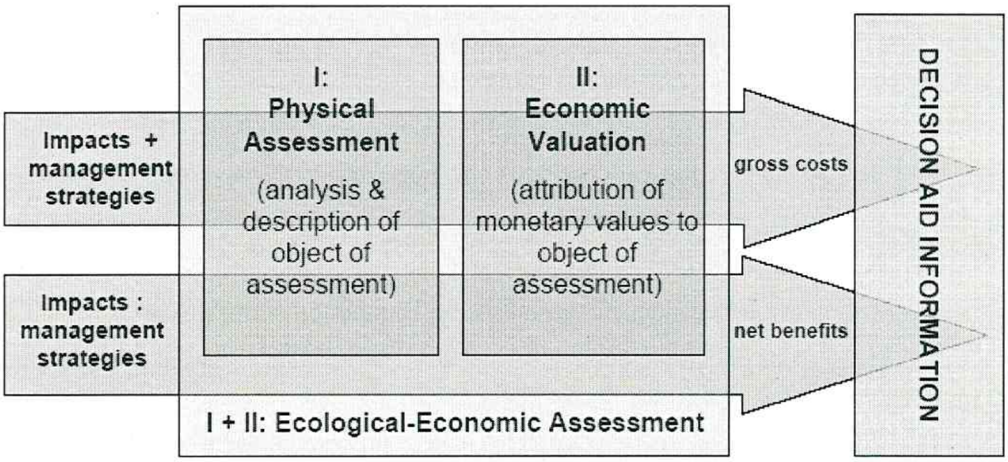
The aim of this thesis is to contribute to the methodology for economic assessments of biological invasions. While taking into account ecological and economic properties of the problem, the thesis considers both steps of economic assessments. By this it will be brought up that major challenges lie within the first step of the whole assessment process. Thus, the thesis concentrates on the physical assessment of ecological properties of biological invasions with the intention to enlarge the scope of economic assessments to real ecological-economic assessments (see Figure 1).

By considering biological invasions in all their complexity, it addresses two major problems for this first stage of ecological-economic assessments: (i) the scope of impacts that have to be covered by ecological-economic assessments, and (ii) the inclusion of uncertainty (as a synonym for incomplete knowledge)[5]. The first problem will be subsumed throughout this thesis as the *scope*-problem. It considers the challenge to encompass *all* impacts that occur due to biological invasions. The second problem is referred to as the *uncertainty*-problem,

[5] Although uncertainty is relevant for the whole assessment process, the focus will be on uncertainty during the physical assessment.

which arises in the context of impacts (when, how, and where do invasive species have effects on ecosystems and humans), but also regarding management options.

Figure 1: Ecological-economic assessments of impacts and management strategies of biological invasions, containing a physical assessment and an economic valuation (own source)

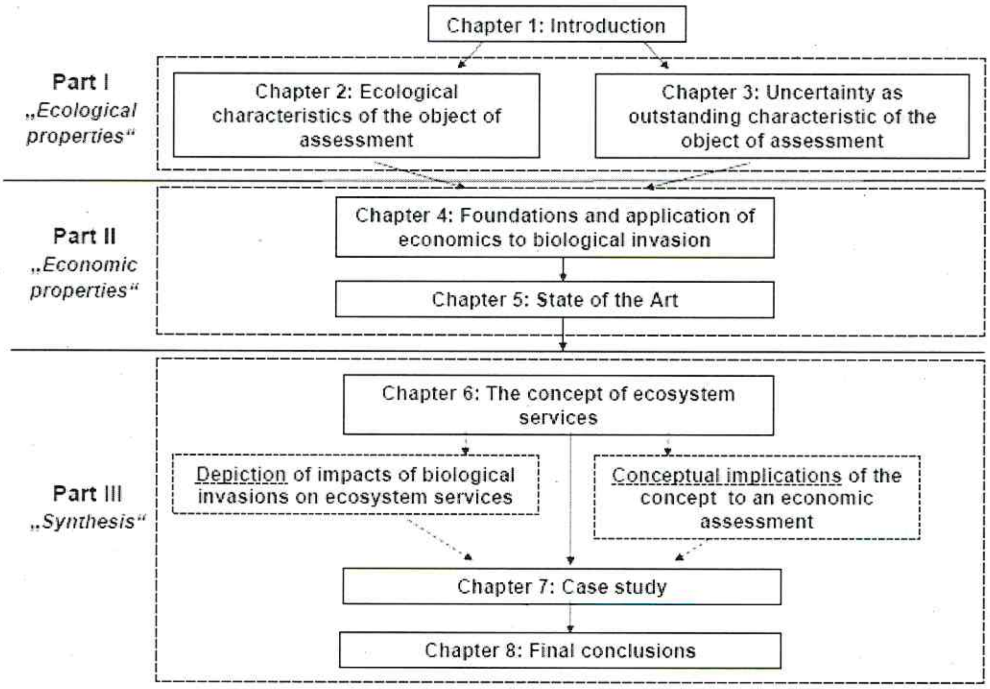


The work will end with a case study. This case study, however, serves to apply the knowledge acquired in the previous theoretical chapters and to illustrate the general conceptual contributions of the thesis to ecological-economic assessments. The whole thesis refrains from conducting a comprehensive empirical case study in which environmental economic evaluation methods are applied. Such studies are very case-specific and results are not easily transferred. An even stronger argument for a strongly restricted transfer of results from one assessment of an invasive species to another is given by the fact that each (potential) invasive species reacts differently to exposed environmental conditions and applied management strategies. By providing conceptual contributions to the physical assessment, this work intends to deliver an approach for an improved economic assessment of biological invasions as a whole. It points to the need for study of the stage of economic assessment which has been neglected so far.

1.3 Structure of the thesis

The thesis is divided into three major parts. Figure 2 illustrates the linkage of the chapters and the mere structure of three parts.

Figure 2: Structure of the work and linkage of chapters



Although these parts are not explicitly referred to throughout the work, they explain the different characteristics of the phenomenon of biological invasions. The first part (chapters 2 and 3) examines ecological properties biological invasions. It explains the invasion process and analyses uncertainty linked to the mechanisms that drive biological invasions. The second part (chapter 4 and 5) brings out economic properties of the invasion problem and introduces methods and tools for an economic valuation of such. Derived from the first and the second part, it becomes clear that biological invasions are a complex social and ecological phenomenon and that current economic assessments of invasive species' impacts fail to take into account the challenges mentioned above. These chapters point to the need that a general improvement of economic assessments of biological invasions should place emphasis on the physical assessment of it in order to become ecological-economic assessments. Therefore, the third part of this work (chapters 6-8) addresses the physical assessment stage. The problems of the scope of impacts and of uncertainty are relevant for any economic assessment of biological invasions, because taking these problems into consideration or failing to account for them seriously affects the results of ecological-economic assessments.

Within the third part the thesis, a conceptual improvement for both problems is proposed. For this purpose the concept of ecosystem services will be introduced. The concept offers some considerable advantages for a physical assessment of biological invasions.

In the following all chapters are briefly described.

Chapter 2, "Biological invasions as ecological phenomenon", introduces the study object: biological invasions and corresponding impacts. Section 2.1 illustrates ecological properties of biological invasions that are important for a general understanding of the invasion problem. Of central importance is the invasion process (section 2.2), which is divided into four phases. Major factors influencing the invasion process are introduced and discussed. Invasive species are mostly feared for their harmful impact; section 2.3 illustrates how manifold such impacts can be, providing an overview of the varied impacts of invasive species, which constitute the motivation for the call for political action. Management strategies suggested by the CBD are explained in section 2.4.

Chapter 3, "Uncertainty linked to biological invasions", analyses the different kinds of uncertainty linked to the invasion problem. First a general taxonomy of uncertainty is developed (section 3.1.). Uncertainty occurs when considering the invasion process (section 3.2), because of the difficulty in predicting which species will become invasive, and when, how and where this invasion will happen. Section 3.3 depicts relevant sources of uncertainty regarding biological invasions. Section 3.4 illustrates uncertainty in the context of management options, in which the success of a measure cannot be predicted clearly. Section 3.5 shows how to deal with different kinds of uncertainty and how science can help reduce these. In addition, repercussions of uncertainty on the assessment of biological invasions are illustrated.

Chapter 4, "Economic valuation: foundations and application to biological invasions", refers to the role of economic studies for decision-making and its potential role for evaluating impacts of invasive species. It depicts the welfare economic aspects of biological invasions (section 4.1), and introduces cost-benefit analysis as an economic tool for decision aid (section 4.2). Section 4.3 introduces typical methods of environmental economic evaluation which can be applied for biological invasions, and therefore shows "how" an invasive species can be evaluated in economic terms. The commonly used concept of the Total Economic Value is described in section 4.4. Section 4.5 analyses the role of economic assessment studies and how far such studies can support decision-making in the choice of appropriate management measures. Section 4.6 addresses shortcomings and criticisms of an economic valuation of biological invasions and outlines the focus of an improvement of ecological-economic assessments. To specify the need where an improvement should start from, this section formulates the two central challenges of a physical assessment of invasive species, namely including uncertainty and covering the scope of impacts.

Chapter 5, "Survey of current studies on the economics of biological invasions", surveys current literature in respect to its usefulness for decision making. The main finding of the survey is that the physical assessment has insufficiently been considered. As a consequence of this finding the following chapters focus on possibilities to enhance the physical assessment.

Chapter 6, "Enhancing economic assessments with the concept of ecosystem services", introduces the concept of ecosystem services, its origins and its present relevance in politics as well as in science (section 6.1). It illustrates the importance of ecosystem services for human well-being. As biological invasions are often blamed for occasioning major impacts on biodiversity, the role of biodiversity will be explained. In section 6.2 it will be shown how the concept can be used to illustrate the manifold impacts of biological invasions. Section 6.3 discussed the advantages of the use of this concept regarding both the *scope*- and the *uncertainty*-problem.

Chapter 7, "Biological invasions in Germany and the case study of *Ambrosia artemisiifolia*", serves to apply the advantages of the concept and the contributions to the challenges of *scope* and *uncertainty*, developed in the chapters before, in practice. To do so, in section 7.1 the special situation in Germany is explained. With the help of ecosystem services the manifold impacts of invasive species in Germany are shown. In section 7.2 a study case of *Ambrosia artemisiifolia* uses the concept of ecosystem service exemplarily for a physical assessment of this invasive species in Germany. This species has been chosen because it is of growing concern for the public due to drastic health problems.

Chapter 8 "Final conclusions and recommendations for further research ", discusses the main thesis´ findings and the usefulness of the ecosystem services concept with its shortcomings and limits. The chapter also shows where further research is needed. It outlines implications of the findings on economic analyses for supporting decision-making in the context of biological invasions in general.

Chapter 9, "Summary (*English* and *German*) summarises the main findings of the thesis, first in English, and second in German.

2 Biological invasions as ecological phenomenon

This chapter aims to provide a basis for understanding the invasion problem and explains why decisions regarding the management of biological invasions have to be taken. The base of the problem are ecological properties. Thus, first, distinct definitions used in invasion biology and in the political context are introduced (section 2.1); in section 2.2 the four phases of an invasion process as well as those factors that promote or restrict the process are explained. Section 2.3 considers the various impacts of invasive species; Section 2.4 explains standard strategies to manage biological invasions;

2.1 Terms and definitions

The literature on biological invasions uses a variety of different terms to describe invasive species and the process under which a species develops to become a biological invasion (Richardson *et al.* 2000a; Heger 2001; Kowarik 2003a). It is worthwhile considering the different definitions, which will also give an impression of the scope of this thesis and clarify the context of this work.

There are two main perspectives on invasive species. The first perspective is the natural science one, widely applied by invasion biologists, where a biological invasion is understood as "a term used to describe the naturalisation and unintended spread of unwanted organisms in areas where they have not previously occurred naturally" (Jay *et al.* 2003: 121). Invasion biology focuses on the description of impacts caused by the invasive species, for instance how an invasion might affect biodiversity[6]. Besides research on changes in the environment, ecology also considers the functional traits that promote the capability of a species to become invasive, and the mechanisms, such as seed transport, that drive the invasive species dispersal (Kühn *et al.* 2004).

There is also a political interest in this problem. This is the second perspective on the phenomenon. Ecological changes due to invasive species can cause huge impacts on humans, such as health problems (e.g. Gabrio *et al.* 2006). As a consequence, particularly the environmental policy sector has realised calls for action, both on the global as on the regional

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[6] In the following, biological diversity (or biodiversity) means "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic systems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (Convention on Biological Diversity, Article 2. Use of Terms).

or local level, to restrict the ongoing dispersion of invasive species worldwide. The Convention on Biological Diversity (CBD) constitutes the base for political action. The CBD broadens the perspective of natural scientists on biological invasions by including societal aspects of impacts of an invasion. The definition of an invasive species included in Art. 8h of the CBD mirrors the difference between the natural scientific view and the environmental political perspective. According to the CBD, an "invasive alien species" refers to an alien species whose "introduction and spread threatens native ecosystems, habitats or species with socio-cultural, economic and/or environmental harm, and/or harm to human health" (CBD 2002). Damaging effects on society and humans, caused by invasive species, are therefore explicitly addressed. In contrast to the natural scientific understanding in invasion biology, the framework convention distinguishes between "alien" and "invasive alien" species. According to the understanding of the CBD, the natural scientific view would only cover "alien" species, defined as "a species, subspecies or lower taxon occurring outside of its natural range (past or present) and dispersal potential and includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce" (CBD 2003: 23). According to the CBD, the definition of an "invasive alien" species requires a connotation of damage, either to the environment or to society, whereas "alien" species are not considered harmful.[7]

There are several terms in literature to refer to invasive species. "Non-native", "exotic", "non-indigenous", "noxious", "non-captive" and "foreign" are terms often found in invasion literature. The use of such terms is confusing because they are used to describe both alien and invasive alien species; the meaning assigned to the term only becomes clear in the respective context. In addition, there are sector-specific terms, for instance in the agricultural sector, where the terms "pests" and "weeds" are used to describe invasive species. Both terms are especially ambiguous as native species can also be classified as "pests" or "weeds", i.e. they can cause the same kinds of damaging effects due to their proliferation. This is the case of the European crouch (*Agropyron repens*), native to central Europe.

[7] However, this understanding of invasive species and the derived call for action holds only for the CBD. Nations that have signed the convention have to develop and implement nation-wide strategies against invasive species. Incentives for such strategies sometimes base on a different understanding to that of the CBD. In Germany, for instance, the Federal Nature Conservation Act (Bundesnaturschutzgesetz, Section 41, Paragraph. 2) states that a species that is out of its natural range, i.e. is alien (*gebietsfremd*), demands conservation management strategies, as it has the *potential* to alter, i.e. to damage, the native flora and fauna, which constitutes the incentive to take action (Ortner 2005).

Terms of the kind explained above are manifold and both, the natural scientific and the political field, use different definitions to classify a species as invasive. In the present thesis the term "invasive species" describes a species according to the understanding of the CBD, i.e. a non-native species that will cause a damaging effect. Likewise, a species is defined as "alien" when it occurs outside its natural range but does not disperse, imposes no damaging effects on society and has not overcome all phases of the invasion process.

Besides the definition of biological invasions of the CBD, it is worthwhile considering the natural scientific distinction between "wanted" and "not wanted" organisms for a certain (bio-geographical) region[8] which is related to an occurring species being either native or non-native (Kowarik 2003: 16; Kühn *et al.* 2004). But how can one draw the line between native and non-native? Native and non-native species result from the evolutionary differentiation of organisms. Geographical isolation has always been decisive for this differentiation. Mountain ridges such as the Alps or large water bodies such as the Atlantic have been natural barriers for millions of years (Barbier 2001). These barriers prevented the exchange of genes and propagules between continents or regions, so that an area-specific fauna and flora evolved, completely adapted to specific environmental settings. Endemic species are species which only exist in one specific region in the world. They correspond to the extreme outcome of such isolation processes.

Non-native species are those that came into the new environment with substantial help of humans. The line of differentiation is 1492, the year Christopher Columbus discovered America (Kowarik 2003: 14). In the post-Columbus period a high number of non-native species were brought back from the New World and the role of humans as drivers for the introduction — both intentional and unintentional — of non-native species increased significantly in importance (Kowarik 2003a).

As a result, human agency, acting through global trade and transport, has significantly increased the number of species with a very short or even no evolutionary history in the environment where they have build stable populations in the present.

[8] The occurrence of a non-native species, however, has been observed on various spatial scales. The problem such species pose is not only region-specific and can be also observed on continental scales. Certain species might even occur on different continents.

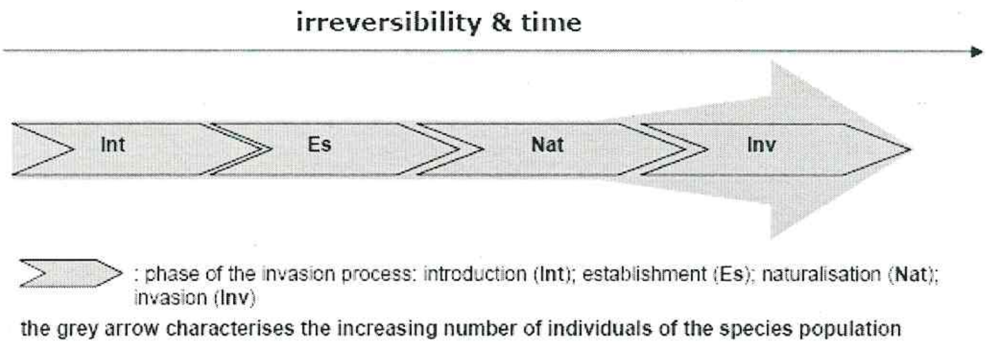
2.2 The invasion process and causing factors

2.2.1 Phases of the invasion process

The phenomenon of biological invasions is strongly characterised by the process in which a non-native species develops to become an invasive species. To study the process is important to understand the complexity of the phenomenon. The understanding of the process is relevant for the choice of management measures selected to restrict the negative impacts of an invasion. Besides, a thorough understanding of a species development is essential to recognise the uncertainty linked to the invasion process (see chapter 3).

A biological invasion is a dynamic, non-linear process that, once initiated, is largely self-perpetuating. Although ecologists consider the process as a continuum in which no clear-cut breaks exist, they do distinguish between four phases within the invasion process (Richardson *et al.* 2000a; Kühn *et al.* 2004). Figure 3 depicts the invasion process.

Figure 3: Phases of the invasion process (own source)



The four phases of the invasion process are:

1. **Introduction**: a species overcomes a major geographical barrier, promoted by human activity such as shipping, and appears in the new environment.
2. **Establishment**: many introduced individuals survive as casuals, reproducing sexually or vegetatively, but failing to maintain or establish a population over a long time period, and hence relying on repeated introductions for their persistence.
3. **Naturalisation**: the species reproduces consistently and establishes a vital population.

4. **Invasion**: the species produce reproductive offspring, often in a very large number and over considerable distance and time.

Whether a species can reach the following phase is always dependent on a series of factors. Estimations give a rule-of-thumb figure of 10 % for species that manage to progress from one phase to the next (Williamson 1996: pp. 31). This would imply that, from 1000 alien species initially introduced, approximately 100 would be able to establish themselves in a new environment; of these 100, 10 species would manage to establish a permanent population in the new habitat (i.e. become naturalised), and one species would develop to become invasive. However, like all generalization rules, it needs to be interpreted with caution (Williamson 1996: pp. 36). The rule is based on statistical data. It can be biased through sampling errors and only holds for certain classes of invasions[9]. Classes of invasions are groups of species that are of taxonomic similarity or affiliation (Lockwood *et al.* 2001). The proportion of the different classes varies in time because there can be a long period of time, often of over 100 years, between a species being introduced and becoming established. This phenomenon is called the "time-lag effect" (Kowarik 1995). Notwithstanding this rule of thumb, we will probably never have a scheme to predict the success of invading species (Drake *et al.* 1989: 365).

2.2.2 Factors influencing the invasion process

The search for the determining factors that cause an invasion is regarded as one of the key challenges, if not the major one, within biological invasion research, see for instance Kühn *et al.* (2004), Prinzing *et al.* (2002), D´Antonio and Kark (2002), Mooney *et al.* (2000).

Various approaches have been used to try to answer the question of which species become invasive and why they do so, but results cannot be generalised from one alien or invasive species to another (Prinzing *et al.* 2002). Ecological studies either consider a set of successful invasive species or compare the characteristics of native and invasive species (Goodwin *et al.* 1999). Undoubtedly, an array of phenomena and factors are involved in the species' ability to progress from one phase to the next of the invasion process. Despite the limited possibilities for finding general factors that advance or impede the invasion process, this thesis will attempt to determine those factors that seem to be relevant for all species which become a

[9] The ten's rule has been mainly investigated in the context of plant invasions, see Williamson (1996: pp. 31).

successful invader[10].

Table 1 lists seven factors relevant for every invasion process. Each factor is characterised through certain parameters.

Table 1: Factors that influence the invasion progress (own source)

Factor	Factor description			Relevant invasion phase
	Anthropogenic parameters	Ecological parameters		
		intrinsic	extrinsic	
Propagule pressure	Deliberate introduction in: - Agriculture - Horticulture - Forestry - Angling - Fishery	Size, number and adherent capacity of propagules		Introduction; Establishment; Naturalisation
	Accidental introduction through: - Transportation - Trade - Tourism			
Disturbance	Human-made disturbance through: - Land use (fallows, fields, pasture) - Construction		Natural disturbance events: - Fires - Erosion - Floods	Introduction; Establishment; Naturalisation
	- (roads, railroads, buildings)			
Lacking enemies			Lacking - Predators - Pathogens - Diseases	Establishment; Naturalisation; Invasion
Niches			Availability of empty niches	Naturalisation; Invasion
Geographical home range	Frequency of human contact for dispersion	Species capacity to: - Adapt to climate - Water availability - Nutrient conditions	Ecosystems characterised through: - Micro-climate - Water availability - Nutrients in the soil	Introduction; Establishment; Naturalisation Invasion;
Life form		- Size and number of propagules - Short juvenile period - Long flowering period		Introduction; Establishment; Naturalisation; Invasion
Use of available		- Adaptation to		Establishment;

[10] This extraction aims to understand the course of the invasions process in general and does not claim to consider all factors.

Factor	Factor description			Relevant invasion phase
	Anthropogenic parameters	Ecological parameters		
		intrinsic	extrinsic	
resources in the new range		resources for diet - Novel predator positions		Naturalisation; Invasion

Anthropogenic parameters are for instance transport systems or the cultivation of alien species for their utilisation. These parameters are relevant for the propagule agency. Ecological parameters promote or inhibit the invasive species' ability to progress. Species-specific traits are for instance a species capability to adapt to a changing climate. Environmental conditions characterise the ecosystem in which the invasion takes place. In the new environment the species will be exposed to specific environmental conditions, such as a certain ecosystem micro-climate and the availability of trophic resources that can determine the species' diet. It is a combination of the specific attributes of the species, i.e. its capacity to adapt to biotope-specific conditions, and of the ecosystem attributes, i.e. how far a new habitat can be invaded by a newly introduced species, that determine the capacity of a species to progress. The above-mentioned factors determine the species capability to survive and increase in density in the new target region. The affected target region will be described as target ecosystem, as ecological literature centres in its analysis on different biotopes or ecosystems that can be invaded by a species[11].

In the following, the above-mentioned factors are explained in relation to the different invasion phases.

2.2.2.1 Propagule pressure

Propagules can be seeds or breeding individuals of a species (Williamson 1996: 41). The propagule pressure can be understood as a "composite measure of the number of individuals released into a region to which they are not native" (Carlton 1996). It is determined by anthropogenic and ecological parameters.

Anthropogenic parameters are related to human activities, which have been acknowledged to be a key driver for biological invasions (CBD 1993; Williamson 1996: 30; Dalmazzone 2000). The contribution to the invasion process of the "human-mediated transfer of a species to a new range" has often been underestimated (Kowarik 2003: 295). Such a transfer happens

[11] While bio-geographical regions are larger scale units in ecological research that correspond with climatic zones, a habitat or ecosystem is shaped by defined abiotic and biotic properties and is a smaller scale unit.

either deliberately or accidentally. Deliberate releases have been widely practised through the import and the cultivation of alien species, e.g., species used in biological control. Species that are cultivated in agriculture, forestry and horticulture, but also those introduced for the purpose of recreation and fishing, are assumed to be more likely to spread into native flora and managed plant systems (Reichard and White 2001). Alien species have been introduced to be used as garden plants (e.g. *Impatiens glandulifera*), as agricultural crops (e.g. *Helianthus tuberosus*), as plants for hedges and shelterbelts (e.g. *Rosa rugosa*), as silvicultural crops (e.g. *Pinus nigra*) and as beekeepers' plants (e.g. *Solidago canadensis*). Alien species also have been imported for the purposes of soil improvement (e.g. *Robinia pseudoacacia*), erosion control (e.g. *Fallopia japonica*) and the enrichment of nature (e.g. *Lysichiton americanus*). The pool of cultivated plants fosters secondary releases. Secondary releases refer to the repeated introduction of a species into the new environment where it was already introduced before. Small populations face the risk of extinction, which stems from random changes in biotic and abiotic factors, relatively less important for larger populations (Heger 2001). Therefore, secondary releases enable a frequent flow of propagules from the pool of cultivated species. As a consequence small populations are better equipped to sustain a vital population. It is even assumed that secondary release might promote the progress of a species beyond the naturalisation phase (Kowarik 2003a). While initial introductions mainly act on the regional or continental scale by crossing the barriers between the native and the new habitat, secondary releases move species within the new geographical range, often even decades and centuries after the initial introduction. This frequently happens on local to regional scales (Kowarik 2003: 296).

Accidental introduction occurs indirectly through transport with seeds or crops, goods, vehicles or directly with people. For instance the accidental release of alien species transported in the ballast water in ships are at present one of the major sources of invasive species in aquatic ecosystems (Tamburri *et al.* 2002; van den Bergh *et al.* 2002; Lovell *et al.* 2006). Accidentally released species contribute significantly to the pool of possibly invasive species.

In general, a positive correlation between increasing human influence on ecosystems and the occurrence of invasive species can be assumed[12], i.e. the higher the degree of human activity in a certain region, the higher is the number of introduced alien species and therefore the possibility that one species might be able to progress to an invasion (Heger 2001).

As mentioned above the propagule pressure is also determined by ecological parameters, namely by the size and number of incoming propagules (Lockwood *et al.* 2005). A plant species with high numbers of propagules which are easily distributed due to their adherent capacity is more likely to overcome its natural range than one without such attributes (Lockwood *et al.* 2005).

The propagule pressure is mostly relevant for the introduction phase, which rather corresponds to a certain moment in time, i.e. the moment when the species enters the new environment for the very first time, than to a time period (Williamson 1996: pp. 33; Kowarik 2003a; Lockwood *et al.* 2005). However, ecologists often refer to the introduction phase of a species. This means that members of a certain species were found outside control or captivity, as a potentially self-sustaining population (Williamson 1996: 34).

2.2.2.2 Disturbance

Disturbance is characterised by anthropogenic and extrinsic ecological parameters. It has been shown that ecosystems are more susceptible to invasions when disturbed. Humans constitute important disturbance agents (Hobbs and Huenneke 1992; Kowarik 2003a). Williamson (1996: 28), however, states that disturbance only "reflects the fact that species are more likely both to be transported from disturbed areas and to arrive in them because of human activity". This holds at least for terrestrial organisms. Anthropogenic disturbance is especially important for isolated areas, such as islands. Remote islands frequently evolved under isolated conditions, so that they are often rich in endemic species (Traveset and Richardson 2006). The state of isolation is positively correlated to the susceptibility of an area. Due to natural barriers, ecological mechanisms have developed over evolutionary timescales, and species interactions, i.e. mechanisms among species and between species and the abiotic environment, have adapted to a high degree (ibid). Ecosystems that are formed by the

[12] Dalmazzone (2000) shows that demographic as well as economic factors, such as GDP per capita, merchandise imports, agricultural area, population density and land use (as permanent pasture as a share of total land area) is positively correlated with the number of introductions, and Kowarik (2003: 92) has used the concept of Hemeroby to illustrate the relationship of anthropogenic disturbances and their influence on distinct ecosystems.

interactions of such adapted species are often severely disrupted by species introductions caused by humans (see Section 2.3).

Disturbance, however, does not only imply the interruption of ecological processes and an influence of ecosystem properties by humans. Fire and floods can be also constitute disturbance events, after which site conditions, such as vegetation cover, can be destroyed or at least seriously disturbed. Bare and open land without any vegetation cover is often the result of these events. Such an interrupted ecological state, formerly constituted by the interaction of vegetation, soil, water and micro-climate, now offers the chance for invasive plant species to establish themselves within the new ecosystem and to expel the originally existing species due to a competition advantage (Kolar and Lodge 2001). In the case of newly invaded sites, invasive species can in turn influence natural disturbance events. Fire events can be promoted by certain types of invasive grasses, and fire suppression can occur due to the presence of certain invasive tree species (Mack and D´Antonio 1998; Brooks *et al.* 2004).

2.2.2.3 Lacking enemies

An important mechanism for some alien species that become invasive relates to the *enemy release hypothesis* (Keane and Crawley 2002). It states that "(plant) species, on introduction to an exotic region, experience a decrease in regulation by herbivores and other natural enemies, resulting in a rapid increase in distribution and abundance" (ibid: 164). Natural enemies can be vertebrate and invertebrate herbivores, pathogens, bacteria, and viral diseases. All such enemies either feed on them or have a negative impact on their health, and therefore on population size. Whether or not such natural enemies exist in the new habitat of a species has a great effect on the species possibility to establish a vital population, as natural enemies of the species usually regulate and balance the number of individuals of the alien species in its home range.

2.2.2.4 Niches

Williamson (1996: pp. 71) mentions the existence of empty ecological niches that can be filled by an invasive species. These niches can promote the ability of a species to establish or even to naturalise. Open niches do not often exist, and according to Williamson, when they do exist they would be of rather limited use in predicting the number of alien species to progress to a further phase. However, examples are known where alien or invasive species used the *same* niche as a native species, which resulted in a severe reduction of the native species' population. An example is the House Gecko (*Hemidactylus frenatus*) introduced in Mascaren

Island, which uses the same niche like native and endemic *Nactus*-species. The native population is under threat of extinction due to the invasive House Gecko (Cole *et al.* 2005).

2.2.2.5 Geographical range

The range of a species in its area of origin is assumed to be related to the probability of its moving and persisting elsewhere (i.e. invading) for two reasons. First, species that are wider-ranging are more likely to come into contact and to be carried with goods via transport. Second, species with larger original ranges are more likely to be pre-adapted to conditions in the new habitat (Goodwin *et al.* 1999). In the case of herbaceous plant species, for instance, their primary (native) latitudinal range seems to be the best predictor of potential invasiveness found so far (Rejmánek and Richardson 1996). Climatic matching is determined by conditions of humidity and temperature and show the species environmental tolerance in the new range. An example of this is frost-resistance. A species from the tropics will not be able to establish or even naturalise in central European countries, because a high mortality rate will place strong limits on the number of reproducing individuals. The match of native range climatic variables with host range ones can be used for predicting the success of a plant species in establishing itself (Curnutt 2000).

2.2.2.6 Life form

For invasive plant species, the self-maintaining capacity of a propagule, i.e. its ability to "survive" until environmental conditions are suitable for species reproduction in a new habitat, influences the possibility of individuals actually developing (Kühn *et al.* 2004). Short juvenile periods, short intervals between large seed crops, large size of seeds, long flowering period, and vegetative reproduction are also attributes of a plant species that promote rapid population growth (Crawley, in Goodwin *et al.* 1999; Rejmánek and Richardson 1996). As a consequence, and as mentioned above, size and number of dispersal units is not only important for the introduction of a species but also for its progress to become a successful invader. Short juvenile periods and short intervals between large seed crops are typical characteristics of plants that manage to establish themselves on disturbed sites. These attributes provide an advantage in the competition for light and space. Particularly ruderal plant species are expected to have high possibilities for establishing a sustainable population under disturbed site conditions (Prinzing *et al.* 2002). However, the high number of non-ruderal species proves that the sole criterion "being a ruderal species" is not a reliable one.

2.2.2.7 Use of available resources

Alien or invasive species are often capable of rapid population growth if they use available resources more efficiently than native species, or if additional resources are available that native species do not use. An example would be *Robinia pseudoaccacia*. This tree can grow on bare soil with low nitrogen content. Due to its capacity to store nitrogen in its rhizomes, the plant counts with an adaptive advantage.

That the differences involved might be subtle is shown in the study of British squirrels. In comparison with the invasive grey squirrels, the native species is better adapted to digest available acorns than the invasive one (Williamson 1996: 42).

The phenomenon of a newly introduced species becoming a predator in the food-chain and being superior to existing trophic structures is well known (Grosholz and Ruiz 1996). This is another aspect of the use of additional (food) resources. Effects of changes in the food web can be enormous and are explained in section 2.3.1.1.

To summarise: For the **introduction** phase, the characteristics of human-caused transport is important and, together with propagules' suitability for being transferred into new ranges, determine the so-called *propagule pressure*. For the **establishment phase**, in which alien species become apparent for the first time, and the **naturalisation** phase, factors that allow the species to *survive* and *increase* from low abundances have a central role. These factors are mainly ecological factors. To progress to the last phase of the invasion process, the **invasion** phase, the capacity to *spread* and to *increase significantly in local abundance* is decisive.

Several factors, such as the species' home range or use of available resources, cannot be attributed exclusively to species-specific traits or environmental conditions. It is always the combination of all factors that determine a competitive advantage of an alien or invasive species over present species compositions.

2.3 The scope of impacts of biological invasions

Up to this point the factors that influence the invasion process have been discussed. But what are the results of a successful invasion? What are the impacts if a species becomes a successful invader? Are there changes or interruptions in ecological processes due to invasions? And, according to the CBD's definition, to what extent are they causing serious damage by disturbing these ecological processes or by having a negative effect on humans?

To answer these questions, this chapter deals with impacts of biological invasions. It provides an idea of the scope of impacts and the magnitude of effects that happen due to invasive species. The majority of impacts occur within ecological systems, but some correspond predominantly to damages to society. Therefore, two major categories are distinguished: impacts that affect ecological systems and impacts with major effects on society.

In regard to the different invasion phases, the most important effects occur during the invasion phase. This is due to the large abundance of individuals of the invading species in the new ecosystem and due to its spread into new ecosystems. A species becoming a successful invader means that the species has developed just those properties that make it superior in competition to other species. Although the previous invasion phases generate impacts, these are considered to be minor (Williamson 1996: 115). According to the rough estimations of the ten's rule, impacts that occur during the invasion phase have a low probability.

2.3.1 Ecological impacts

Impacts of biological invasions on ecological processes have been studied in invasion biology for years, e.g. (Drake *et al.* 1989; Hobbs and Huenneke 1992; Mooney *et al.* 2000). Ecosystems contain biotic and abiotic components. The biotic components of an environment are organisms (plants, animals, fungi, protozoa, bacteria). Soil, climate and water constitute main abiotic components of ecosystems. One can distinguish between the major effects invasive species have on biotic properties and on abiotic properties. Due to ecosystem interactions, effects on species communities due to biological invasions (biotic properties) often result in changes of soil conditions or water availability (abiotic elements) and vice versa. Invasive species might also have an effect on the level of genes.

2.3.1.1 Impacts on genetic diversity

Invasive species are sometimes feared for their "invasional meltdown" impact: their causing genetic extinction through hybridisation and introgression (Rhymer and Simberloff 1996: 21; Simberloff and Von Holle 1999). Hybridisation means "the interbreeding of individuals from what are believed to be genetically distinct populations", whereas introgression is "a gene flow between populations whose individuals hybridize, achieved when hybrids backcross to one or both parental populations" (Rhymer and Simberloff 1996: 84). In general, one can understand such processes as a mixing of genes of formerly distinct gene pools. The effect was observed with mallard ducks (*Anas platyrhynchos*) introduced to New Zealand and eastern USA: native duck species became more mallard-like and the formerly distinct genetic pool of the native species declined (Rhymer and Simberloff 1996).

2.3.1.2 Impacts on inter-species processes

"Being an enemy, which here means eating or consuming other species, is the most likely way that an introduced species will have a major effect" (Williamson 1996: 126). Regarding impacts on biotic communities, invasive species either impact vertical trophical structures by becoming a new predator, or influence horizontal food-chain processes through competition. Competition is thought of as "any negative-negative effect between two species", whereas amensalism is "a negative-zero effect where one species affects another species but is not itself affected", and swamping refers to one species overgrowing all others (Williamson 1996: 137).

2.3.1.3 Impacts on vertical food-chain processes

Numerous ways in which invasive species can impact on vertical food-chain processes are known. Probably the most recognised enemies are mammals, like feral cats, rats, dogs or rabbits. Such species enter at the top of the food-chain and meet organisms that have no defence mechanisms. Feral cats and rats can have drastic effects on ground-nesting bird species that originally did not have natural enemies on the ground (Williamson 1996: 127). This problem has been mostly observed for islands (ibid). But also grazing can create severe effects, as shown in the case of feral goats and rabbits in Mexico and New Zealand. In the case of feral goats in Mexico, some native endemic species only survived on steep coastal cliffs while the goats turned the natural habitat of these species into grassy pasture (ibid).

Mammals are not the only predators to cause serious effects on food-chains. Introduced ants are another example in which an introduced species caused effects on other ant species, reptiles and snails (Williamson 1996: 130). In aquatic systems, the introduction of invasive fish species has in many cases lead to the extinction of endemic species. This outcome of an invasion has also been observed in many other terrestrial communities (Williamson 1996; Williamson 1999; Nugent *et al.* 2001; Towns *et al.* 2001; Traveset and Richardson 2006). The island Guam is an example: the unintentional introduction of the Brown tree snake (*Boiga irregularis*) in the 1950s led to various cascading effects due to the high preying pressure of the snake. It resulted in the extinction of twelve of Guam's endemic bird species, and 9 of 12 native lizard species are expected to become extinct in the short term (Wittenberg and Cock. 2001). Extinction of native species is the worst effect caused by an invasive species, whether the species is directly involved in the extinction or is only "a contributing factor for a species that is already in trouble"; extinction is in all cases irreversible (Gurevitch and Padilla 2004: 471).

2.3.1.4 Impacts on horizontal food-chains

As mentioned above there are different ways how an invasive species can exert influence on horizontal food-chain processes. Only a few will be mentioned.

Competition or amensalism between two species can be subtle and complex. An example of amensalism is the case of the introduced House Gecko, already mentioned above. It became apparent that the competitive advantage of the invader resides in the fact that members of the native species avoid physical proximity with the larger invader. Although the native species reproduces asexually, which is a reproductive advantage compared to the invaders that reproduce sexually, the native geckos suffered from the loss in feeding that comes from this avoidance behaviour (Williamson 1996: 139).

Next to competition, swamping is also a well-known phenomenon. One evident example is the water hyacinth (*Eichornia crassipes*), a floating weed that reproduces asexually. It is a typical swamping invasive species. The plant manages to survive in very little fragments and to eventually proliferate, overgrowing other native species. In the case of the water hyacinth a fragment does not need to be larger than one cm for it to survive and be able to reproduce itself.

2.3.1.5 Impacts on ecosystem processes

As mentioned above, biological invasions not only directly affect species communities, but in certain cases the predominant effect of an invasive species can be on the abiotic properties of the invaded ecosystem. Well-known examples in which invasive plant species have had an effect on soil conditions are *Tamarix*-species and *Robinia pseudoacacia*. The latter species is able to add nitrogen to the soil due to its ability to fix nitrogen in its rhizome system (Kowarik 2003: 34). A dense population of *Tamarix*-species creates a salinisation effect due to its deep roots which drain the soil dry (Lesica and Miles 2004). Animals are also known to alter ecosystem states. The European periwinkle (*Littorina littorea*), an intertidal snail, has caused a shift of coastal landscapes from mud flats and salt marshes to rocky shores (Williamson 1996: 145).

2.3.2 Societal impacts

The above-outlined examples of species' impact on ecological processes need not become apparent to society. However, some impacts of biological invasions affect human societies directly. Every impact is basically determined by ecological mechanisms, such as advantages in competition. Such a mechanism has effects on ecological systems, but also on human systems, e.g. if an invasive pest overgrows a cultivated crop. Whether impacts are damaging or beneficial to humans will be explained in the next section and are thus often referred to as economic impacts.

2.3.2.1 Impacts on production yield

Impacts of invasive species are well-known in the agricultural and forestry sector. Decreased production yield is known in forestry and agriculture, but also to some extent in horticulture. Swamping stands of the black cherry (*Prunus serotina*), which forms thick stands in the understory of forests, can result in wood-yield losses due to their preventing the regeneration of native broad-leaved tree species (Rode *et al.* 2002: 175).

Many plant species once introduced from Europe became severe pests in the USA and are now the main reason for a lower yield of crop, fodder and other agricultural products (Headrick and Goeden 2001). But also invasive animal species, such as the Gypsy moth, are today one of the most feared pests in agricultural production (Lacey *et al.* 2001).

A species responsible for a large conflict of interests is the Nile perch (*Lates niloticus*). Introduced into Lake Victoria with the aim of improving natural fish resources in the 1960's, the fish caused a rapid and drastic change in the ecology of the lake. While 200 of the former endemic fish species disappeared due to the high preying pressure of the invasive species, the fish catch of the Nile perch increased and improved the local commercial fish market. In ecological terms the invasion has been a disaster, whereas in terms of providing additional food resources to the region it is a success.

2.3.2.2 Invasive species used for biological control

In the case of biological control, introduced species can cause beneficial effects. There are two distinct ways of using alien species as biological control agents to control undesirable species: classical and inundative biological control.

In classical biological control the area of origin of a damaging invasive species is located. By exploring the home range of the species natural enemies can be identified and be used as control agents in the new range. The purpose is to control the pest with its natural enemies in the new range without further costs (Williamson 1996: 122).

Inundative biological control applies a microbial biological control agent, such as a chemical pesticide, by spraying it on the pest. A well-known example for this is the use of *Bacillus thuringiensis*, a bacterium that produces a protein toxic to insects and which is predominately alien to its geographical area of application (Williamson 1996: 122).

Although biological control is an example where the use of alien species is beneficial, there is always a risk that the introduced enemy might prey non-target species (Cullen *et al.* 1995).

2.3.2.3 Health effects

According to Williamson (1996), the probability of an invasive species becoming harmful to human health is low, but if the species does produce damaging effects, these can be serious. Impacts of invasive species on human health are considered to be some of the better-known effects due to such species. Examples for this impact are the Common Ragweed (*Ambrosia artemisiifolia*) and the Giant Hogweed (*Heracleum mantegazzianum*). The Common Ragweed releases highly allergic pollen, which is assumed to be several times more aggressive than natural weed pollen (Gergen *et al.* 2000). The Giant Hogweed produces phototoxic substances that when touching the plant severely irritate the skin (Lagey *et al.* 1995). Species which have a direct effect on human health are considered to be particularly damaging.

2.3.2.4 Damages to public facilities

Human-installed facilities like dykes, water pumps or traffic systems are sometimes subject to impacts of biological invasions. The Zebra Mussel (*Dreisena polymorpha*), for example, is well-known for its blocking water-pipelines or causing biofouling problems in reservoirs, docks, navigation channels and bridges (MacIsaak 1996; Aldridge *et al.* 2004).

Before turning to the options to counter the impacts of biological invasions, it is worthwhile considering the nature of invasive species effects, i.e. the ongoing irreversibility during the invasion process. For a successful management of invasive species irreversibility has to be taken into account.

2.3.3 Irreversibility during the invasion process

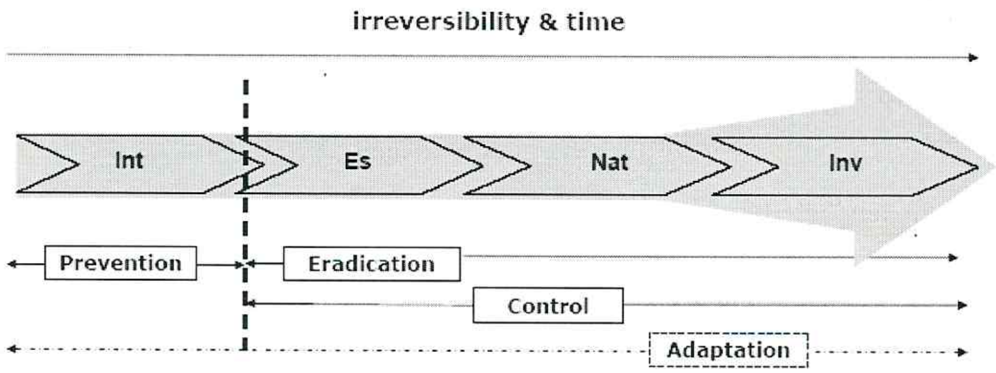
In general, the self-perpetuating character of the invasion process implies an increasing degree of irreversibility the further the invasion has developed. The more phases a species manages to overcome within an invasion process, the higher the degree of irreversibility. This means it becomes increasingly cost-intensive to reduce the number of individuals or populations of invasive species and to restore the original status-quo before the invasion took place (see Figure 3). This has implications for the management of invasive species (see section 2.4). Particularly the invasion phase implies a high degree of irreversibility. The high number of reproductive off-spring and the species' dispersal over considerable distances impedes the success of a measure aiming to reverse the process to a minimum or even to eliminate the impacts. Irreversibility during the invasion process has two aspects: (i) the reversion of the cause of impacts, which is related to the number of individuals of the invasive species, and (ii) the reversion of impacts. Regarding the first aspect, the reversion of the cause of impacts, i.e. the eradication of all individuals of the invasive species, is frequently possible in the short term. In the long term it is re-invasion that constitutes the main challenge, as the propagule pressure is often too high to prevent re-introduction. This is a typical case of secondary-release effects (see above). Even if re-introduction can be prevented, minimal remains of propagules can lead to a fast re-invasion of formerly cleared sites. Thus, the prevention of a re-invasion over a long time span is often more challenging than a singular abatement action.

The second aspect, the irreversibility of impacts, corresponds to environmental problems due to ecological processes and mechanisms having changed irreversibly, and subsequent ecosystem conditions only allowing for the establishment of a secondary, and thus different, ecosystem. The difference between both aspects of the irreversibility of biological invasions lies in the fact that even if an invasive species' population does no longer exist the impacts it caused might still continue to do so. The most prominent example is the irreversible alteration of a native species community, which, in the worst case, results in the extinction of an endemic species. This cannot be reversed by any means, even if humans are eventually able to reverse the invasion that caused the extinction (Andersen *et al.* 2004).

2.4 Management options of biological invasions

When taking the increasing irreversibility into account, the main management options to counter biological invasions are the following: first **prevention**, second **eradication**, third **control,** and fourth **adaptation**. The first three are recommended by the CBD, Art. 8(h). In the Guiding Principle 2b, the CBD demands the hierarchical implementation of the management options (CBD 2002). A hierarchical implementation means that prevention should be preferred before eradication, eradication before control, and control before adaptation. The increasing degree of irreversibility along the invasion process (see section 2.3.3) explains the usefulness of and the reasons for requiring a hierarchical implementation. It takes into account that the more a species has advanced in its development towards becoming a successful invader, the harder it is to reverse the invasion process by eradicating the invasive species. Figure 4 links the invasion process with these management strategies.

Figure 4: The invasion process and corresponding management options (own source)



The following targets for the four management strategies can be defined:

<u>Target of *prevention*</u>: Prevention aims at the total **avoidance** of **all** accidentally incoming alien **species**. Total avoidance means that no alien species becomes accidentally introduced into a foreign environment. Prevention relates to the introduction phase. As a result of zero incoming alien species, all possibly occurring impacts in the course of the invasion process are avoided.

<u>Target of *eradication*</u>: Eradication aims at the total **extermination** of the population or reproducing individuals of one species **after** its **introduction**, no matter if the species is established, naturalised or invasive. Total extermination means the **cessation** of **all impacts** of the alien or invasive species.

<u>Target of *control*</u>: Control aims at the regulation, mostly **reduction**, of the population of **one species** to a certain degree of the original size of the population. This goal is set by decision makers responsible for invasive species management. Via the reduction of the species

population, control aims to **lessen all** occurring **impacts** or to keep them at a certain level.

Target of *adaptation*: The goal of an adaptation strategy is to directly **reduce the impacts**, and not the species' population as source of the impacts. Adaptation focuses mostly on one specific impact of an invasive species. The reduction goal, e.g. whether the damaging impacts should be halved or reduced to zero, is often not specified. The emphasis is placed on reducing the impact.

Each of the general strategies discussed above comprises several measures. Prevention, for instance, is achieved through measures such as quarantine restrictions, black lists that only permit the introduction of certain species, or the treatment of trade imports as potential vector of invasive species. Both eradication and control are applied after the introduction or secondary release. They are also applicable during the whole invasion process.

Eradication efforts normally take place during the early phases of the invasion, when the species' number of individuals is comparatively low and a limited number of sites are affected, although it is likely that "exterminating an exotic species once it is established is nearly impossible" (Pimentel 2005: 699). Control is usually applied after a species has entered the invasion phase. The degree of irreversibility is high at this phase, and eradication is often not successful. Adaptation is a strategy not included in the CBD suggestions. It covers actions that reduce the impacts of introduction, establishment or spread of the species, without changing the likelihood that the progress of the species will happen (Perrings 2005). In comparison to the CBD-strategies, which aim to affect the species' population directly as the source of the impact(s), adaptation strategies aim at the reduction of impacts, but do not intend to influence the source of damage, i.e. the population of the alien invasive species that causes the impacts. This might be one reason why the CBD does not recognise this measure. An example of adaptation would be the move to alternative production sites because former sites are infested with alien invasive species. Education and information campaigns to raise public awareness about the damaging effects of invasive species are further examples.

Prevention, eradication, control and adaptation are general management strategies for dealing with invasive species. Each type covers a set of measures that can be employed in the situations at hand.

3 Uncertainty linked to biological invasions

"The credibility of assessments is closely linked to how they address what is not known in addition to what is known" (Millennium Ecosystem Assessment 2003: 23). Such credibility is also necessary for economic assessments of biological invasions because the prediction of a new invasion and the availability of data about impacts as well as about management effects is still a central challenge. Reliable predictions have not been possible up to now (D´Antonio and Kark 2002).

The high degree of uncertainty regarding biological invasions explains why the CBD has been demanding a precautionary approach that favours prevention. In economic terms prevention is encouraged because costs at that phase of the invasion process are usually lower than for late-stage management. Although prevention is cost intensive, the benefits outweigh the costs, as impacts remain negligible compared to late-stage impacts (Finnoff *et al.* 2006). In order to assist sound decision-making in terms of suggesting the appropriate management options, an extensive analysis of uncertainty linked to biological invasions will be done in this thesis. This analysis enables to derive recommendations how to deal with uncertainty of biological invasions.

Incomplete knowledge, which in the following will be considered as a synonym for uncertainty in the broad sense[13], becomes evident when considering the whole invasion process. It is still unclear at what point in time a species becomes invasive, and the reasons for a certain species to develop into an invader (i.e. the specific ecological mechanisms that drive the invasion) are still not fully understood. Without this information, an estimation of the success of a management strategy is very difficult. Therefore, uncertainty arises concerning:

1. The baseline: Who will become invasive and when?
2. The impacts: What will happen (i.e. which effects are expected), and where will these impacts become apparent? If effects can be predicted, are they transferable to other

[13] Uncertainty covers several situations of incomplete knowledge. Uncertainty in the narrow sense, in the following referred to as uncertainty i.n.s., is used to describe one specific type of incomplete knowledge (see page 34).

invasions or invasible ecosystems[14]?

3. The management: How successful can management strategies be?

The following chapter will discuss where we have to deal with incomplete knowledge regarding biological invasions. However, there will be no theoretical discussion on uncertainty as a general phenomenon regarding environmental problems. Instead, the examination of uncertainty focuses on the invasion process and its mechanisms. The aim is to reflect on possibilities of including incomplete knowledge in ecological-economic assessments of biological invasions and to deal with it appropriately.

For this analysis, in a first step a taxonomy of uncertainty will be introduced which comprises different types of uncertainty. In the second section these different types of uncertainty are linked to biological invasions´ impacts. Section 3.3 examines the sources of uncertainty which are crucial for the design of management options and the aim to reduce uncertainty. Section 3.4 shows where uncertainty occurs in the context of management options. The last section suggests approaches how to deal with uncertainty and shows typical decision rules used in economics for situations of uncertainty. Finally, the rules are considered in the context of biological invasions and how uncertainty can be reduced.

3.1 A taxonomy of uncertainty

Scientific approaches to uncertainty have been various. Today research on this topic, especially in the context of environmental problems, is a field with various disciplinary approaches[15]. All approaches aim to estimate and appropriately handle situations where society has to make decisions regarding problematic events on the basis of incomplete knowledge. Basically two dimensions become important when classifying and characterising different situations of uncertainty. On one hand, there is the *outcome* of a decision, i.e. an event which usually contains harmful or at least unfavourable aspects to society or individuals. On the other hand there is the *probability* of an outcome to occur. An outcome in the context of invasions is considered to be *any effect (or event) that occurs due to an alien species or its regulating management in the course of the species development to become an*

[14] Although impacts often refer to damages, in the following effects and impacts are synonyms and do not have any negative connotation. They simply describe that biological invasions cause changes in ecosystem properties which are either valued negatively or positively. This valuation will not be object of examination.

[15] For a comprehensive overview of historic and current disciplinary approaches to the field of environmental risk and hazard research, see for instance Buergin (1999).

invader. An effect can be spatially defined and described in its character and relevance for human well-being. Whether the outcome contains positive or negative impacts relevant for humans is a matter of perception. In the context of an invasive species process, invasive species effects often contain a negative character (i.e. unfavourable to humans), whereas in the course of applied management options impacts mostly contain a positive character (i.e. benefiting to humans). However, impact in its primary understanding does not contain any value.

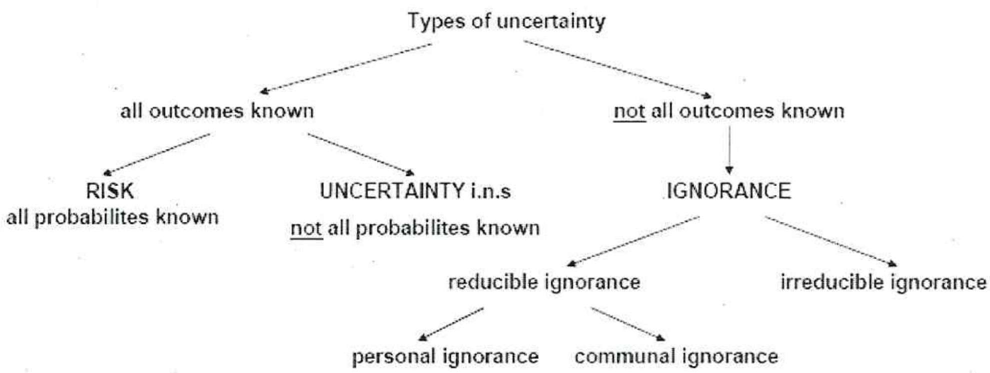
Approaches that work with outcomes and probabilities in the literature of decision theory can be all traced back to the fundamental distinction of risk and uncertainty i.n.s. by Knight in 1921 (Wätzold 1998: 70). Knight states that "the practical difference between the two categories, risk and uncertainty [i.n.s.] is that in the former the distribution of the outcome in a group of instances is known (either through calculations a priori or from statistics of past experience), while in the case of uncertainty [i.n.s.] this is not true. The reason being in general is, that it is impossible to form a group of instances, because the situation dealt with is in a high degree unique" (Knight 1921: 233). On this basis theoretical approaches have been developed that suggest rules to deal with uncertainty. The most prominent ones will be introduced in section 3.5. But before turning to concrete decision problems under incomplete knowledge one has to understand the different types of uncertainty. In the context of environmental problems, usually perceived as damaging outcomes of human decision-making, different taxonomies have been developed. They usually comprise a broadened and more detailed understanding of risk and uncertainty i.n.s., than the Knightian (see for example Stirling (2002: pp. 33); Klauer and Brown (1996); WBGU (1999: pp. 58); Walker *et al.* (2003); Common and Stagl (2005: pp. 379); Hansjürgens (1999: pp. 344)).

In the following, a nomenclature has been developed that is based on the taxonomy of Faber *et al.* (1992). This taxonomy is suitable because it offers a description of uncertainty which explicitly addresses one situation that is very typical in the context of biological invasions. It is the situation of ignorance. Ignorance exists when decisions have to be made under circumstances for which neither all outcomes of an invasion nor their probabilities can be completely anticipated. The authors of this taxonomy consider a deepened understanding of uncertainty a way to change the attitude towards environmental problems, with openness and flexibility as characteristics. They acknowledge that human knowledge is restricted, as environmental problems by nature are often global, long-term and very often involve the emergence of unpredictable events. Such unpredictable events generate *surprise*, which happens if *ex-ante* expectations did not become true from an *ex-post* perspective. Faber *et al.* (1992: 218) ask fundamental questions such as "What can we know? What possibilities for action do we have? What can we do?". Regarding biological invasions, the same questions arise, such as "What do we know? What predictions are reliable or even possible? What possibilities (i.e. management options) exist to encounter invasions impacts?" And, "how will the invader respond to certain management options?".

Related to the different *sources of surprise*, the following *types of uncertainty* can be distinguished (see also Figure 5)[16]:

1. **Risk** is a situation for which all potential outcomes can be characterised and probabilities for each outcome are known.
2. **Uncertainty in the narrow sense** is a situation for which all potential outcomes can be characterised, but not for all outcomes are probabilities known.
3. **Ignorance** is a situation for which not all potential outcomes can be characterised because events might occur that are outside the scope of expected possibilities.

Figure 5: Types of uncertainty linked to biological invasions as sources of surprise (derived from Faber *et al.* 1992)



The exact contrast between the different types of uncertainty exists if the future event will happen with certainty. In order to complete the categorisation, this situation will be referred to as:

[16] In literature one can also find other definitions of the three types of uncertainty regarding environmental problems, see for instance Faucheux and Froger (1995); Hansjürgens (1999: 344).

4. **Certainty**: it is a situation when the probabilities for the outcomes to occur reduce to just one value (at 100%).

The taxonomy of Faber *et al.* (1992) is appealing, since it assumes being aware of a situation of ignorance there are approaches to reduce it. In the context of biological invasions, the worst case is if we are in a state of ***ignorance***[17], i.e. we are able to address particular questions regarding a certain area of knowledge. Scientific effort, however, implies the generation of new and more incomplete knowledge that is in line with the general tenet "the more I know, the more I know I do not know". Additional information always implies that certain questions have been answered, but that at the same time new questions emerge. This is why we are always to a certain degree in a state of ***irreducible ignorance***. Irreducible ignorance occurs in the context of biological invasions if we acknowledge that randomness plays a vital part in the whole invasion process. To a large extent it is random whether a certain species might become introduced into a new range although there are factors known to restrict or promote its introduction (see section 2.2.2). Regarding the possibility of answering questions, one is in a state of ***reducible ignorance*** when ignorance "may be lessened, or even eliminated" (Faber *et al.* 1992: 226). Reducible ignorance contains ***communal ignorance,*** which is when a society is more or less confident that one can understand unknown phenomena more fully through scientific explorations. It is usually reduced via science and research. ***Personal ignorance*** is when individuals use available information inefficiently.

The "belief" of science is that our uncertain ignorance can be converted into knowledge because it assumes reducible ignorance. The underlying assumption is that uncertain events and outcomes of the invasion process are amenable to scientific examination, and possibilities exist to convert communal ignorance into uncertainty i.n.s. or even risk over the course of time. The fact that biological invasion research has developed results that offered a possibility to reduce our communal ignorance to uncertainty i.n.s. in the past indicates that this research is amenable to scientific examinations. Research results have facilitated the formulation of one reliable criterion so far, namely the estimation of the invasion potential of a species if it has already been invasive in other regions of the world (Grosholz and Ruiz 1996). This species is most probable to become invasive in another target region if newly introduced. The criterion does not allow a situation of risk, as this criterion still does not have stochastic

[17] Ignorance in this context belongs to open ignorance, where "*one will become attentive, e.g., of events and information*" and where "*one will try to understand surprising events by learning and research*" (Faber *et al.* 1992: 225).

properties for all regions of the world.

The taxonomy of uncertainty will now be applied to biological invasions.

3.2 Uncertainty regarding biological invasions impacts

When reflecting on uncertainty regarding biological invasions impacts mainly two questions arise. They regard:

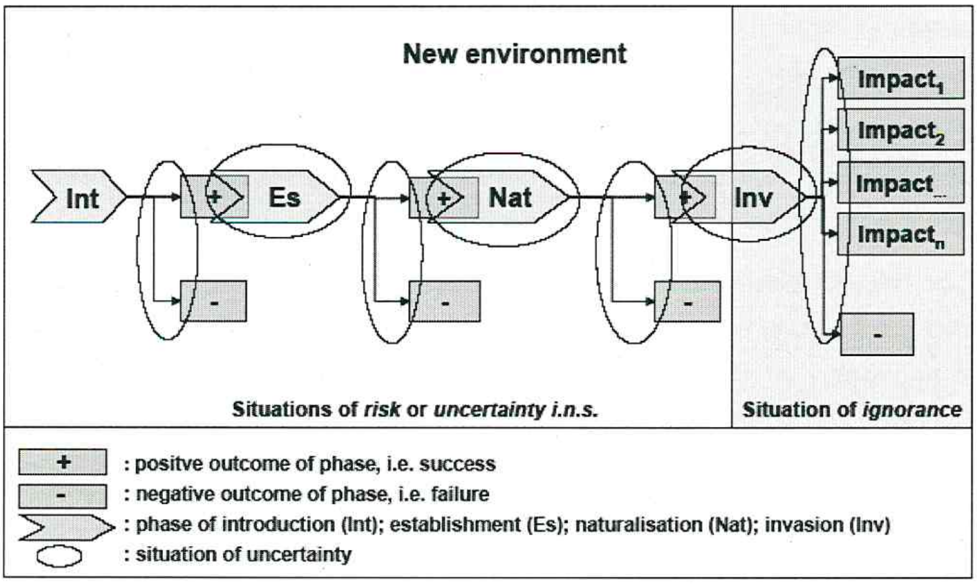
1. The time dimension: At what point in time will a species be introduced, and when will it progress from one phase to the next to become actually invasive?
2. The impact dimension: What are possible impacts in the course of the invasion process, to what extent do they affect ecosystems and humans, and where will they happen?

3.2.1 Preparatory considerations

To understand both dimensions on which incomplete knowledge states a problem for the management of invasive species, it is useful to consider the dynamic and non-linear process of an invasion.

Figure 6 depicts the invasion process in a model, where the invasion process is divided into phases (arrows) and outcomes (boxes).

Figure 6: Outcomes of the invasion process for a single-species perspective (own source)

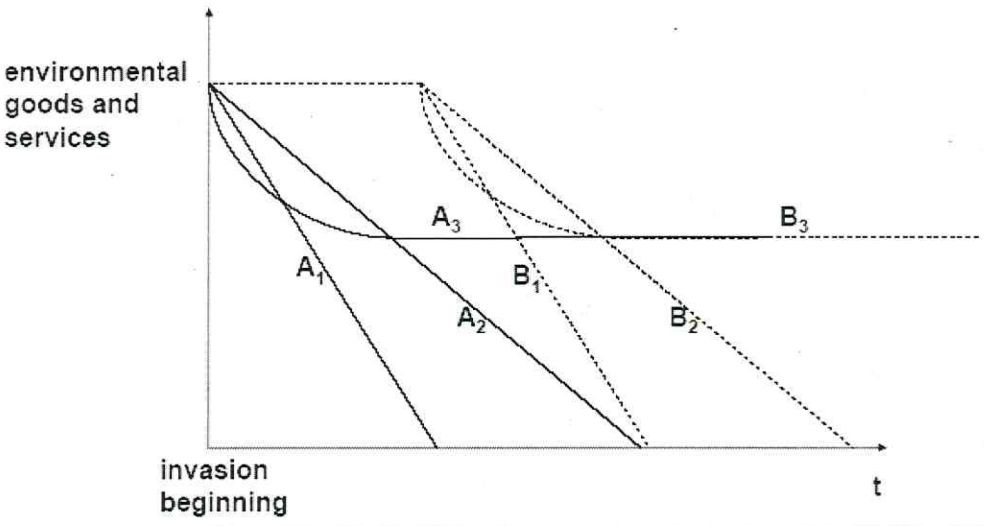


This flow-diagram allows a more systematic identification of sources of uncertainty and shows how to use available knowledge for the design of a general handling of uncertainty. The model refers to outcomes of an invasion of one single species. As one can see for the first three phases (i.e. **introduction**, **establishment**, and **naturalisation**), there are two possible outcomes: failure or success. If a species fails, it either does not survive in the new environment or it does not progress to the next phase, i.e. it remains in the current phase. If a species succeeds, it is able to enter the next phase of the invasion process. Success during the first three phases does not contain any harmful impacts as they are fairly small during that time and can be neglected in its relevance for influencing human well-being (Williamson 1996: 31). Concerning the **invasion** phase, there are several outcomes (impact$_1$-impacts$_n$) next to the outcome "failure". These impacts do have an influence on human well-being as we have learned from chapter 2 before. A situation of uncertainty (marked with a ring in Figure 6) exists during each phase and during the transition from one phase to the next. This means one does not know whether a species fails or succeeds (possible outcomes of transition), and whether the species develops during a phase to progress or to fail (outcome of each phase).

Of course, the type of uncertainty differs concerning whether one or multiple species are regarded. This means the pool of considered species is important as it can be assumed that the higher the number of considered species the less information is available about impacts. Therefore, predictions become increasingly difficult. In the following, different situations of uncertainty are illustrated according to the number of considered species. The maximum number of species that might become invasive in one certain region of examination comprises all species of the world that are currently not present in this target region but might be possibly introduced one day. The minimum number of potentially alien species is one species. The above mentioned preparatory considerations are important for the impact dimension.

For the temporal dimension, we are uncertain whether a species becomes immediately invasive (species A) or if it has to surpass a time-lag (species B). Figure 7 shows that we are uncertain whether an invasion will very rapidly lead to a reduction of environmental properties affected by the invasions (e.g. original species abundance, (A_1,A_2 or B_1,B_2)) or whether the invasion will at some point be at an equilibrium where damages do not increase any more (A_3 or B_3).

Figure 7: Possible courses of an invasion and its impacts on environmental goods and services (own source)



A_1 ,A_2, A_3: possible courses of an invasion of species A
B_1, B_2, B_3: possible courses of an invasion of species B

3.2.2 Uncertainty regarding the time dimension

As stated in the beginning of this section, uncertainty concerning biological invasions′ impacts has to be regarded on two dimensions. In the following the time dimension of impacts will be regarded.

Concerning this dimension of impacts we (i) do not know when the time of transition will be (i.e. the point in time a species progresses from one to the next phase) and (ii) the temporal extent of each phase. For the different phases, however, it is worth considering each separately. Outcomes differ regarding whether one or several species are regarded.

Concerning the **introduction** phase, a distinction has to be made between species that become introduced deliberately and the ones that cross the native range accidentally.

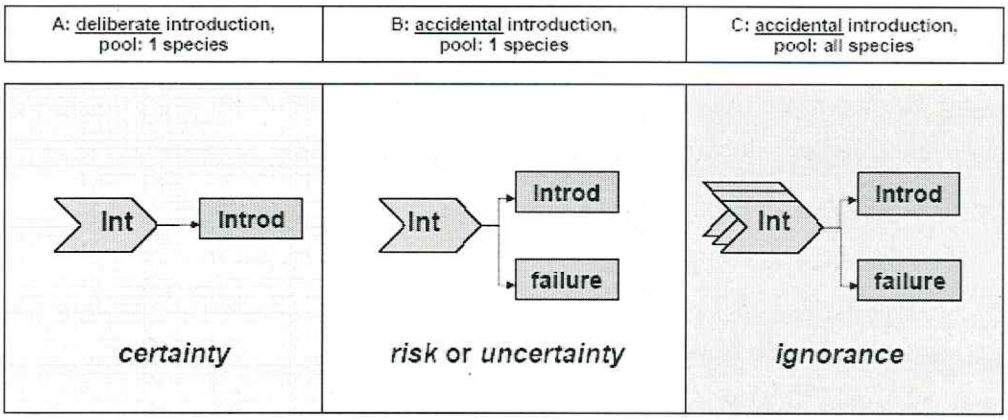
Case A: Species imported for deliberate cultivation. The point in time of entrance can be identified via trading lists of horticulture, agriculture or forestry, which record the number of species and the reason of introduction. Using the nomenclature from above, we have a situation of certainty, as the outcome (introduction) and its probability to occur is fully known and (see Figure 8).

For accidentally introduced species the situation is different. The overall number of incoming species is highly dependent on the degree of trade and transport activities in the region under analysis, as several studies underline (see section 2.2). We have to distinguish whether a single species or the maximum number of possibly incoming species is object of examination.

Case B: If a single species occurs in adjacent habitats or regions, the probability of that species to become introduced is much higher than for a species whose habitat is not in proximity to the region or habitat in question. However, identifying the exact point in time of entrance still remains impossible; there can be only approximate estimations. Therefore the situation will vary from risk to uncertainty i.n.s., depending on the knowledge about previous occurrence of the species.

Case C: In the case of all possibly introducible species, it is impossible to identify the species that might be introduced out of such a huge pool of species. The situation when all species that might become introduced is of concern, if decision makers reflect on a suitable prevention strategy for the introduction phase. Targeting all potentially incoming species causes a situation of ignorance, as we do not know which species will manage to overcome the new range.

Figure 8: Outcomes of the introduction phase, depending on the considered pool of species and the way of introduction (own source)



For the subsequent phases of **establishment**, **naturalisation** and **invasion**, it is not important whether a species has been introduced deliberately or accidentally. From the establishment phase on, the alien species becomes apparent in its environment either by establishing a viable population or occurring as individuals at certain spots. Regarding the progression time during each of these phases, we are either in a situation of *risk*, in which probabilities can be assigned to outcomes, or of *uncertainty i.n.s.*, if probabilities are not available.

3.2.3 Uncertainty regarding the impact dimension

Analysing impacts of invasive species on humans or ecosystems has always been central for ecological-economic assessments of biological invasion. Alien species increase in public interest the more they manage to overcome initial establishment phases. Of most public interest are species that have reached the last phase, the invasion phase where species somehow become harmful.

Now impacts of alien species during all phases of the invasion process will be examined. Referring to the definition of outcome, we will consider all *effects (or events) that occur due to an alien species or its management in the course of the species' development into becoming an invader.* These effects will be described with respect to character and relevancy for human well-being.

For the phases of **introduction**, **establishment** and **naturalisation,** the two possible impacts of success and failure have already been mentioned. Regarding the relevancy for human well-being, success is an outcome that has been identified but is not considered to be damaging to human well-being (see section 3.2.1), i.e. during the first three phases, impacts of invasive species are negligible. Working with the above mentioned nomenclature of probabilities and outcomes, we are in ***situations of uncertainty i.n.s.*** or of ***risk,*** as all outcomes can be predicted (see Figure 9)[18]. However, associated probabilities for the outcomes to occur depend on the knowledge already existing for the species at hand. The more we know, for instance if a species has been found in adjacent ecosystems, the more we are in a situation of risk with available probabilities to predict how the species will react in the new environment.

The last phase of the invasion process, the **invasion** phase, differs from the above explained situations of uncertainty i.n.s. or risk. It is the phase of the invasion process which has been most discussed, since it is at this phase that impacts become inevitably apparent. The main difference lies in the character of an outcome itself. Outcomes of the invasion phase are unfavourable for humans. Although we consider only one species, all outcomes that could occur due to a successful invader cannot clearly be identified. One possible outcome might be that a species fails to become invasive, i.e. it will stay in the state of naturalisation and, therefore, will not be damaging. However, what if the species manages to enter that new phase? Possible outcomes of this phase are often unforeseen. Neither the magnitude of outcomes (i.e. the number of different impacts) nor the spatial extent and the location of the invasion can be anticipated. Especially cascading effects of an impact (e.g. on trophical structures) often remain unclear. Therefore, we have to deal with a ***situation of ignorance*** or ***of uncertainty i.n.s.*** that is we often do not know all outcomes, and there might be some that are outside the set of expected outcomes.

To illustrate the situation of ignorance, consider a situation in which a plant species has managed to reach the state of naturalisation and has now progressed to produce reproductive offspring in a very large number and with extensive ranges. It is not clear whether the species will mainly affect present plant species communities in the new environment due to competition for light and space or if it has effects on pollinators, as it might provide new sources of pollination. Furthermore, it is not apparent whether the given species influences local water availability due to its water consumption or if it influences humans directly by containing properties that directly affect humans. This means a species might have one, several or all of the mentioned effects from an *ex-ante* perspective. Each presents a possible

[18] The exceptional case is the introduction phase (see Figure 10), where it is crucial whether one regards one or all potentially invading species.

outcome of the invasion process. Thus, regarding each phase of the invasion process, we are optimally in a situation of uncertainty i.n.s. if we can expect the magnitude of impacts. Including possible cascading effects (e.g. on trophical structures) makes the identification of each possible impact difficult. As cascading effects often cannot be completely predicted, taking them also into account implies that we are in a situation of ignorance as not all outcomes can be identified.

3.3 Sources of uncertainty *of biological invasions*

In section 3.2.1, the invasion process was depicted in a model. It has been claimed that uncertainty exists regarding the transition from one phase to the next and concerning the temporal extent of each phase. This recognized uncertainty results from a lack of knowledge of invasion mechanisms. Usually it is not known what exactly enables or restricts a species to progress, i.e. promotion or restriction factors are unclear. This is why Heger (2001) states that every invasion is very case specific. Thus, invasion mechanisms are main sources of uncertainty concerning biological invasions. They have to be taken into account when trying to understand the causes of biological invasions and also when developing management strategies to influence such invasion mechanisms. They will be the focal point of any well-aimed strategy in order to minimise invasion damages.

In section 2.2, factors that drive or cause invasions have been explained (see Table 2), as well as a distinction between ecological (i.e. species specific or intrinsic), environmental and anthropogenic factors has been made. Table 2 now relates the different factors of propagule pressure, disturbance, the lack of enemies, open niches, the species home range and its life form, as well as available resources on the new range, to the relevant invasion phase and depicts which factor is most relevant for which invasion phase. These factors are mentioned to be the most predominant factors of invasions (see chapter 2.2.2). However, it is always an interplay of these and other factors that has to be considered in detail to explain invasion mechanisms and to understand what drives or hinders the species' development. Although the dispersal of propagules can be subsumed under propagule pressure for instance, a species propagules´ survival will be also dependent on the size of its seed bank, recipient soil and climatic conditions or special ecosystem properties, such as fire regimes (Brooks *et al.* 2004). This rather simplistic example shows how difficult it is to understand the whole range of patterns and factors that drive invasions.

Table 2: Major factors that influence the invasion process (ref. to Table 1).

Factors	Relevant invasion phase*	Dimensions of factors and related questions		
		anthropogenic	Ecological	
			intrinsic	extrinsic
Propagule pressure	**Int**/Es/Nat	X	X	
Disturbance	Int/**Es/Nat**	X		X
Lacking enemies	**Es/Nat**/Inv			X
Niches	**Nat**/Inv			X
Geographical home range	Int/**Es/Nat**/Inv	X	X	X
Life form	Int/**Es/Nat/Inv**		X	
Use of available resources	Es/Nat/**Inv**		X	

* Int: Introduction; Es: Establishment; Nat: Naturalisation; Inv: Invasion

Phases in bold show that a certain factor is especially relevant for this phase

Although we can relate the different factors to certain invasion phases, there are still open questions regarding the exact and detailed interplay of the various invasion mechanisms. Generally, one can state that invasion biology examines the role of randomness in the progress of a species. This randomness, even if detected, cannot be reduced and will remain inherent to the invasion problem; it is part of irreducible ignorance. However, as part of reducible ignorance, the scientific community assumes that unknown invasion mechanisms are amenable to scientific examinations and can be explained one day. Therefore invasion ecologists address the following questions that are related to driving factors of the invasion process:

- Which set of factors influence the invasion process, i.e. how many and what factors are relevant for a species´ progress?
- Are the factors from Table 2 the most important ones for every potentially invasive species?
- What is the specific set of factors for each potentially invasive species?
- How important is each factor for the invasions process, i.e. which ones are more important than other ones?
- What is the exact interplay of the ecological and anthropogenic factors?

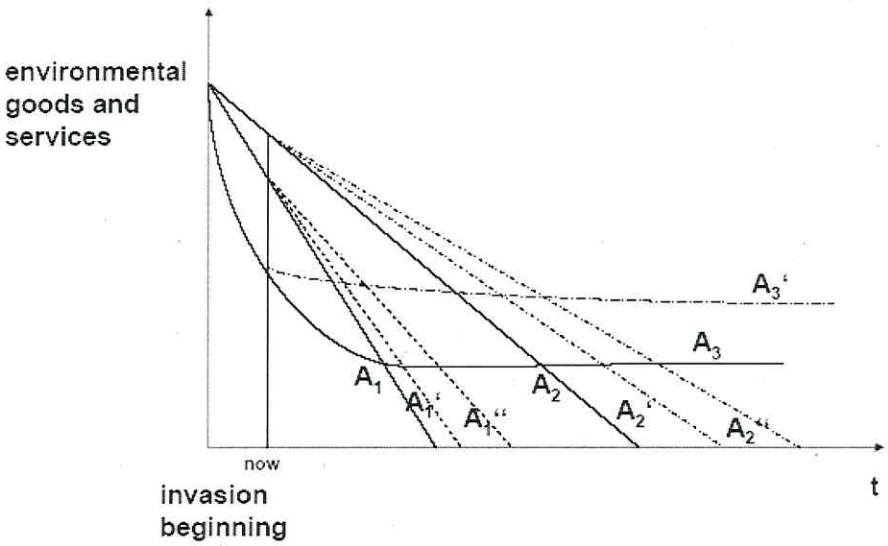
Referring to the open-ended questions from above explains why we are in a *situation of* *uncertainty* when regarding biological invasions′ impacts. We usually do not have probabilities for a species to overcome each phase of the process or when this will happen, as we do not exactly know the interplay of the specific factors that hinder or drive the invasion. This is why time-lags, for instance, are still a challenge to invasion biology research.

3.4 Uncertainty regarding the management of invasive species

As a result of uncertainty about ecological mechanisms that enable the invasion process, forecasting the success of applied measures is increasingly difficult. It is uncertain whether the interaction of mentioned factors will produce a successful invader and how the subsequent impacts will occur in the course of time (see Figure 12: A_1, A_2, A_3). Thus, it is even more difficult to predict how an applied measure will decrease damaging impacts (see

Figure 9: A_1', A_1'', A_2', A_2'', A_3'), if the baseline (A_1, A_2, A_3) is even unknown.

Figure 9: Possible courses of invasive species impact management and corresponding success (own source)



A_1, A_2, A_3: possible courses of an invasion of species A
A_1', A_1'': possible courses of reduced impacts due to management of impact course A_1
A_2', A_2'': possible courses of reduced impacts due to management of impact course A_2
A_3': possible course of reduced impacts due to management of impact course A_3

Acknowledging that "every single process of invasion seems to be unique and for every rule an exception exists" (Heger 2001: 3), generalisations about appropriate measures to counteract impacts of invasions are even more difficult to suggest. Uncertainty is linked with the reducing effect of a measure, that is the assumed (*ex-ante*) and factual (*ex-post*) ability of a measure to reduce the biological invasion or its impacts at hand. The reducing effect of a management strategy is mirrored by its success. This success is closely linked to a strategy's ecological effectiveness[19]. Ecological effectiveness is understood as *the property of a management strategy to achieve the predetermined goal of environmental policy.* It provides information about the success of a measure, as it indicates how far the target, the reference point of a management strategy, has been reached. Targets for all four management strategies (prevention, eradication, control and adaptation) have been explained in section 2.4. The different targets of the different measures refer to the way the invasion problem is approached. Measures can either reduce the invasive species' population (by prevention, eradication, and control), and hence indirectly affect an invasive species´ impacts, or they directly aim at the reduction of a species' impacts (via adaptation) without aiming to reduce the species' population itself.

When using the definition of an outcome[20], one recognises that in the context of biological invasions management the "effect" means to reduce the damaging impact of an invasive species by an appropriate management strategy. The reducing effect will be described by the measures´ ecological effectiveness, i.e. a management options´ reducing property on the invasion.

Considering the different targets of each measure and the different invasion phases for which a measure is suitable (see Figure 4), we can identify two groups of measures:

1. Measures for which a target can only be reached or failed and only two outcomes are possible, according to the measures definition: success or failure. This is the case for prevention or eradication (section 3.4.1).
2. Measures for which the target can be reached more or less, i.e. various outcomes are

[19] Referring to literature, there are other terms such as ecological accuracy (Endres and Ohl 2005), or "ökologische Treffsicherheit" (Feess 1997: 50). Ecological effectiveness has been usually referred to as one criterion in the choice of environmental economic instruments to reach or maintain a predefined environmental quality regarding emissions. In the present context this term will be used analogous to the discussion about the choice of environmental economic instruments.

[20] i.e. any effect (or event) that occurs due to an alien species or its regulating management in the course of the species' development to become an invader. These effects can be described in their character and relevance for human well-being.

possible. This is the case for control and adaptation measures, which aim to reduce the invasion (control) or its impacts (adaptation) as much as possible. A partial success, however, can be still considered as success (section 3.4.2).

3.4.1 Uncertainty regarding the ecological effectiveness of prevention and eradication strategies

For the application of prevention, measures either succeed or fail. Figure 10 depicts both possible outcomes of prevention and eradication strategies for all four invasion phases. As stated earlier, prevention only applies before the introduction phase. Thus, according to the target, prevention fails as soon as one alien species enters the environment[21]. This means ecological accuracy is100%; less would imply failure.

Regarding eradication measures suggested for the phases after introduction, the ecological effectiveness can be either 100% or zero.

Figure 10: Outcomes of prevention and eradication management options (own source)

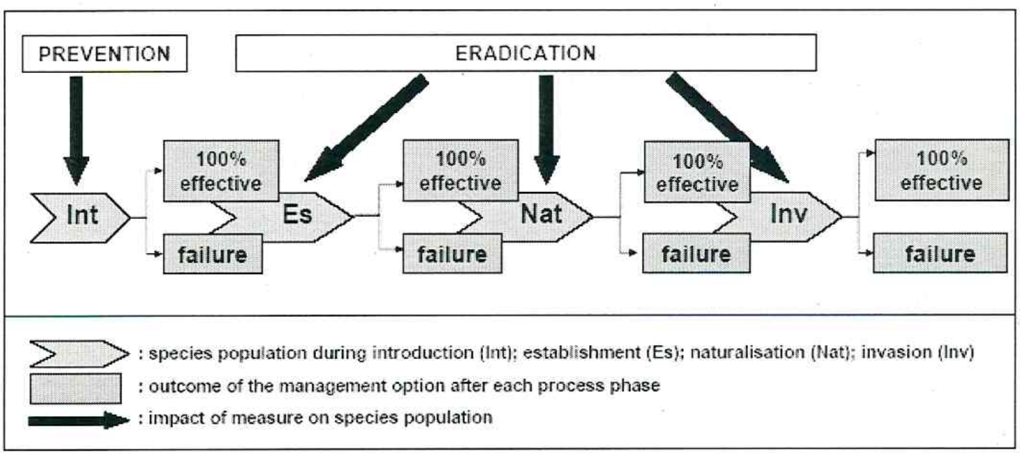


Figure 10 shows that eradication applies for the phases of establishment, naturalisation and invasion. According to its target, this strategy fails if the extermination process of the species' population allows at least one individual of the species population to survive, no matter in which specific invasion phases this happens. If eradication results in a reduction of the species' population only to a certain level, this means failure (with respect to the target).

[21] However, prevention can still have a reducing effect on the species number that might become introduced, although one would say that prevention has failed.

However, if we consider any reducing effect of eradication as success instead, we switch to the case of another applicable strategy, which is control. The opposite outcome of eradication would be if a measure exactly reaches its target: no species survives eradication. This measure would be successful.

This strict understanding of a measure either being a success or failure explains why prevention and eradication often fails in reality. Even islands, such as Australia or New Zealand for which prevention is rather feasible due to its clear cut spatial boundaries, often face incoming alien species, because such measures cannot prevent every species from entering a new range (Bertram 1999).

3.4.2 Uncertainty regarding the ecological effectiveness of control and adaptation strategies

For control or adaptation, the goal is to reduce the species' population (in the case of control) or its impacts (in the case of adaptation) to a certain threshold. Failure means the measure has no reducing effect at all. On the other hand, with respect to prevention and eradication, success is considered as any reduction below the target and above zero. For instance, if the target of control is a reduction of the species' population to 50%, the measure is still considered effective if only a reduction of 30% has been reached although the target itself was not reached.

Figure 11: Outcomes of control and adaptation measures (own source)

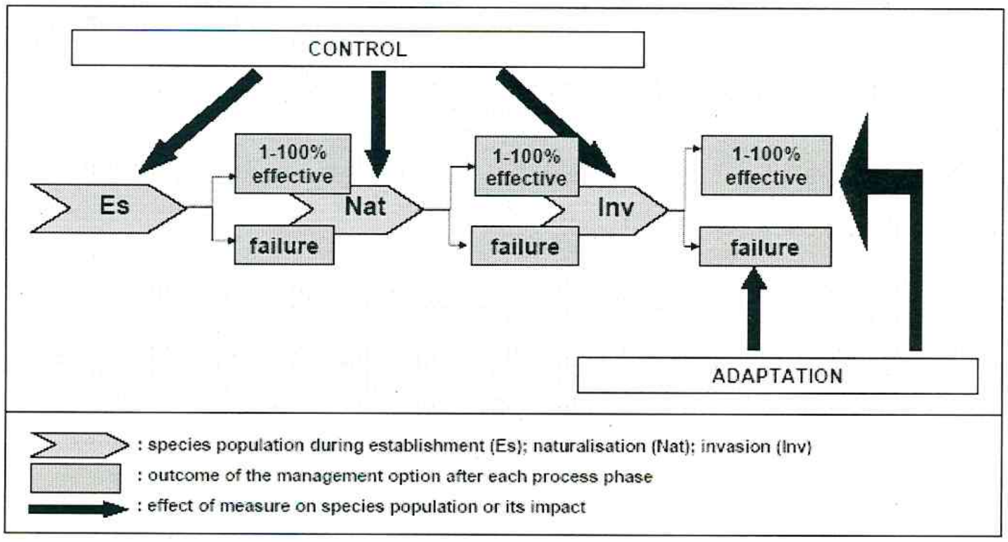


Figure 11 shows the different outcomes of both strategies. They can be between zero (which is failure) and hundred percent effective. As one can see, control applies for the phases of establishment, naturalisation and invasion, whereas adaptation only can be applied for the invasion phase (see section 2.4).

Ecological effectiveness of both strategies, control and adaptation, is therefore not as strict as prevention or eradication. Reduction effects still have value even though the target was not met. Nevertheless, those measures are often unfavourable because even though they are effective to some degree, the effort spent is too high and net benefits are negative. Although there might be some reduction of the invasion and its impacts, too much money has been invested in relation to avoided impacts. Therefore, reaching the goal and gaining complete success is especially important in the application of control and adaptation strategies.

In order to link possible outcomes of all four applicable management strategies with the above explained categories of uncertainty, we are in a ***situation of uncertainty i.n.s.***. Although there are no probabilities for the outcome to occur, we can assume all possible outcomes. In the case of prevention and eradication, there are two possible outcomes. Regarding control and adaptation, there are various outcomes in addition to failure and success; however, all outcomes can be identified.

The situation of uncertainty i.n.s. changes if we take into account that external effects of a measure can occur. As stated above, impacts of invasive species are complex, and cascading effects have been recognised. Hence, management strategies to reduce damaging impacts of invasive species are also complex in their effects. External effects can occur. The majority of such external effects are positive, since a management strategy usually aims at the reduction of the species' abundance. We normally expect that linked impacts will also decrease due to this endeavour. However, we have to acknowledge that we cannot anticipate all external effects of invasive species management.

3.5 *Dealing with* uncertainty *linked to biological invasions*

After having explained why there is uncertainty concerning biological invasions, namely due to unknown invasion mechanisms, and the consequences of these unknown mechanisms on the prediction of impacts and the success of management options, the following section suggests how to deal with uncertainty. For that decision rules are relevant that suggest how decisions can be guided under circumstances of incomplete knowledge (Sigel 2007: 41). The first part of this section introduces common decision rules in economics, and in the second part these rules are linked to biological invasions. By linking these rules to biological invasions this section shows how these rules are applicable in that context, and to what extent they are suitable for recommendations about how to deal with biological invasions, especially in the situation of ignorance which is predominant regarding the whole invasion process.

3.5.1 General recommendations

Investigating current economic literature on biological invasions, one realises that a systematic examination of uncertainty, especially regarding the entire invasion process, is missing. Usually studies concentrate on the invasion phase, on one certain species, and assume a situation of risk. Especially economic studies focus on certain management options for situations conditioned either by the environment or by the invasive species. The majority of studies deal with control strategies and their optimization, subject to specific characteristics of the invasive species, e.g. respiration parameters or specific leaf area (Buhle *et al.* 2005; Finnoff *et al.* 2005; Finnoff and Tschirhart 2005; Leung *et al.* 2005; Finnoff *et al.* 2006). Eradication and prevention strategies are investigated as well (Finnoff *et al.* 2006; Anaman 1994). Such single case examinations tend to neglect complex side conditions under which the invasion process takes place, the multiple effects an invasive species can have and represent a very reductionistic view on biological invasions, as these examinations do not correspond to real world conditions. Such situations of risk enable the preparation of a cost-benefit analysis (CBA), as probabilities are available for the different project outcomes that are planned (see section 4.2).

These observations, i.e. that economic studies tend to have a reductionistic view and assume situations of risk, are in line with general approaches of environmental economics which provide rules under which decisions can be taken if incomplete knowledge concerning an action and its outcomes (in the context of invasive species, impacts of a certain invasion phase) is prevailing. These approaches are common and are based on the fundamental distinction of risk and uncertainty i.n.s., such as introduced by Knight (1920: 233). Ignorance is usually subsumed under uncertainty i.n.s., as it is understood as the most extreme version of uncertainty i.n.s.[22]. Thus, decision rules, that explicitly address the situation of ignorance has

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[22] In the case of uncertainty all outcomes can be enumerated, but not all probabilities for each outcome, whereas ignorance is just a situation where not all outcomes can be enumerated. Thus, ignorance will be referred to as "radical uncertainty" (Perman *et al.* 1999: 431).

usually yet been neglected. The most prominent rules will be now explained[23].

Central elements that are relevant to understand the decision rules are briefly explained and refer to Sigel (2007: pp. 58).

Alternative of action: Alternatives of action, or just alternatives, are those options among which a decision maker will chose. They can be described by objective magnitudes to the decision maker and are often named as decision variables. This means a decision maker will chose among those alternatives he assumes to be realisable at all.

States of nature: States of nature are those magnitudes that are not obvious to the decision maker and cannot be influenced by him, although they influence the outcome of a decision. States of nature are usually depicted in terms of probabilities. Objective probabilities are assigned on the basis of experience (Common and Stagl 2005: 379) or knowledge (Perman *et al.* 1999: 431) and are based on statistical data. Subjective probabilities are used if no objective probabilities are available and refer to the basic assumption that if no experience concerning the occurrence of an outcome exists, at least there is a notion about the occurrence of it and one outcome can be judged to be more probable than another. Thus, on the basis of judgements the decision maker is able to set weights on the outcomes which satisfy the requirements for probabilities (Perman *et al.* 1999: 31).

Outcomes: outcomes are the consequences of an alternative of action. Each alternative will be compared according to its consequences after the decision.

Pay-off matrix: A pay-off matrix depicts the assumption of the decision maker if neither the states of nature nor probabilities for the states of nature are available. The matrix states all alternatives of action, the possible states of nature and the combination of each alternative and each state of nature that are assumed by the decision maker.

3.5.1.1 Decision rules for a situation of risk

The most prominent rules for decisions under risk are (i) the μ-, (ii) the (μ, σ) - criterion, and the Bernoulli-principle.

The **μ- criterion** (also μ- rule or μ- principle) is probably the most well-known and easiest

[23] Mentioned decision rules will not be depicted in detail as there are several books that extensively explain decisions under risk and uncertainty. See, for example, Wätzold (1997: pp. 81), Cansier (1996: pp. 46), (Perman *et al.* 1999: pp. 430).

rule for decisions under risk. It suggests the maximisation of the expected value (also expectation or mean) (Perman *et al.* 1999: 433). The expected value is often referred to as μ and thus also called μ-rule. The expected value of an alternative of action A_l is defined as the sum of the possible outcomes E_{ij} of this alternative weighted with the probabilities $p(S_j)$ of its occurrence. S_j describes all possible states of nature of the alternative.

$$EV(A_i) = \sum_{j=1}^{m} p(S_j) \cdot E_{ij}$$

A decision maker now will determine the expected value for each alternative of action and will chose the one with the highest expected value.

The problem with this rule is that it orientates on the average value, the expected value, but does not take into account its statistical distribution around the expected value, described by the standard deviation (and perhaps higher moments of the distribution). A decision maker who adopts the μ-criterion would be indifferent between an alternative that gives the same outcome under all states of nature and an alternative that gives a high benefit for some states of nature, but a high damage for other states of nature, such that the expected value of both alternatives are the same. Most decision makers would prefer the first alternative to avoid potential damages - they are risk-averse.

A possibility to include the standard deviation of the expected value is the **(μ, σ)-criterion.** σ represents the standard deviation. With this rule the decision maker can act according to his or her attitude towards risk, i.e. whether he or she is risk averse, risk neutral or willing to take a risk. A decision maker who is risk averse will prefer a small standard deviation, whereas someone who is not risk averse is indifferent to changes in σ.

The third rule is the **Bernoulli-principle** which is a more general decision rule than both former rules, which can be considered as special cases of the Bernoulli-principle. It uses the expected utility (EU). According to this rule an alternative will be favoured for which the expected utility of an outcome is maximised. The expected utility U is the sum of the different outcomes (E_{ij}) weighed with the probability of the outcome to occur $p(S_j)$.

$$EU(A_i) = \sum_{j=1}^{m} p(S_j) \cdot U(E_{ij})$$

U is described by a utility function that reflects the risk attitude of the decision maker (Sigel 2007: 62). The Bernoulli-principle constitutes the base for the expected utility theory (ibid). This theory deals with the determination of probabilities and to a large extent with the question how they can be defined.

3.5.1.2 Decision rules for situations of uncertainty i.n.s.

Decision rules for uncertainty i.n.s. have to be distinguished in two categories. On the one hand, there are rules that aim to convert situations of uncertainty i.n.s. into situations of risk and to treat such problems with rules of risk. The most prominent rules of this group are (i) the concept of subjective probabilities, (ii) the Laplace-rule. On the other hand there are rules that approach decisions under uncertainty i.n.s. differently than the first group. Each rule will be explained below. The most well-known rules of the second group are (iii) the Minimax-regret rule, (iv) the Maximin-strategy and (v) the Hurwitz-criterion[24].

Such as explained above, the **concept of subjective probabilities** follows the assumption that we usually have at least an idea whether one outcome of an alternative of action will be more probable than another. Thus people usually do have subjective judgements on the distribution of probabilities (Wätzold 1998: 90). To gain data experts are often questioned in order to increase the credibility of the expectations (ibid). Subjective probabilities are then treated as objective probabilities. The concept of subjective probabilities is the standard model for decisions under uncertainty i.n.s. in economics (Cansier 1993: 46).

If there is no idea for a subjective probability for each outcome at hand, that allows a subjective prioritisation of the outcomes, the **Laplace-rule** can be applied. It uses the same probability for each of all possible outcomes, i.e. a uniform probability distribution (Wätzold 1998: 87; Cansier 1993: 48). These probabilities are then handled as being objective and rules of risk are applicable.

The **Minimax-Regret rule** is one of the rules that acknowledge that a conversion of situations of uncertainty i.n.s. into a situation of risk is often not possible. Thus, it prioritises another strategy, namely the minimisation of the outcome with a content of maximal regret. This rule is also known as the Savage-Niehans rule (Wätzold 1998: 93). Table 3 shows a pay-off matrix

[24] In addition there exist generalizations of expected utility maximization that account for non-additive probabilities, e.g. Choquet expected utility (Schmeidler 1989).

and illustrates how the rule works.

1. The maximal regret-values are created. This happens by subtracting each outcome z_{ij} (the number before the →) of a row from the maximum $\max_i E_{ij}$ of the row.
2. From these values (the number after the →) the highest value will be chosen and depicted in the maximal regret-value row. This number shows the maximum regret-value of each alternative of action.
3. Out of the maximal regret-values, the minimal one will be chosen, which in the present case, implies a preference of A_2.

Table 3: Pay-off matrix of regret-values for the Minimax-Regret rule (source: Wätzold 1998: 94)

Alternatives of action	States of nature			Maximal regret-values
	B_1	B_2	B_3	
A_1	20 → 40	50 → 0	70 → 0	40
A_2	60 → 0	30 → 20	40 → 30	30
$\max_i z_{ij}$	60	50	70	

The **Maximin-strategy** favours the alternative of action with the least bad worst outcome, independent from the outcomes´ probability to occur (Wätzold 1998: 92). Table 4 shows how this rule works.

Table 4: Pay-off matrix of values for the Maximin-strategy (own source)

	N_1	N_2	N_3	N_4
A_1	2	2	0	1
A_2	1	1	1	1
A_3	0	4	0	0
A_4	1	3	0	0

1. In a matrix all possible states of nature (N) and all possible alternatives of action (A) are depicted. For each line (alternative of action) the smallest value will be chosen. (For A_1 it is 0 (N_3); for A_2 it is 1 (N_1-N_4); for A_3 it is 0 (N_3); for A_1 it is 0 (N_1,N_3 ,N_4)

2. Out of these values the highest one will be chosen and thus the alternative of action with the maximal value for the least unfavourable outcome can be determined (here: A_2 (=1)).

The Maximin-strategy has been criticized for ignoring most of the information on the pay-off matrix. It is based on the assumption that decisions are made under the most unfavourable options. This means if one alternative of action is marginally better, i.e. its outcome is only slightly better than the worst one, the worst one will be chosen no matter how much more preferable the second worst is under all other states of nature (Perman *et al.* 1999: 445).

The last rule mentioned here is the ***Hurwitz-criterion***. Due to the fact that this rule concentrates on a weighed combination of the most pessimistic and the most optimistic outcome, this rule has been also called the "pessimism-optimism-criterion" (Cansier 1993: 47). By putting a weight on the most optimistic (with the weight 1-α) or pessimistic (with the weight α) outcome the decision maker can express his or her attitude towards uncertainty i.n.s.. The weight of both values is usually between 0 and 1. This means, if a decision maker, for instance, puts a value of 0.8 for α, and 1-α is then 0.2, he or she favours smaller values and is of rather pessimistic nature. The best solution according to this rule would be the alternative of action with the weighed maximal benefit (Wätzold 1998: 95).

3.5.2 Specific suggestions for biological invasions

As we can see, with the rules mentioned above, there are several approaches available that can guide decisions in different situations of uncertainty. They all aim at providing an optimal solution if harmful results of an action are expected (Wätzold 1998: 100), i.e. the outcome of an action which is least damaging to people will be favoured. To put these rules in the context of invasive species, one will wonder how far these decision rules are useful for decision-making regarding biological invasions and their management. Next to the question whether it is possible to aim at an optimal solution under uncertainty, one will wonder if these rules are appropriate for the situation of ignorance, which has not been discussed so far. The general reflection how uncertainty can be generally handled and reduced regarding biological invasions and their management will be subject of this section.

3.5.2.1 Scientific research to reduce uncertainty

A general way to reduce uncertainty is the generation of knowledge. To address questions derived from Table 2 are a good starting point for scientific research. They refer to the mechanisms that drive the invasion process and have been discussed in section 2.2 and 3.3.

Anthropogenic and ecological factors enable or hinder the invasion process, and are responsible for the progress of a species. They are subject of scientific research. Such questions are, for instance, which set of factors do influence the invasion process, i.e. how many factors are relevant for a species´ progress, or what role does randomness play in the invasion process? Answers to these questions would be a first step to convert communal ignorance into uncertainty i.n.s. or even risk. Communal ignorance is prevailing because mechanisms that enable an invasion of alien species into a new range are usually unknown. The conversion of ignorance in situations of less uncertainty means much effort has to be spent to gain detailed knowledge for individual species, if the aim is to provide objective probabilities for invasion outcomes to occur. Next to communal ignorance, where we as a society can generate knowledge that has not been available before, personal ignorance can be reduced by starting information campaigns which are based on knowledge received through research and science. Dispersing and making societal knowledge accessible means to enable subjective probabilities for lay people. Thus individual decisions regarding the handling of biological invasions then are not only a task of public decision-making.

However, although there are reasons to believe that with science we are able to convert communal ignorance into uncertainty i.n.s., risk, or even certainty, there will be always some aspects of irreducible ignorance, because the increase of knowledge regarding biological invasions emerges on the edge of the knowledge of a society. "As long as we do not know for sure that our ignorance has to be interpreted as irreducible, we are to a certain degree entitled to hope that it will turn out to be reducible" (Faber *et al.* 1992. 227). To acknowledge this fact is of increasing importance when taking humans as drivers for biological invasions into account. Human behaviour will not be amenable to pure biological science, as it is determined also by social relations and habits, which are outside the scope of ecological processes and even harder to understand (Faber *et al.* 1992).

3.5.2.2 Decision rules related to economic assessment tools

As this thesis regards the enhancement of ecological-economic assessments in order to improve decision-making concerning the management of invasive species, typical economic valuation tools are now connected with the different types of uncertainty. The decision rules from above are suitable for different economic valuations methods.

Table 5 links the decision rules for the different situations of uncertainty with respective economic valuation methods and shows that certain methods suit only for specific situations of knowledge. To generating knowledge, of course, is essential part of all methods and helps to handle the specific situation of incomplete knowledge respectively. For a *situation of risk,* we can apply typical environmental economic valuation methods that aid decision-making, such as cost-benefit analysis (CBA) or cost-effectiveness analysis (CEA). Although they are usually applied for facts that are certain, i.e. in the case of biological invasions for impacts that have been factually recorded and are quantifiable, they also work for a situation of risk. How the different valuation tools work will be explained in chapter 4.

By focussing on individual aspects of one certain invasive species, probabilities for the success of an applied management strategy and its capacity to reduce damaging impacts of the species might be available. As soon as more than one invasive species is under focus, to apply rules for situations of risk become difficult as probabilities are lacking.

Table 5: Overview of decision rules under uncertainty and related economic valuation tools (own source)

Situation of uncertainty	Decision rule	Typical economic valuation methods	
Risk	μ- criterion	▪ CBA ▪ CEA	Generation of knowledge (as strategy to reduce uncertainty)
	(μ, σ)- criterion		
	Bernoulli-principle		
Uncertainty i.n.s.	Laplace-Rule		
	Subjective probabilities		
	Maximin-criterion	▪ Sensitivity analysis ▪ Scenarios	
	Minimax-regret rule		
	Hurwitz-principle		
Ignorance	Precautionary principle		

Throughout this chapter it has become clear, that this is often the case with biological invasions.

For a *situation of uncertainty i.n.s.* certain well-known tools of environmental economic evaluation are also applicable, namely if rules allow the conversion of a situation of uncertainty i.n.s. into risk. This is possible for the Laplace-rule or if subjective probabilities are available. Furthermore it is appropriate to work with scenarios or with sensitivity analysis. This implies making a decision based on expected minimum and maximum scenarios of either the damaging potential of an invasive species or the beneficial potential of a management strategy, e.g. Pearce *et al.* (2006: 61). This approach enables the handling of uncertainty i.n.s. linked to biological invasions without having probabilities.

But, what can be suggested for a *situation of ignorance*? The statement from Heger (2001: 3) that "every single process of invasion seems to be unique and for every rule an exception exists", underlines how scarce knowledge about biological invasions is and that often not even subjective probabilities are at hand. Regarding the entire invasion process we have seen that for the introduction phase as well as for the invasion phase decisions usually have to be made under ignorance. Table 5 illustrates that typical environmental economic valuation tools are not suitable for ignorance. For this type of uncertainty even subjective probabilities are not available. If neither magnitude and number of impacts are known *ex-ante,* nor the probability of the impacts to occur, the precautionary principle seems appropriate. In order to avoid an impact that might be disastrous and is just outside the set of expected outcomes at the moment, one would follow the logic to avoid such an impact from the very beginning. In such a situation, the economic calculus of regarding costs and benefits is not fundamental anymore. What is important is the strict prevention of any damaging impact.

3.5.3 Conclusions from the analysis of uncertainty

In conclusion, scientific research or to say it in a rather generalised way, the generation of knowledge is one important approach to reduce uncertainty. While we wait for scientific research to answer questions raised, the question as how to deal with uncertainty in the meantime remains, as decisions about the appropriate management of biological invasions have to be made soon. The second suggestion is to apply depicted decision rules and economic valuation tools as far as possible. However, the question surfaces when to apply a CBA or CEA, when the precautionary principle and when to apply the scenario approach.

To answer this question, we have to consider the different phases of the invasion process. They determine the respective decision context of the suitable management option and whether one or several species have to be subjected to examination.

During the **introduction** phase, the application of the precautionary principle should be aimed at, since for this phase of the invasion process strategies always focus on every potentially incoming species. It is not possible to focus on a single species during that phase, especially for decision-making that aims at a general goal of management of all invasive species for one target region. Such nationwide strategies are often demanded by the government (Bertram 1999). Due to this, it is not clear which species will be the one that will go beyond its old range. This means that regarding the baseline "who and when" we are in a *situation of ignorance*.

For the phases of **establishment** and **naturalisation,** we are usually in a situation of uncertainty i.n.s., as we focus on a single species development and know possible outcomes, failure and success. A single-species focus, however, does not imply that only certain traits and certain environmental conditions are analysed, such as revealed when surveying current literature. An explanation of ecological and anthropogenic mechanisms will be needed to understand the complexity of the invasion of one certain species. Thus, the application of scenarios will support decision-making.

For the phase of **invasion**, we are in a *situation of ignorance* if we are concerned with the magnitude of possible damaging impacts of an invasive species. Ignorance especially occurs in the context of cascading effects due to one initial impact that triggers several other, maybe hidden impacts. As a consequence of uncertainty regarding the invasion mechanisms, we hardly can predict the success of a measure. Following the suggestions from above, we should try to apply the precautionary principle. Due to high irreversibility during that phase, however, the use of this principle is not feasible. Therefore, other suggestions have to be provided, especially if economics should deliver a contribution to decision-making, which is not feasible for the precautionary principle.

As stated above, decisions have to be made very soon and decision rules from above very often they fail because there are no probabilities at hand, neither objective nor subjective. This demands a third suggestions, namely to make uncertainty at least explicit. The focus of any economic assessment that aims to aid decision-making is, of course, to provide optimal solutions. But in the case of biological invasions such an aim very often is not possible due to the uniqueness of every invasion (see Heger 1001). Especially for a situation of ignorance, for which no decision rules are at hand, we should refrain from finding an optimal solution. If we acknowledge that we are in a situation of uncertainty or even ignorance, and if uncertainty has been made explicit, economic assessments still ought to be part of the decision-making process. To explicitly mention the different types of uncertainty, would make economic assessments to real ecological-economic assessments. Such an explicit mentioning means to acknowledge that regarding the phase of invasion where damaging impacts of invasive species become apparent, neither the whole range of possible impacts can be anticipated nor that all feedbacks between occurring impacts can be recorded, especially the ones occurring due to cascading effects of impacts.

The analysis of uncertainty linked to biological invasions has shown that there at the moment there are limited possibilities to anticipate impacts of biological invasions. The comprehensive examination of the different types of uncertainty is important to understand biological invasions as a problem and to design effective strategies while acknowledging that coming economic decision rules often do not work. An important step, however, is to make uncertainty linked to the ecological properties of biological invasions explicit. How economic properties of the problem are considered, will be explained in the following chapter.

4 Economic valuation: foundations and application to biological invasions

If economics can support decision-making regarding the choice of appropriate management options it is necessary to shed light on its foundations and the way economics rationalises why and when costs or benefits occur.

This chapter deals with foundations of any economic assessment of environmental goods. The focus within this chapter is on the second step, the economic valuation, because the economic value of an environmental good is the central argument in a political decision context. The physical assessment as first phase of economic assessments is of minor role in this chapter. Economic valuation is also relevant regarding biological invasions. Whether and how one values impacts of an invasive species will influence a decision concerning the management of an invasion. Thus, this chapter not only deals with core assumptions of welfare economic valuation (section 4.1), section 4.2 shows the role of economic assessments of biological invasions. Section 4.3 depicts cost-benefit analysis (CBA) as a prominent decision tool and section 4.4 explains available valuation techniques for environmental goods. Section 4.5 introduces the Total Economic Value (TEV), which is a concept that helps to structure and include all environmental goods that are relevant for an economic valuation. The last section depicts a series of critiques that have been referred to the welfare economic valuation in principle, but also in the special context of invasive species. In the end of this section challenges for economic assessments as a whole are outlined with special regard to biological invasions.

4.1 Welfare economic aspects of biological invasions

4.1.1 Individual preferences as source of economic value

The economic valuation of the environment and its provided goods has its origin in neoclassical welfare economics. Its basic objective is "to formulate propositions by which we can say that the welfare" of individuals, people, or the society as a whole, "in one situation is higher than in another" (Ng 1983: 3). Approximate synonyms for social welfare are the concepts of happiness, utility, benefits or well-being (Fischer 2003: 19). Welfare economics assumes that every economic activity aims to maximise utility, which can be explained as the optimal satisfaction of interests of human individuals. It is always based on individuals, their needs, wants and interest, the so called methodological individualism

(Hansjürgens 2004: 242). Thus, every decision is taken from an anthropocentric view (WBGU 1999: 49). Society is understood as the sum of all individuals. Whether an action supports welfare or not is indicated by individual choice, which is the precondition for any action and that reflects individual preferences[25]. They are to be taken as the source of value. Individual preferences become obvious in every choice when certain goods are preferred (i.e. chosen) vis-à-vis other goods and services. Costs are understood as reductions in human well-being, benefits as an increase of it (ibid).

To make the different levels of welfare comparable[26] and determinable, economists such as Pigou (1920) used money as an appropriate reference unit that enables the transformation of welfare into a comparable format. The transformation of welfare into monetary units assumes a trade-off between a preferred good and the certain amount of money an individual is willing to pay. The so called paradigm of substitutability assumes that every good can be substituted by a certain amount of money (Fromm 1997: 97; WBGU 1999: 62). Money and the consumption of goods are perfectly interchangeable as more of them is assumed to increase individual welfare. The fact that an individual is willing to pay for a preferred good is called willingness to pay (WTP). It is the "standard measuring stick of benefit in economics" (Hussen 2000: 290). WTP reflects the preference for a benefit and a willingness to accept (WTA) the cost (Pearce *et al.* 2006: 27). Du to the assumption that society is simply the sum of individuals, social benefit or cost is simply the aggregation of individual benefits or costs (ibid).

4.1.2 Private and public goods

The satisfaction of human wants to increase welfare is gained via the consumption of goods. Economics essentially distinguishes between market and non-market goods. Normally private goods are traded in markets. For the majority of environmental amenities, however, there are no markets for goods at hand and no prices that reflect the value of a good in its character to contribute human welfare. This refers to the public good character of such assets and makes them different from market goods. Obviously, the consumption of a public good, such as the

[25] Preferences and interests are used as synonyms in the following.

[26] Utility in economics orientates on marginal changes in the availability of goods, i.e. it focuses on shifts in welfare by considering how far one additional unit of a consumed good increases welfare (Hampicke 1999).

[27] The idea of WTP and WTA can also include the WTP for a cost to be avoided, and WTA be compensated for a forego benefit.

prominent example of clean air, benefits individuals and increases welfare for them. The action of consuming clean air is characterised by non-exclusiveness on the one hand and non-rivalry in consumption on the other hand. Both criteria are common in the identification of a pure public good (Ostrom and Ostrom 1977). "Exclusion occurs when potential users can be denied goods or services unless they meet the terms and conditions of the vendor" (ibid: 76). If the consumption of one person precludes another, there is no jointness in consumption, i.e. there exists no rivalry, and the consumption by one individual does not affect another individual's potential for gaining benefits through consumption. Due to the public good character of such good, there are no existing markets for the resources to be allocated efficiently.

Closely connected with the concept of public goods is the notion of externalities. Externalities are "conditions arising when the actions of some individuals have direct (negative or positive) effect on the welfare or utility of other individuals, none of whom have direct control over the activity" (Hussen 2000: 98). They can occur if private goods have a joint effect on the welfare of other individuals, and this joint effect is not reflected in the price of a private good, which causes an externality due to the goods production or if there are no private goods to which externalities can be attributed. Such externalities can have positive (beneficial) as well as negative (damaging) effects on individuals that are not involved in the market transaction of the good that causes the joint effects. Positive externalities are conventionally called external benefits, and negative externalities are understood as external costs. Positive effects occur, for instance, if farmers sustain a unique landscape by cultivating their land. As a joint product of the cultivation of crop and fodder, the landscape characterised by fields is also source of enjoyment for tourists. The positive externality for tourists will not enter the market price of fodder and crop. This is the reason why farmers usually do not take this aspect into account while producing crop and fodder which leads to the fact that such landscapes are too less produced for society. Negative externalities are known from the production of energy. With respect to energy, harmful emissions occur as joint products of the energy production for which the producing company is not responsible. Costs occur insofar as such emissions can cause health problems that have to be carried by third persons, i.e. the damaging costs of energy production are externalised.

For certain environmental amenities it can be observed that although the application of the criterion of non-excludability is possible, there is competition in use and overexploitation can be the result. Examples for that can be the overexploitation of fish stocks in the oceans. This is commonly discussed as *open access* (Garrod and Willis 1999: 18). The reason for that is the lack of ownership rights over the resource. There are also cases known where endangered species have been overexploited, such as whales. Common agreements of all interested parties or the intervention of the government that assumes property rights can lead to a solution of the issue, thus the externality can be reduced or abolished.

4.1.3 The public good character of biological invasions

Human welfare is affected in several ways from biological invasions. From the economic point of view, they are an example of an environmental issue with a public good character (Perrings *et al.* 2002). In terms of "goods and bads", the circumstance of biological invasions is "framed as the management of an impure public bad", as impacts of biological invasions are considered more damaging than beneficial in nature (Finnoff *et al.* 2005: 369). Thus, related externalities of biological invasions arise in the context of humans that deal with invasive species in terms of managing them before or after introduction. The above mentioned criteria of exclusion and jointness in use fit in that context (Ostrom and Ostrom 1977). The application of both criteria reveals that neither the exclusion from experiencing positive effects of management strategies nor the refusal of jointness in use is possible. An appropriate example is the control of *Ambrosia artemisiifolia*. As a result of this control, the content of allergic pollen in the air decreases. An undefined number of people will benefit from the control measure through open access to the benefit and thus the success of such a measure is a benefit in terms of reduced impacts and is an external effect of the management of an invasive species (Perrings *et al.* 2002). The management of biological invasions, therefore, can imply positive external effects or, in other words, external benefits as the repercussions of human activities (namely the management) influence the well-being of others (Feess 1997).

On the other hand, invasive species management can also have negative external effects, for instance if pesticides or biological control agents are used in control. They can have effects on non-target species, as known from biological control of pest management (Tisdell 1990). In this circumstance, species are affected, not humans. However, if those species are of human interest they also affect humans, and such an impact on a non-target species would be an indirect, negative external effect. The application of pesticides might affect the local fresh water which, in turn, reduces fresh water quality relevant for local communities.

However, external effects of biological invasions´ management are different from what conventionally is understood in economics as externalities (Perrings *et al.* 2002). The common economic understanding assumes an identifiable source of externalities, a continuing flow of such and the possibility to stop the externalities. The difference stems from the self-perpetuating character of the invasion process which is continuing, increasing and usually irreversible (Perrings *et al.* 2000). The main problem arises regarding the introduction. Even if a deliberate introduction has had a positive intention, (e.g. an alien plant has been imported for agricultural usage) and an initial source of introduction might be identified (e.g. the trading company that imported the species), an invasion which has started because the species escaped from the cultivation site and developed to an invader, respective damages will continue although the initial source has been known. A well-known example are contaminated vessels that accidentally introduced several of the 145 invasive species into the Great Lakes (Horan and Lupi 2005). The subsequent invasion and its responding impacts did not cease with the vessel identification. In the majority of cases we have to deal with irreversible facts, if a species has overcome the phase of naturalisation. The introduction of an alien species can usually not be reversed. (Andersen *et al.* 2004).

So, what is problematic regarding the public good character of biological invasions? It is the lacking incentive for private actors to take actions against damaging impacts of invasive species. Personal engagement in the management of biological invasions will be low as the engaged person will take the burden of management costs, and other people will benefit from this action due to the missing excludability and rivalness in use (as shown in the example of the management of *Ambrosia artemisiifolia*). People that are not engaged in the control or eradication of harmful invasive species maximise their individual benefit by not sharing the costs for invasive species′ management and hence, the management of biological invasions is usually not provided by private actors. This fact calls for governmental interference, such as demanded by the CBD, in order to encounter biological invasions and its damages.

4.2 The role of economic assessments of biological invasions

No matter if private or public actors intend to take action against invasive species there is always the question what will be the best or optimal strategy to encounter the invasion in order to increase private or social welfare. Selecting the optimal management strategy is often difficult. One reason lies in the rising costs for management options in the course of the invasion process and its increasing irreversibility. If one out of 100 introduced species becomes invasive, early-stage management implies useless investment aimed at controlling a great number of species which will not become invasive anyway. Late-stage management often fails because the invasive species has proliferated too much to be controlled.

While the CBD suggests certain management strategies for respective invasion phases and requires early-stage management, Art. 8h also provides advice for the decision concerning the choice of management option(s). The CBD advises that "each contracting party shall, **as far as possible and as appropriate** prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species" (CBD 2002). Therefore, the CBD not only calls for action to counter invasive species and offers a set of management strategies, the convention also demands that management options ought to be taken as far as possible and as appropriate. This leads to the question of what appropriateness of management options means.

4.2.1 Costs and benefits of biological invasions

One way to assess the appropriateness of management options is to focus on economic costs and benefits. On the one hand there are costs and benefits of impacts of invasive species, on the other there are costs and benefits associated with management strategies. In economic terms costs in general arise due to a foregone use ("opportunity costs"), and benefits are created if the use is captured. How this becomes important in the context of invasive species will be explained in this section.

Costs and benefits of invasive species impacts

Impact costs are the quantification of the damages caused by invasive species to humans or ecological systems, expressed in monetary terms. An example is the loss of crop yield due to the presence of an invasive weed species expressed in € per ha. Benefits due to invasive species occur if the species can be directly utilised or is somehow favourable for humans. An increasing amount of utilisable firewood due to the increasing number of invasive *Acacia* species in the Cape Floristic Region is one example for such benefits (Le Maitre *et al.* 2002).

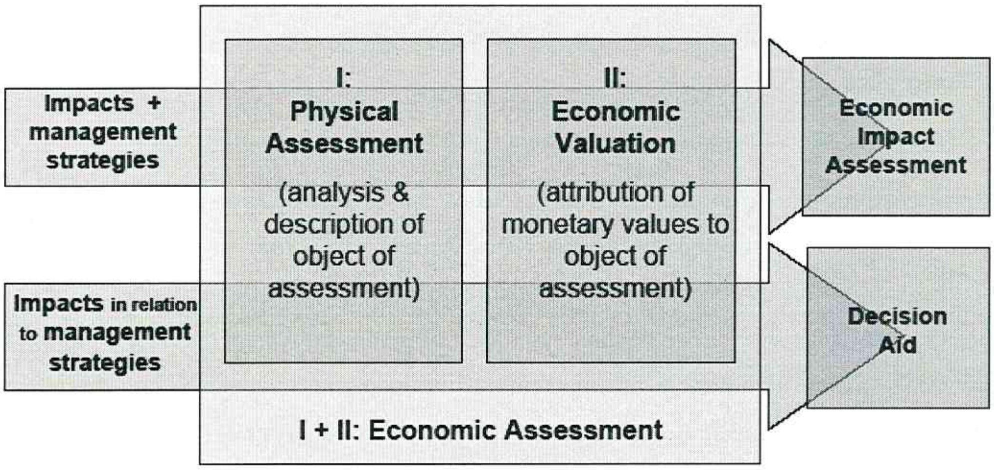
Costs and benefits of management options

Costs regarding management options are due to the provision of machinery, equipment, labour and products such as pesticides which are necessary for the measure. Benefits of management options arise due to a decrease in the damaging impact caused by an applied management option. Thus, benefits are created due to reduced (damage) costs, i.e. they are negative-impact or damage costs. An example is the control of the above mentioned invasive *Acacia*-species. Due to its high water requirements the species reduces the freshwater available to local communities by taking up water from river flows in South Africa. Control measures allow a reduction of the damaging effect and provide a benefit by increased freshwater availability to the local communities (Le Maitre *et al.* 2002).

4.2.2 Assessments of costs and benefits

For economic assessments of the costs and benefits of invasive species' impacts and respective management options the two-step procedure explained in the introduction is necessary: first a physical assessment, and second, an economic valuation. Figure 12 is a slight modification of Figure 1 from the introduction. The difference between both figures is to show the two distinct outcomes of economic assessments. Although each approach needs (i) a physical assessment and (ii) an economic valuation (see introduction), the emphasis of the following considerations are on the economic valuation, since both approaches argue with costs and benefits as aspects of monetary valuation[28].

Figure 12: Objects of economic assessments and corresponding decision-aid results (own source)



The two general approaches are explained in the following. Both procedural approaches are distinguished for realising an economic valuation regarding biological invasions, with the aim of supporting decision-making:

1. The term economic impact assessment will be used if impacts of invasive species and their gross costs to society are considered. Impacts measured as gross costs are a rough

[28] Due to the systematic disregard of the physical assessment, and thus of ecological properties of biological invasions, the procedure now will be only referred to as economic assessment.

indicator for the urgency of action, as they indicate the problematic character of biological invasions for society in terms of high costs.

2. If decisions have to be made regarding the appropriateness of a management option, the costs of invasive species' impacts in relation to the costs of the management strategy have to be included. Therefore net benefits are considered. This economic assessment approach aims to deliver information about the proportion of impact and management costs and will be referred to as decision aid in the following. Decision aid corresponds to the demanded clause on appropriateness of management strategies in Section 8h of the CBD as it assumes the *appropriate* measure provides the best proportion of net benefits.

Both assessment types are explained in the following sections.

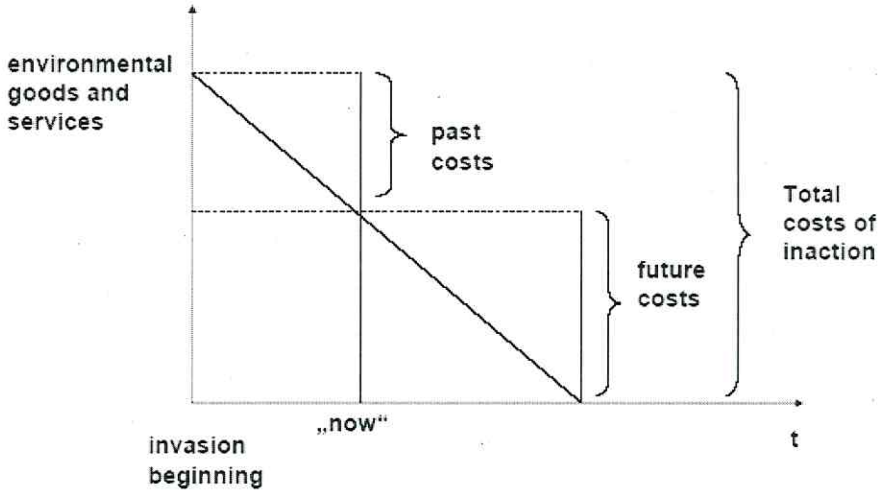
4.2.3 Economic impact assessment: assessing gross costs

Economic impact assessments have been used for the recognition of the damaging potential of biological invasions. As starting point they use a moment in time in which policy was inactive. This is why such an assessment has also been called COPI - Costs Of Political Inaction (Bakkes *et al.* 2006). Inaction in this context means "effectively the counterfactual (or reference), from which the benefits of different public policy and other actions can be evaluated" (ENV/EPOC 2007/6: 4). This type of economic assessment is usually applied early in the policy lifecycle, with the aim of identifying the most pressing environmental problems by regarding their gross costs to society.

An economic impact assessment accounts for gross impact costs of political inaction, i.e. no well-aimed intervention was implemented before the assessment. It assumes a business-as-usual approach and considers damages caused to society by the decrease in the provision of environmental goods.

Political inaction can have two reference points: (i) no policy whatever has been implemented to address the management of invasive species, and (ii) if a revision of the existing policy framework is necessary as existing policy has been ineffective (ENV/EPOC 2005). Thus, there is no absolute notion of "inaction" as both reference points are used for the baseline of the assessment, i.e. in both cases policy has not reacted appropriately to the challenge of reducing impacts, so that either any policy at all or a revised policy was necessary. Regarding the baseline we distinguish (i) economic impact assessments *without any* policy (see Figure 13), and (ii) economic impact assessment *with some* policy intervention (see Figure 14).

Figure 13: Economic Impact Assessment: damage costs in the absence of *any* political action (own source)

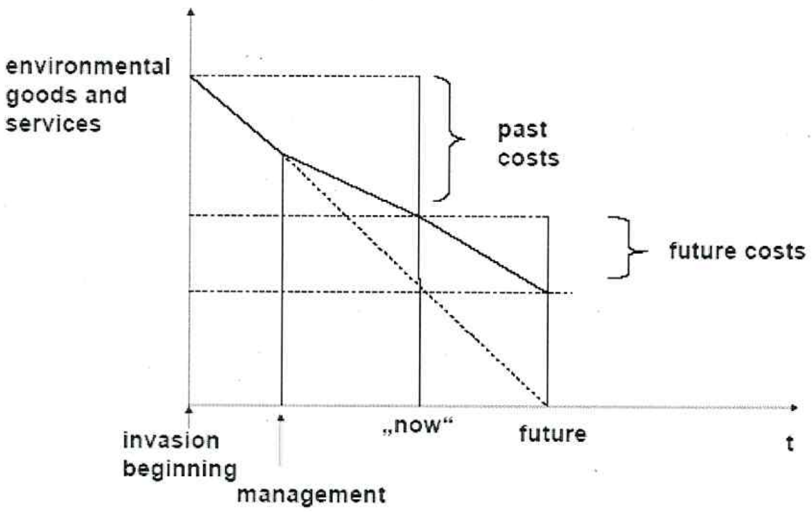


On the axes of Figure 13 environmental goods and services (y-axis) and the time (x-axis) are depicted. The figure shows that the provision of environmental goods and services decreases with an invasion over time. The loss of the environmental goods and services has to be understood as a foregone use of the services due to the invasion, which corresponds to costs. If an assessment of these costs happens at a certain point in time (in Figure 13 "now"), there are costs that already occurred due to the invasion in the past (past costs), and costs that will occur in the future if nothing will be done against the invader (future costs). Both aspects are the total costs of inaction. So COPI tells the decision maker that a considerable degree of costs can be avoided, namely all future costs, if he or she will interfere in or cease the ongoing invasion.

A good example for the first case, in which no political action has yet taken place, is the reduced production of timber in forest ecosystems due to the proliferating thickets of black cherry (*Prunus serotina*). This invasive species affects ecological processes by disturbing the natural undergrowth of forest ecosystem and imposes costs to society (Reinhardt *et al.* 2003: pp. 37). Dense thickets of *Prunus serotina* hinder fir and spruce in their growth, as they fail in the competition with the invasive plant for space and light. Invasive species can also directly affect humans, for instance if a species destroys human-made facilities, as it is the case of the Zebra Mussel which damages dams and water pipes (Aldridge *et al.* 2004).

An economic impact assessment can also refer to a baseline that includes some intervention (see Figure 14). This second type considers damage costs in the absence of *revised* policy, as current policy still implies future impact costs.

Figure 14: Economic Impact Assessment: damage costs in the absence of *revised* political action (own source)



This means that at some point in time policy to manage biological invasions was implemented (the point at the time-axis when management starts), which lead to a reduction of impacts, which, however, was not complete or sufficient. (This is why the line in Figure 14 after the implementation of management is less steep than before. The dotted line would happen if no intervention would have happened at all, such as in Figure 13). Total costs of such an economic impact assessment account for past impact costs (costs before management) plus costs of current management options, as it refers to a baseline with *some* political action (management costs are not marked in Figure 14). Due to the fact that certain damaging impacts still occur in spite of the existence of some management, a revised policy is required to reduce the future costs implied. (This is depicted in Figure 14 by the curve bowed downward. If there are no impacts anymore and the management is successful in terms of a total reduction of impacts, the line is an asymptote to the time-axis). Although future impact costs have been decreased due to the implementation of management strategies, there are still some due to the insufficient management at the point of the assessment ("now" in Figure 14). Avoided impact costs which are the positive result of the management, are not of interest for a

COPI-study as it only considers gross costs.

The effect of such economic impact assessment studies is to raise awareness and to illustrate the high social costs of biological invasions. Because it presents the costs due to impacts of invasive species and those generated by invasive species management, the contribution to decision-making lies in illustrating all costs society faces due to the biological invasion. Damage costs usually either account for costs due to a number of invasive species in a certain region or certain ecosystem (Le Maitre *et al.* 2002; Pimentel *et al.* 2002), or account for the costs one single species causes through its various impacts (Barbier 2001).

Although such studies help raise awareness and are often an essential argument to launch the political call for action[29], they cannot support decisions concerning the choice of management options, as they neglect net benefits of management options that reduce invasive species impacts. By accounting for gross costs of different species, such studies can help identify the "costly" species by comparing the total sum of damages. However, this identification does not answer the question of which strategy is the appropriate one to counter the impacts of invasive species.

4.2.4 Decision aid: net benefits of management options

Studies dealing with both costs and benefits are defined as decision aid, since they aim to calculate net benefits of invasive species management. This second type of economic assessment is usually applied at a later stage of the policy lifecycle because it proceeds through the evaluation of different management alternatives (Bakkes *et al.* 2006). The typical environmental economic tool that corresponds to this approach is cost-benefit analysis (see section 4.3). Decision aid is concerned with the marginal net benefits of management options when impacts of invasive species are reduced, and considers avoided impact costs and its proportion to respective marginal management costs. It supports decisions concerning the choice of the appropriate strategy because it is concerned with net costs and benefits. Based on this, a decision about the appropriate strategy can be made. Only if management costs are lower than impact costs, one would argue for any action at all. Strategies can either target the invasion itself, via prevention, eradication or control, or mitigate impacts via adaptation strategies (see section 2.4).

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[29] The study of Pimentel (2002) is certainly the most prominent example. Estimated costs for the world incur a sum of $1.4 trillion per year for damages and management. A review of invasion literature shows that this study is very frequently cited.

Decision aid can have two different foci, which are usually hierarchical:

(i) Selection of the optimal measure for **one** target species out of a set of measures. This assessment helps identify the most cost-efficient strategy out of a set of alternative measures, e.g. mechanical, chemical or biological control strategies, and reveals the measure with the best net benefit-cost ratio for a particular target species. The most cost-efficient strategy is the one for which the reducing effect on the invasion is highest compared to its costs.

(ii) Comparison of the most cost-efficient strategies for **several** species, so as to select the invasive species management option with the highest benefit-cost ratio out of a set of several species. It allows a ranking of the management options along their cost-benefit ratio or net benefits.

Both objectives can be achieved either by an *ex-ante* or an *ex-post* evaluation. The latter intends to examine whether or not a measure proved to be efficient by measuring its benefit-cost ratio *ex-post.* To deliver information necessary for decision aid, an evaluation has to be conducted *ex-ante*. *Ex-post* evaluations can only confirm or disprove the efficiency of a measure.

4.2.5 Comparison of economic impact assessment and decision aid

Results of economic impact assessments and of decision aid can lead to contradictory outcomes. Decision advice derived from the results might differ greatly. The example of a choice regarding the management of three invasive species serves to illustrate this point (Table 6).

Table 6: Cost-benefit ratio and gross costs for the management of different invasive species (own source)

	Damage Costs	Management Costs	Gross costs	Net Benefits	Costs: Benefits
Species A	13	9	22	4	9:4
Species B	5	3	8	2	3:2
Species C	7	2	9	5	2:5

An economic impact assessment accounts for gross costs to indicate the highest need for action among all three species. It delivers a total sum of damage costs for all species plus the related management costs. Both cost types are the gross costs of the species´ invasion. The species with the highest damage costs plus management costs would be chosen to start management (Species A in Table 6).

In contrast to this, decision aid would choose the species with the highest net benefits due to reduced impacts (species C). Net benefits are damage costs to which the management costs of the species are subtracted. (Precondition for this result is the assumption that the management is able to reduce impacts of invasive species to zero). In this case (species C), damage costs are even lower than for species A. The cost benefit ratio for this species is the highest (2:5) in comparison to species A and B.

Next to the difference in decision aid results, a second difference between economic impact assessment and decision aid lies in their perspective on the object of assessment. Economic impact assessments mostly consider a number of invasive species and associated impacts and focus on a certain region. Decision aid, on the other hand, mainly considers one single case for which certain impacts due to one species have to be countered with the most cost-efficient management option. A set of options is compared to reveal the most cost-efficient measure, each option showing the ratio of expected costs and expected benefits due to reduced impacts. Decision aid also helps explore repercussions of each management measure on the reduction of impacts.

To follow this approach, environmental economics usually employs the method of cost-benefit analysis (CBA), which will be explained in section 4.3. Decision aid or costs benefit studies, however, are much more complex to undertake than economic impact assessment or COPI studies, because information about the beneficial effect of management options is often only vague. Effects of invasive species' management are associated with a high degree of uncertainty, which was explained in chapter 3.

4.3 Cost-benefit analysis as tool for decision aid

As explained above, welfare economics aims to maximise social welfare. Cost-benefit analysis (CBA) helps to maximise such social welfare in a concrete decision context, namely if alternative political actions are at stake. It is used as a tool for policy and project analysis throughout the world (Hanley 2001). Next to other methods, such as cost-effectiveness analysis (CEA), multi-criteria-analysis (MCA), or environmental impact assessment, CBA is a prominent example of methods that support decision-making (Hanley 2001). It has been appealing for public decision-making for many years (Hanley and Spash 1993; Perman *et al.* 1999; Hanley 2001; Bräuer 2003; Hansjürgens 2004; Common and Stagl 2005; Pearce *et al.* 2006). The reason for this is the provision of costs and benefits of a project or policy, which can be compared. Clear propositions pro or contra an action can be therefore formulated. In this sense, CBA is one answer to the demanded appropriateness of management options which has been demanded by the CBD regarding biological invasions. Of course there have been critical voices with good arguments against this method during the last decade[30]. However, first of all the general intention of CBA will be illustrated and it will be shown how it can support decision. The discussion about CBA will be outlined in section 4.6, but not going into too much detail.

4.3.1 Purpose of a cost-benefit analysis

CBA aims to examine different policies, projects or management alternatives in terms of respective societal costs (i.e. public losses of welfare) and benefits (i.e. public gains of welfare). Accordingly, CBA enables rational decision-making for an efficient allocation of resources by showing the alternative with the highest cost-benefit ratio and where net benefits are maximised (Pearce *et al.* 2006: 52). This means CBA can compare advantages and disadvantages of different projects, policies or even management alternatives and helps:

1. To decide the profitability of any public project,
2. To choose one project, management alternative or policy out of several.

While accounting for net costs and benefits, CBA regards economic costs and therefore aims to include external effects of the project at hand. An alternative is considered to be economically useful if benefits of the alternative exceed its costs and if the net present value is highest.

As CBA is based on welfare economics, individual preferences, performed by either WTP or WTA, are the source of value that show whether a project or policy is favourable or disadvantageous for society. WTP or WTA are examined by either looking at actual market prices or by constructing hypothetical markets. Individual preferences are then aggregated for the affected region where the project or policy will influence welfare. The first and most fundamental question is whether CBA regards policies or projects (investments) and whether

[30] See for example Funtowicz and Ravetz (1993), Munda (1996), Hansjürgens (1999), Funtowicz (2002), Hansjürgens 2004.

it starts from an *ex-ante* or *ex-post* point of view. *Ex-ante* as well as *ex-post* CBA corresponds to the outlined possibilities of decision aid in the context of biological invasions. In the same way as decision aid can tell *ex-ante* whether "something that has not been done should be done" or ex-post "finding out whether something that has been done should have been done", CBA has the same view on a project or policy (Pearce *et al.* 2006: 52).

4.3.2 Stages of practical cost-benefit analysis

Three general stages of a CBA must be conducted independent, however, from the above mentioned point of view, i.e. either *ex-ante* or *ex-post*. These stages are outlined in the following[31].

Stage one: Defining the options

Starting from a reasonably defined goal several options to reach the goal are determined. Usually different options are available that should be divided into feasible and non-feasible ones. Defining the options to be valued means to set the framework of the valuation context, spatially as well as politically. The spatial extent refers to reflecting on gainers and losers of the project or policy to be conducted, and which costs and benefits are to be aggregated (Hanley and Spash 1993: 8). This aspect of the framework also decides whether people in the vicinity of the project area are taken into account and how the area of interest has been extended.

At the end of step one, the whole project or policy to be valued has been determined and described.

Stage two: Identification and assessment of project impacts

Once the project and its options have been set, the next step is concerned with the impacts that relate to the implementation of the policy or project. Such impacts include resources necessary to implement the project (labour hours, material, machinery) and effects that are caused if the project would have been implemented (*ex-ante* perspective) or that have happened when the project has been implemented (*ex-post* perspective). Such effects are listed and measured in its physical character and usually relate to a baseline with a "business as usual" scenario.

[31] For more detailed information about CBA see for instance Hanley and Spash (1993) or Pearce *et al.* (2006), Hansjürgens (1999).

Effects can be regarded either from the perspective of *additionality*, that is to consider net impacts and a reduction of effects that might have occurred without the project, or *displacement,* which means effects of displacement either in the project area or within the authorities' area of responsibility that might be much larger than the project area.

Impacts of the project are considered in relation to the time horizon, i.e. how far into the future impacts are relevant for society. Although the time horizon has to be set, no hard rules exist[32].

This second step strongly correlates to what will be called physical assessment in the context of biological invasions in this study (see section 1.2). During that stage, the analysis structure of what will be economically valued is elaborated and offers a set of goods and their physical changes in the environment due to the project.

Stage three: Monetary valuation of impacts

Once impacts have been physically assessed they have to be valued. This commonly happens in monetary terms and makes physical impacts and resources (e.g. labour hours and machinery) comparable (Bräuer 2003). As stated above, values are expressed by individual preferences that are reflected via prices to show the relative scarcity of a good at hand. For changes in quality or quantity of the different goods affect by the project, different valuation techniques are available (see section 4.4). This stage of monetarisation in general intends:

1. To estimate prices for goods that might change in the future in its availability

2. To correct market prices, e.g. if governmental intervention distorts them or externalities occur

3. To create prices for goods that are not traded on markets, such as public goods.

Discounting is a crucial point in the valuation process with respect to monetary values of expected or occurring impacts. Discounting takes into account that there is a time preference of individuals. It implies some form of impatience concerning the use of a good (Perman *et al.* 1999: 92). This fact is expressed in the time preference rate, which is usually positive and depicts the value of a small increment of utility which falls as its day of receipts is delayed (ibid). Once the flows of costs and benefits are expressed they have to be adjusted with the chosen discount rate to gain the net present value, which is the value of the project and respective costs and benefits in the course of time at the exact point in time of the decision.

[32] For an overview of different approaches to set the time horizon for impacts, see Pearce *et al.* (2006: 57).

This conversion of all costs and benefits into the net present value is necessary due "to time value of money, or time preference" (Hanley and Spash 1993: 16). If present costs and benefits are more important for the decision maker than occurring in the future, he or she will use a higher discount rate than a person who does not prefer the good to be used right now. Discount rates are highly discussed, as the weight of costs and benefits relative to their occurrence in the future decisively influences the result of the economic valuation.

The third step also includes the consideration of net benefits, which is also called the net present value test (Hanley and Spash 1993: 17). The question at hand simply asks whether gains (benefits) exceed discounted losses (costs). If yes, there are net benefits. The comparison of costs and benefits reveals whether a project or policy represents an efficient shift in the allocation of resources when referring to the framework in stage one and two. Regarding this stage, one realises that decision aid calculates net benefits of certain management options in terms of its efficiency.

At the end of the process, CBA will indicate the "optimal" solution (i.e. the option with the highest cost-benefit ratio) and will provide this result to decision makers. Therefore, CBA is the corresponding method to the general approach of decision aid, introduced in section 4.2.

4.4 Environmental economic valuation techniques

As stated above, when impacts of a project are valued CBA can work either with existing market prices or has to create hypothetical markets in which goods are virtually traded, which is the case for many public goods. Environmental economics uses money to attribute a value to goods that create welfare. A way has to be found to assign monetary values to environmental phenomena that benefit or damage human well-being. Although this thesis centres on shortcomings concerning the first step of any economic valuation, the physical assessment of impacts, it is useful to know *how* current environmental economic evaluation methods approach the evaluation of environmental goods for which no market prices exist. This is especially necessary to understand the state of the art and to elicit the shortcomings of current economic assessment studies (see chapter 5). This section introduces evaluation techniques that allow for the valuation of the environment as a public good, but also to value changes in private goods availability.

The methods are explained without claiming to give an encompassing overview about the pro and cons of each method as this is not major topic of the thesis. The section rather aims to give a brief overview of different ways, possible for the evaluation of the environment within the environmental economic valuation frame[33].

Current valuation techniques are generally divided into two categories: indirect, also called revealed, and direct or also called stated preference evaluation approaches. What distinguishes both approaches is the role of the consumer of the environmental good, whether he or she directly assigns a value to the change in utility due to the consumption of the good or the changed utility is deduced from already existing market prices. Both approaches observe the consumers′ choice and therefore individual preferences.

4.4.1 Revealed preferences methods

Where no markets exist for the good to be valued, environmental economic valuation can either use conventional markets to reveal the value of a non-market good or implicit markets (Bojö *et al.* 1992: pp. 75). The first group embraces the opportunity cost approach and the defensive expenditure technique, the latter, often referred to as the classical revealed preference methods, comprises hedonic pricing and the travel cost method. Both methods observe the demand for environmental goods by "examining the purchase of related goods in the private market place" (Garrod and Willis 1999: 7). Existing prices are used to show the implicit price for the environmental good at hand.

Opportunity cost approach

Governmental regulations affect the production of market goods. These effects become evident in terms of costs and prices or the available quantity of the market goods. The costs of environmental regulation or the costs to preserve a certain environmental good are in other words lost benefits that could be achieved alternatively through market output. The consideration of a foregone value of market output and the gained benefit through environmental protection is regarded with the *opportunity cost approach*[34]. However, the opportunity cost approach only matches if the analysis is conducted on the business level where governmental regulations influence the output or production to maintain certain environmental goods. When taking social opportunity costs into account, only the opportunity cost approach can be used for the economic valuation, as within this technique appropriate data can be used.

[33] For further reading on environmental economic valuation techniques see for instance Hanley (1992; 1997; 2001), Garrod and Willis (1999).

[34] Also found under the topic of "Changes in Production" (Bojö *et al.* 1992: 72).

This approach has been the basis for governmental compensation for compulsory payments of land and property or compensation to farmers that expect fewer yields due to environmentally friendly cultivation schemes that refrain from intensive cultivation with a high amount of fertiliser[35].

Defensive expenditures

Defensive expenditures regard "those expenditures by households and the public sector that are made in order to "defend" households from environmental damage" (Bojö *et al.* 1992: 49). This approach includes both *preventive expenditures* and *replacement costs*. The first approach regards the value of the environment that is deduced from what people are willing or prepared to spend to prevent the degradation of the environment or to avoid negative effects that decline individual well-being (Garrod and Willis 1999: 7). Both approaches are closely linked. The amount of money and time that has to be spent if people move from noisy or polluted areas to places with greater distances, for instance from their working place, are considered with these approaches.

Replacement costs value an environmental good by the costs incurred in restoring the environment to its original state after it has been degraded in quality or quantity (Garrod and Willis 1999: 7).

Travel Cost Method

The Travel Cost Method and Hedonic Pricing are the most common approaches used in environmental economics and use implicit markets to evaluate environmental goods. Both assume links between the consumption of market goods and the consumption of non-market environmental goods (Bojö *et al.* 1992: 76).

The Travel Cost Method (TCM) uses expenditures of individuals for leisure time activities. It takes the amount of money that is spent in travelling to recreation sites, such as to National Parks, to reveal the monetary value people are willing to pay to visit recreation sites by using the private good of transport. The travel costs are used as a proxy for price (Hanley 1992). This technique observes the demand for the private good transport to reach the recreation site in terms of the number of visits that varies according to the price that has to be paid for the transport.

[35] This approach assumes clear property rights over the use of land, because only if a government claims privately owned land is it forced to compensate the owner.

Hedonic Pricing Method

The Hedonic Pricing Method refers to the assumption that every good supplies a bundle of attributes and characteristics that are fundamental for human well-being (Hanley 1992; Chee 2004). A house, for instance, provides shelter and security and due to its location it provides certain qualitative and quantitative aspects of access to public services, such as schools, cultural activities or public transport. The demand for houses is a derived demand for the attributes that are linked with housing, and the price people pay for such reflects structural characteristics, such as plot size, rooms etc., but also environmental characteristics, such quietness, access to public transport etc. (Garrod and Willis 1999: pp. 87). The Hedonic Pricing approach permits the price for houses to show the implicit value of people for the environmental characteristics at hand.

4.4.2 Stated preferences methods

Stated or expressed preference methods encompass valuation techniques that measure the demand for environmental goods "by examining individual's expressed preferences for these goods relative to their demand for other assets" (Garrod and Willis 1999: 7). They do not attempt to either find complementary goods, such as travel or housing, or substitute goods to illustrate the individual's value of an environmental asset or service. They help to elicit the value of a given good by directly asking consumers about their choice and, hence, to make the attributed monetary value explicit. Such methods create hypothetical situations to regard the demand of an environmental good.

Contingent Valuation Method

The Contingent Valuation Method (CVM) has gained growing attention in the last two decades, especially since it was used for the 1989 Exxon Valdez oil spill case, which occurred in a unique coastal area of Alaska. For that case, the typical procedure of the method was applied. First, the unique composition of bird and mammal species was described. Second, the number of deaths in each species was estimated, the percentage of each species killed and the time it would take each species to recover its original size. Third, a programme was introduced, which was successful elsewhere, to prevent such a disaster to occur again. Facing these three aspects, people were asked if they would be willing to pay a one-time $ 10 fee to the government to support the programme to prevent another accident, such as an oil spill, and, if the charge would be $ 10, whether they would vote for the programme or not.

The main field of application becomes clear. CVM questions people to value a good in its entirety but without considering its various attributes. The good has to be valued holistically and afterwards certain individual elements can be valued. Different formats exist for the asking of such questions. Dichotomous choices can contain open-ended questionnaires, iterative bidding or payment card formats (Garrod and Willis 1999: 9). One reason why CVM became a favourite method is its possibility to assess especially challenging non-use values which are values assigned by individuals even though they do not actually use the good or plan to do so (Fischer 2003: 31). Identified valuation techniques do not allow the evaluation of non-use values of public goods, which is important for the valuation e.g. of wilderness, the preservation of landscapes or of biodiversity. For such goods there are no related marked goods at all, especially when it comes to the point that people value the mere existence of the good. CVM has been acknowledges to be the "non-marked analogue to measuring the price [...] for a private good" (Carson 1998: 16).

Although CVM offers a possibility to value non-use values, a lot of criticism has been voiced against the method (Carson *et al.* 2001). It has been subject to mainly three types of problems: (i) biased responses due to the information provided to respondents, (ii) the aggregation technique, and (iii) the choice of welfare measure (Hanley 1992). For further details on issues of CVM see for instance (Sagoff 1998; Garrod and Willis 1999: pp. 127; Fischer 2003: pp. 35).

Choice experiments

While CVM evaluates a particular scenario or a particular aspect of a quality change in the environment, in contrast choice experiments, also called *conjoint analysis*, can be used to examine individual values of the attributes of a scenario and of the scenario as a whole. Preferences are identified via the different attributes of a scenario. This technique of choice experiments tries to identify the utility of attributes of environmental goods or services for individuals by examining the trade-offs they make between the different attributes when making choices. The assumption that preferences are based not only on single attributes and that they originate always from joint or several attributes (hence the synonym of conjoint analysis) is underlying. An alternative to choice experiments are *contingent ranking methods*. In the same way that individuals have to choose the most preferred alternative along its attributes, this method also focuses on the ranking of alternatives (for further reading, see Garrod and Willis 1999: pp. 211)

4.5 The Total Economic Value

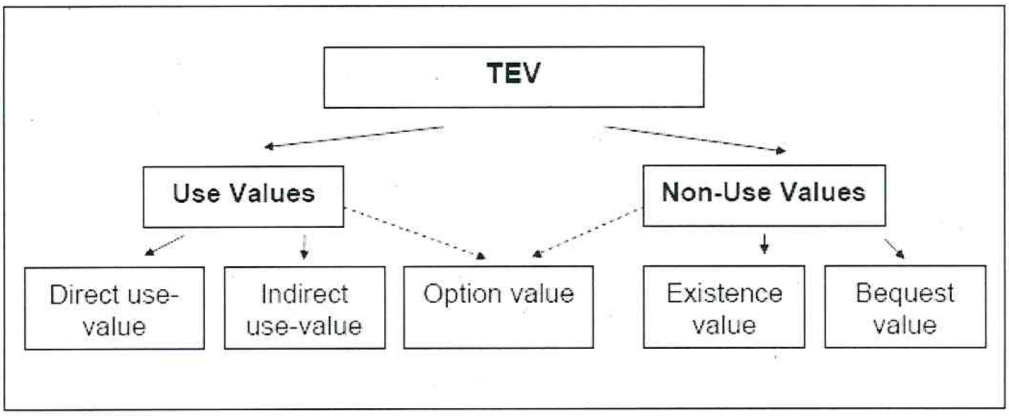
Due to the character of a public good impacts and the success of corresponding management options of biological invasions cannot be traded on markets. If markets are lacking, there are usually no prices that can reflect individual preferences for the mitigation or abatement of biological invasions. How the preferences of individuals for or against invasive species can be ascertained has been explained in the previous section. However, what types of environmental goods are affected and how they are relevant to humans has not been answered yet. The concept of the Total Economic Value (TEV) is one approach that helps to depict the wide range of environmental goods and how they are relevant to humans. Applied to biological invasions this concept can show how biological invasions affect humans from an economic point of view.

4.5.1 The idea of the Total Economic Value

It has been explained above that a variety of environmental goods contribute to human welfare for which no markets and thus no market prices exist. The concept of the Total Economic Value is one approach often used in environmental economics to illustrate that the environment provides goods and services that benefit humans[36]. The concept links environmental or ecosystem processes and functioning with the output of goods and services to which monetary values can be assigned in terms of WTP and WTA (Fromm 1997; WBGU 1999: 53; Turner *et al.* 2000: 73; Pearce *et al.* 2006: 45). The TEV is in line with the basic assumptions of welfare environmental economics: (i) humans always express values and ascribe them according to a goods utility and its value potential to change welfare. Thus the concept also argues from an anthropocentric view. Following this understanding only those goods are of value that somehow affect individual preferences (WBGU 1999: 49); (ii) each value is being quantified in terms of monetary values that correspond to the economic tradition to use money as a format of welfare. To each environmental asset a price can be assigned. How this can be done has been explained in section 4.4. Figure 15 depicts the structure of the concept of the TEV.

[36] For a comprising overview about the different values that can be assigned to environmental properties and the different approaches to categorise the values, see Fromm (2000).

Figure 15: The concept of the Total Economic Value (source: Pearce and Turner 1990: 131)



Several value categories are added up in the TEV. They all orientate on use-value of the environmental good at hand. In other words the TEV categories take into account how far a good is useful for humans (instrumental value). A main distinction is drawn between use and non-use values. Use-values derive from all the direct and indirect ways in which people make physical use of the environment. They can be divided into direct use-values and indirect use-values. Non-economists consider direct use-values often to be economic values, since for many of them market prices exist. These values comprise the direct utilisation of, for example biomass in terms of crop, timber or fish, all naturally produced assets. Also the utilisation of landscapes in terms of recreation is a direct use-value, for which, however, no market price exists. Indirect use-values are usually generated by ecological functions and processes. Humans benefit from these ecological functions indirectly, as for instance soil formation and soil retention are a prerequisite for the production of crop and the self-purification of freshwater is the preconditions for the provision of drinking water (Fromm 1997: 79). Option values are values attributed to maintaining the potential use of environmental assets or services of the ecosystem later (Pearce and Turner 1990: 131). This means option values contain some kind of uncertainty, since the consumer might not be able to say at the moment whether he or she will use the option in the future or not, or it has not been proven yet that the asset contains some utility for humans. Option values may be attributed to use-values and/or to non–use values (Pearce and Turner 1990: 132). A prominent example is the preservation of the rain forest to sustain the wide range of plants with pharmaceutical use that have not been discovered yet. Due to these reasons one tries keep the option value of the rain forest. Non-use values go beyond usage and often beyond a restrained economic assessment. They are sometimes discussed as intrinsic values, as consumers ascribe a positive value to a good for its pure existence without planning to use it at some point in time (Cansier 1996). Non-use values contain on the one hand existence values, and on the other hand bequest values. Existence values are "unrelated to any actual potential use of the good" (Pearce and Turner 1990: 134). How these values are discussed departs from the strict anthropocentric view, as this value category can be outside an instrumental, i.e. use-value attribution (Turner *et al.* 2003). A prominent example for an existence value is the expressed value of whales. Even if people do not expect to experience them at some point in the future, they are interested in sustaining a lively population of whales to guarantee their existence and ascribe a positive value to the knowledge that whales exist. A second form of non-use values are bequest values. They mirror the interest of people in maintaining an asset for future generations to inherit. Rationales for such a desire might be religiously motivated but are also the notion of responsibility for one's children and for future generations.

Although non-use values do not have a direct effect on humans use, the disappearance of non-use values poses a decrease in human well-being, according to the TEV. Economists assume altruistic or selfish motives of individuals to increase welfare by assigning existence values. Altruism gives utility to the giver, and the giver's utility depends on the utility of other people or beings (Pearce and Turner 1990: 135).

The different value categories logically overlap, and clear cut boundaries do not exist for the different value categories, especially regarding non use-values (Hampicke 1991: 118; Turner *et al.* 2003). The existence value is often subsumed under the bequest value or vice versa (Pearce and Turner 1990: 135).

With the TEV it can be clearly indicated that impacts on ecosystems, such as caused by biological invasions, cannot be divided into economic and ecological effects, which has very often been assumed by ecologists. How far the TEV embraces all different kinds of biological invasions´ impacts will be shown in the following.

4.5.2 The TEV applied to biological invasions

First of all, each invasive species has an impact on an ecosystem and its ecological functions and processes. Each of these impacts has also an economic aspect that can be explained with the TEV. Referring to this concept an impact of an invasive species can be also understood as damage as all environmental goods are of human interest due to their direct or indirect satisfaction of individual preferences. With respect to the TEV, all occurring impacts of biological invasions can now be attributed to TEV categories and are therefore relevant for economic assessments. With the help of the concept the different damage categories of invasive species can be depicted.

The ecosystem of the Cape region, the fynbos, serves as an example to illustrate how much biological invasions have an impact on the different value categories by changing their availability (Turpie *et al.* 2003). The fynbos ecosystem encompasses all value categories, all affected by invasive plant species. Direct use-values, such as building material and non-timber forest products, are affected by the invasive *Acacia*-species that compete with native flora. This type of change has often been referred to as economic damages (as the availability of direct use-values decreases). An alteration in species composition due to competition between native and invasive species in the fynbos vegetation resulted in changes of the direct use-value "recreation". Natural resource based tourism is one of the greatest income generators in the region, and changes in species composition influence the attractiveness of the landscape. Fewer visitors are the consequence of the increasing number of invasive species that are less attractive than the native ones. An example for an indirect use-value is the function of pollinators. They are essential for commercial fruit production and therefore for the provision of marketed fruits, as pollinators are necessary for the fruits and plants to grow. The option value lies in the rich biodiversity in the Cape region, which might contain future use options for the local people. An example for a non-use value, the bequest value, is the pure existence of the integer fynbos ecosystem and the possibility of bequeathing such an ecosystem to future generations.

The above mentioned examples of impacts of biological invasions illustrate (i) how different impacts of biological invasions are and that they can be attributed to all values categories. It (ii) also shows that what is often considered to be only of ecological relevance (i.e. if ecological processes are disturbed by invasive species) is also of relevance for humans, as such processes indirectly contribute to human well-being, and changes of such values imply a shift in welfare. The examples (iii) also show that there are interdependencies between the different properties, as illustrated with the pollinator-example. However, the TEV by itself does not provide this kind of information that is whether or not there are independencies between goods.

4.5.3 Limits of the TEV

The use of the concept of biological invasions that change and damage the provision of environmental goods has implications for economic assessments. Each affected value category has to be explained to elicit how far a good is of use for humans. The specific beneficial character of the good has to be explained with other attributes than the goods usefulness. The fynbos-example not only serves to describe the different TEV categories, it also depicts the high abstractness of the concept. Without further information about each value one would not have understood how the Cape region has been affected by the invasion. Additional information has to be provided to identify whether an indirect use-value stems, for instance, from the regulation of water or from the prevention of erosion. The high level of abstractness originates from the strong orientation on utility, which itself is an abstract term. The TEV itself does not provide information whether derived benefits are based on an environmental process or on an environmental good and whether there are dependencies between the goods to be valued. Each value has to be specified, and further attributes beyond its usability for humans are needed. How far the abstractness of the concept contains certain limits for ecological-economic assessments will be explained with the following paragraphs.

4.5.3.1 Lack of an analysis structure

The TEV adds up all value categories to a total sum of values (see Figure 16). This total sum, consisting of direct use-values (v_x), indirect use-values (v_y), option values (v_z), and non use-values (v_k), guarantees that all relevant aspects of environmental assets that influence human well-being are considered during the valuation process.

Figure 16: Equation of the TEV (own source)

$$TEV = \sum_x \sum_y \sum_z \sum_k (x^*v_x + y^*v_y + z^*v_z + k^*v_k)$$

v_x = direct use-values; v_y = indirect use-values; v_z= option use-values; v_k = non-use values

The major problem in the context of applying the TEV for economic assessments is the missing analysis structure for the different value categories. An analysis structure is understood as a framework to determine a set of goods or values that have to be taken into account in every assessment procedure. This analysis structure is the constituting part of every physical assessment of the goods to be valued (see section 1.2) and is the initial step before an economic valuation. Within this first step damages to human welfare in terms of changes of certain environmental goods are measured in physical, biological or chemical terms (Hampicke 1991: 78). Only the physical, biological or chemical description allows a closer characterisation of the value to humans. The fact, however, that the TEV does not provide an analysis structure as compulsory precondition of any economic valuation (see section 1.2) has often been neglected, if one surveys current literature, see chapter 5. By regarding the undefined number of *ex-ante* values that have to be assessed, it is clear that the TEV does not provide such a defined framework. Assuming a total sum of values means that 100% of all relevant values are considered. But how can one reveal which and how many values are relevant for an assessment and guarantee the completeness of such a set of values? Does a total sum include 7, 10 or 15 different values? And how is the ratio between the different value categories? The TEV neither defines the ratio of the value categories (i.e. the proportion between direct, indirect, option and non- use values) nor the exact number of x, y, z and k. This means there is no defined number of assets that enter the total sum, as the TEV is not concerned with the question of providing an analysis structure. Both, the ratio of the different value categories as well as the determination of x, y, z, or k depend on the focus of the valuation when using the TEV. As a result of the lacking analysis structure, relevant values, i.e. the number and the ratio of the different goods and their utility character to humans, will differ from assessment to assessment.

An example of an assessment of impacts of a road construction with the TEV can illustrate such difficulties. If the focus of the assessment is on impacts on biodiversity, indirect use-values, such as aesthetical properties of species, landscapes and ecosystems, as well as ecosystem processes that sustain diversity of species in the ecosystem. Also, bequest values might be affected, if people want to pass the ecosystem to future generations. This is now threatened in its originality by the road construction. Such an assessment would rather refrain from the consideration of many direct use-values, as, for instance, less agricultural yield on production sites that ought to be used as road construction areas might not be regarded.

The focus changes in the case of an assessment that considers forgone agricultural yield. This might become relevant if farmers demand compensation for land that use to be cultivated for agricultural production but is now used for road construction. In such as case, direct use-values will be focussed on, and option or indirect use-values might be of minor interest.

Both examples show that the total sum of values can be different. By using the TEV for economic assessments there is no fix set of value categories that have to be considered during the valuation. This is why the results of both cases will be very different.

4.5.3.2 Ecosystem values beyond use-values

Next to the missing analysis structure, the concept also lacks a value of the environment or of the ecosystem that goes beyond the notion of utility for humans and the provision of instrumental values. It considers the overarching value of an ecosystem with its self-organising capacity and its functions that stabilise the system as a whole and that guarantees its maintenance (Gren *et al.* 1994; Turner *et al.* 2000; Hampicke 2001: 156). This so called *primary value*, also referred to as infrastructure value or contributory value, acknowledges the ecosystem's self-organising capacity and follows the notion that an ecosystem brings all different components together (Gren *et al.* 1994). It also includes the dynamic changes of an ecosystem over time and its resilience, understood as an ecosystem's capacity to recover from disturbances (ibid).

Instrumental values (e.g. the production of timber or the provision of ornamentals) are also called secondary values, which are the values of life-support functions and ecological services and goods that the primary value (i.e. the ecosystem's self-organising capacity) generates. According to this understanding, the primary value is the precondition for the provision of goods and services. This can be underpinned by the fact that the provision of timber, crop or fish cannot be considered separately from corresponding ecological processes that enable and sustain their growth. Examples for ecological processes that pose indirect use-values and sustain the provision of such environmental goods are processes of soil renewal, nutrient uptake and nutrient availability (that allow crop or timber to grow in terrestrial ecosystems), and the process of water purification and its natural regulation (that allow fish to grow in aquatic ecosystems). This means common TEV-categories do not reveal interdependencies between certain environmental goods and services. The fynbos-examples shows that one can say that harvesting commercial fruits needs pollinators to grow and visitors come to the Cape region to make use of the recreation value of the landscape, which is affected by the decreasing attractiveness due to dominating invasive tree species.

4.5.3.3 Challenges for economic assessments due to limits of the TEV

The consideration of primary values poses one challenge for the economic valuation of ecosystems, or in the case of biological invasions, changes of the primary value in an ecosystem due to an invasive species. The role of this value is not clear when using the TEV.

Primary values have been neglected when using the TEV for an assessment of biological invasions impacts (see section 5.5). Turner *et al.* (2000) suggests to consider this value to be outside any economic valuation as the primary value is the very base for every good of use- and non-use value to humans. This seems reasonable because primary values cannot be subject of economic, i.e. of marginal consideration because the primary value cannot be substituted and the overall functioning of an ecosystem can only be considered as a whole. This has implications for economic assessments as a whole. It runs the risk to neglect such values generally as only economic values are taken into account within the valuation procedure. Although primary values are outside of economic valuation, an assessment still should record whether there are influences on or changes of such a primary value of an ecosystem by an invasive species. Such a record needs other formats than money.

Double counting during the valuation is another problem of the TEV. It will bias decision supporting results of an economic valuation. One environmental asset or service can have several values, as joint products are inherent in most of nature's processes (Turner *et al.* 2003). For instance, the production of timber contributes to climate regulation, nutrient cycling and the regulation of water. The TEV neither explains which processes are interlinked nor does it determine how many joint products occur for one assessed environmental good.

As shown above, the most crucial challenge for an application of the TEV, however, is the lacking analysis structure. Of course, one will argue that the TEV does not intend to measure impacts on utilised environmental goods in physical or biological terms because it belongs to the realm of economic valuation. An analysis structure, however, is part of a physical assessment. In reality the fact that this physical assessment is the compulsory precondition for the economic valuation has been neglected very often. The survey of current literature in chapter 5 depicts the problem. Although one crucial advantage of the TEV is its use of the comparable format money, it still does not guarantee that the results can be compared, as different assessment results stem from lacking information on the type of assessable goods and the undefined number of values. The TEV has a high educational value because it explains how far a certain environmental asset is of human use, and that non-use values have to be included in an economic valuation. But assessment studies that only use the TEV deliver results that are normally not comparable with each other if no clear analysis structure has been established before the valuation (see section 5.5). An ended comparability of study results is, however, is in the context of biological invasions impacts of major interest. An economic valuation of biological invasions aims to support decision-making by allowing the ranking of impacts of invasive species and the different management options to reduce them. Thus it is necessary to focus on the establishment of a generalised analysis structure. Only with such, results of the valuation will be comparable. A well structured assessment of the impacts to be evaluated seems even more important than the information about a goods utility to humans.

4.6 Critiques of the welfare economic valuation of biological invasions

Since this work intends to enhance economic assessments of biological invasions, it is also necessary to reflect on problems that are linked to the economic valuation of such. In current literature one can find criticism that is concerned with the fundament of welfare economics, other disapprovals are related to the empirical application of certain methods, such as CBA, and valuation techniques. In any case, critical voices are manifold. In this section only those criticisms will be mentioned that are relevant in the context of biological invasions. Thus, first more general aspects of a criticism are regarded that results from the fundament of welfare economic valuation and its assumptions. Second these critical aspects are referred to the context of biological invasions and their economic valuation; and third, the challenges which become especially relevant for a sound economic assessment of biological invasions which intends to deliver results useful for decision support, are outlined.

4.6.1 General criticism of welfare economic valuation of the environment

A series of critiques and problems of welfare economic valuation are related to its fundament and to its basic assumptions. Very basic critiques refer for instance to the assumed stability of preferences (see Chee 2004; Spash 1997), the character of preferences (Fischer 2003; Norton *et al.* 1998), appropriate techniques to elicit the value of environmental goods (Bojö *et al.* 1992: pp. 71; MacMillan *et al.* 1998) and the problem of aggregation (Hanley 1992). Certain fundamental problems become especially relevant in the context of an economic valuation of biodiversity for which biological invasions pose a serious threat (Tacconi and Bennett 1995; Oecd 1996: 47; Tacconi 2000; Christie *et al.* 2006). Thus problems when evaluating invasive species are assumed to be part of the economic valuation of biodiversity. In the context of biological invasions five major arguments regarding the neoclassical paradigm are taken up here, because they are relevant in the context of an economic valuation of such. In this subsection they are depicted from a general point of view, i.e. these arguments also apply for an economic valuation of the environment in general[37].

1. Mono-dimensionality of values

The mono-dimensionality of values (as any evaluation uses the format of money) has been criticised for not reflecting the real value of an environmental good. Such a *commodification of environmental goods* should be neglected, as there are intangible values, such as improved quality of life or the aesthetic and symbolic properties of the environment that are beyond economics and which are not commensurable (Funtowicz and Ravetz 1993). Such unmeasurable values can only be described in qualitative terms that are "noneconomic in nature" (Hussen 2000: 307)[38].

2. Substitutability of goods

The comparable unit of welfare is money and substitutability between money and a demanded good is assumed (see section 4.1). Economic valuation is always concerned with marginal changes (Cansier 1996: 78). Although marginal valuation for a variety of ecosystem goods is possible, e.g. the availability of harvestable timber, there are properties of environmental goods and their provisioning which restrict such a valuation. These properties refer to the ecological interdependency between ecosystem goods and functions. First, many of them cannot easily be substituted by another and marginal valuation and the paradigm of substitutability has been noted to reach its limits when evaluating such aspects of ecosystems or environmental goods and services (Fromm 1997: 97 pp; Fromm and Brüggemann 1999; WBGU 1999: 62; Fromm 2000). Second, the total value of a whole ecosystem does not equal with the sum of all constituents which can be marginally evaluated (Hampicke 2001: pp. 156). It is much greater than that and should be outside marginal consideration. This is due to the primary value which can be only considered in its completeness (see section 4.5.4.2).

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[37] For further reading on other general critiques on a neo-classical evaluation of the environment see for instance Bojö *et al.* (1992); Foster (1997); Hussen (2000: pp. 307), regarding pros and cons of CBA see Hanjürgens (2004); regarding CVM as valuation technique and its biases, see Garrod and Willis (1999: pp. 125), Hussen (2000: pp. 304); regarding problems due to the hypothetical character of the technique Faucheux and Noël (2001: pp. 335), Bojö *et al.* (1992: pp. 82); Navrud and Pruckner (1997).

[38] One approach to include such qualitative properties of the environment is to combine monetary valuation with deliberative processes, sometimes called deliberative monetary valuation (DMV) (Spash 2007). This rather participatory valuation emphasises on the quality of social processes in valuation. Within the last decade this approach has been of increasing interest (Sagoff 1998; Howarth and Farber 2002; Wilson and Howarth 2002).

3. Preferences and beliefs

Choices in economics are based on individual preferences according to the so called "methodological individualism" (Hansjürgens 2004: 242). Decision-making, however, is suspected to be guided by personal judgements based on believes and attitudes when evaluating the environment. There seems to be a crucial confusion between individual preferences and beliefs, e.g. Sagoff (1996); Sagoff (1998); Spash (1997).

4. Uncertainty

High uncertainty makes the measurement of environmental damages very difficult. The evaluation of damages on certain environmental goods seems even impossible, as some of them cannot be replaced and no close substitutes exist (Krutilla and Fischer 1985). Irreversibility, changing attitudes and the development of new stakes are inherent to environmental problems and their valuation is very context dependent (Funtowicz *et al.* 2002: 53). Natural systems have been recognized as dynamic and complex (Funtowicz and Ravetz 1993). There are unexpected events due to the non-linearity of ecological systems. Thresholds for one stable state of a system flipping to another stable state are usually not predictable (Batabyal *et al.* 2003).

4.6.2 Challenges for an economic valuation of biological invasions

The critiques mentioned above are now discussed in relation to biological invasions and why they are relevant in that context.

1. Mono-dimensionality

The Mono-dimensionality of values is problematic due to the variety of effects caused by biological invasions. Variety not only means that most effects are in the first hand ecological impacts on genes, species or ecosystems, see section 2.3.1, it also refers to the fact that the perception of impacts by humans is very subjective and heterogeneous. For one person a certain invasive species might be damaging, for another beneficial (Binimelis *et al.* 2007: pp. 337). Regarding the great number of ecological impacts, it is not only difficult to detect changes in species composition or in the genetic pool of a species, the conversion from a physical, chemical or biological measurement unit into the format of money is also challenging. A lot of information will be lost during the conversion process as the different formats, e.g. decrease in life quality or loss of certain species will be transferred into the unit of money.

Such a conversion usually happens by addressing questions about the willingness to pay for the impact at hand. However, it is doubtful that respondents do have any preferences concerning ecological impacts, such as changes in a food web or the increase in biomass production due to an invasive species. Such impacts are suspected to be at the utility function of individuals, the criterion of economic relevance when conducting a CBA (Hanley and Spash 1993: 10; Faucheux and Noël 2001: 343). Although such impacts are not of major concern for an economic valuation, they usually are the base for human relevant effects, such as crop or timber production (see section 2.3.2).

2. Substitutability

Substitutability among goods becomes especially problematic if one concerns the case of species extinction due to invasive species, i.e. a fact of irreversibility. Whether this fact can be compensated via money or other species is questionable, because the lost species usually has had a unique position within the ecosystem (WBGU 1999: 62). Substitutability among species would only exist if they would be identical in their genetic structure (ibid). However, the assumption of substitutability is not only problematic regarding irreversible facts of invasive species. When taking primary values into account, i.e. the glue value of ecosystems, one will recognise that dependencies among goods and services and their provision by ecosystems are also conflicting with the assumption that one such ecological good can be substituted by another. As shown above, for primary values there are no substitutes at all. To maintain the primary value of an ecosystem is too essential than so assume that there is any substitute (Gren *et al.* 1994).

3. Preferences and beliefs

There is a big discussion whether decisions about biological invasions are guided by individual and rational preferences or by believes and attitudes (Eser 1998; Shrader-Frechette 2001) The fact that societal views are of great importance becomes apparent when considering the questions that guide decision-making about the management of biological invasions. The question "what should we as a society do to encounter invasive species?" rather guides such decision than the welfare economic oriented question of "what is good for us, what increases our individual benefits?". Apparently the whole debate refrains from the reference to personal well-being in terms of satisfying individual interests. Collective views guide decision-making concerning the management of biological invasions. Normative value judgements play a vital role in every activism against invasive species. General societal attitudes are reflected by the simple fact that damaging invasive weeds are always "worse" than native weeds. Although a native weed, such as the European crouch (*Agropyron repens*) perhaps might have the same damaging effect on agricultural crop yield, such as an invasive weed, the public has a different attitude towards both weeds.

The management of biological invasions is guided by a strong feeling to need preservation of the *Heimat* of European landscapes (Eser 1998: pp. 115). *Heimat,* which is the state of the central European environment between the 18th and 19th century, has been essential for the idea of nature conservation in central Europe. It is still relevant today. Invasive species threatening this original state of landscapes and mans natural environment are object of conservation strategies. Eser (1998: pp. 18) states that the problem with invasive species rather arises from a species characteristic of being a stranger than acknowledging that they are recently emerging problem which possess the potential to alternate *Heimat* and the fact that they are human made.

4. Uncertainty

Uncertainty plays a vital role in the issue of invasive species (see chapter 3). Implications for an economic valuation when conducting a CBA are for instance setting the time-horizon for when damages of biological invasions occur. This is often not possible from an *ex-ante* point of view, as the complexity of impacts and unknown invasion mechanisms and lacking data do not allow for such a determination. As consequence of the previous argument an appropriate discount rate is quite subjective.

The high degree of uncertainty regarding the ecological accuracy of an action and its capacity to reduce impacts of the species often denies the possibility to set a clear cut goal for a management project. This is due to the unknown invasion mechanisms that restrict or promote a management strategy's success (see section 2.3). Although there are different management strategies suggested to be appropriate for the different phases of the invasion process (see section 2.4), probabilities for the success of such a management strategy are usually not available. This fact makes the great difference between objects of standard economic valuation and biological invasions. For typical objects of CBA, the success or the outcome of the policy or the project can be mostly clearly framed and at least anticipated.

4.6.3 Challenges for ecological-economic assessments of biological invasions

As mentioned in the introduction, this thesis intends to contribute to enhancing the methodology of economic assessments of biological invasions. From above, one can see that an economic valuation of biological invasions faces several problems which affect the results for decision-making. This thesis has argued that economic assessments are a two step procedure, which also contains the physical assessment of biological invasions´ impacts (see introduction). The quality of assessment results is determined on the one hand by "how" impacts are evaluated, i.e. by the economic valuation, but also by "what" has to be evaluated, i.e. by the impacts determined during the physical assessment. Such an analysis structure, however, has been scarcely or rather in an abstract way discussed yet. In contrast to what happens quite often, i.e. only the second stage (the economic valuation) has been focussed, the physical assessment is even more relevant in the context of biological invasions as it determines what will enter the economic valuation. The physical assessment determines the object of assessment which in the context of biological invasions are the different and manifold impacts due to an invasive species. During that stage it is already of importance to detect who will be affected by the invasion, what impacts are expected, at what point in time and how could the management strategies look like. As shown with chapter 3, this is often difficult due to uncertainty.

The TEV has been mentioned to provide information about "what" has to be assessed. In section 4.5.4.3 it has become clear that the TEV is very abstract and only the assignment of physical, chemical or biological attributes shows how a certain good can be utilised by humans. It has been claimed that such an abstract concept does not help to analyse the different impacts of biological invasions and to detect where we have to deal with uncertainty concerning these impacts. An analysis structure for the different impacts of biological invasions to be recorded is of crucial importance because the TEV does not intend to function as such. Additionally, biological invasions´ impacts often affect indirect or non use-values, and even values that are outside the scope TEV categories because they are too fundamental than being object of a marginal valuation. However, due to their essential functioning there is a need at least to consider them and whether there are impacts of biological invasions or not, without intending to valuate them economically.

Thus, taking into account all requests that have been mentioned above and which are relevant for an improved economic assessment of biological invasions, one can summarise that at the moment there is no proper approach for a physical assessment of biological invasions in terms of recording and describing all impacts relevant for human well-being. It is useful to turn to the base of any economic assessment in this work: to the physical assessment that produces results that enter the valuation step. Related to the questions above, predominantly two problems become obvious for a physical assessment which ought to be explored and have been often disregarded: 1. the high uncertainty, and 2. the scope of impacts.

1. Uncertainty

This problem will be referred to as the *uncertainty*-problem in the following. A proper physical assessment must include and reveal the high degree of uncertainty about impacts in the measurement procedure. This is necessary for delivering sound data that enters the valuation stage and which can be used for decision aid.

Regarding biological invasions an analysis structure should detect uncertainty concerning:

- The species: which species will be introduced? Are there several species or only one that can or will be introduced into a new range?
- The time: at what point in time a species will progress from one phase to the next?
- The location: where will the invasion take place and which ecosystem and ecosystem assets are affected?
- The impacts: what damaging impacts can be expected if an alien species has become a successful invader? How frequent and how profound will the damages be?

The first three questions concerning biological invasions are outside economic concern as they deal with ecological knowledge. Still, they are important for decision advice. The last two questions are of economic concern, namely if effects of biological invasions touch human concern. Hence, the *uncertainty*-problem addresses the last two questions about impacts of invasive species, and how available knowledge about impacts can be depicted.

2. The scope of impacts

This problem will be referred to as the *scope*-problem in the following. A proper physical assessment must consider all impacts and must cover the scope of impacts that can happen due to an invasive species, no matter if they touch only use-values, or values outside the scope of the TEV. This is necessary in order to guarantee that results are comparable and cover the scope of values that are touched by the invasion. This means it must be reflected on questions regarding:

- Which values are touched?
- Are also values touched that go beyond use- and non-use values?
- How can a completeness of all impacts due to biological invasions be guaranteed?

All previous chapters have presented several problems linked to biological invasions. In the following the focus if this work will be on the *uncertainty*-problem and the *scope*-problem. The concentration on these two major problems is necessary because they essentially influence the quality of any economic assessment of biological invasions.

First it will be examined how far recent economic assessment studies have considered both challenges and if such studies underline the demand to approach and conceptually enhance a physical assessment. The result of the review in chapter 5 will be used as a deficit analysis and will underpin the demand to reflect on both problems. Second, a new concept will be introduced that offers advantages for a conceptual enhancement of economic assessments in terms of providing an analysis structure for a physical assessment (see chapter 6).

5 Survey of current studies on the economics of biological invasions

5.1 Introductory remarks

This chapter intends to bring economic evaluation and assessment studies together and survey them under the central question: "To what extent are current economic evaluation studies suitable for policy advice?" In other words, do they provide appropriate information to aid actual policy processes? Are the numbers "hard enough" to be used as a decision aid?[39]

Within this chapter there is no clear distinction between economic assessment and economic valuation as distinguished in section 1.2. This is due to the fact that many studies do not make that distinction. The chapter serves to review the studies on the main challenges of economic assessments of biological invasions: the *scope*- and the *uncertainty*-problem. However, the chapter goes beyond the mere questions whether economic studies recognise both challenges. Instead it gives a broad overview of existing studies that worked on the topic of economics of invasions and shows what has been achieved so far in this field of research and what has been left open.

For the survey there are three central questions that guide the review process. They illustrate the suitability of the valuation results for policy advice. The three questions are called cornerstones in the following and frame the survey in terms of three major perspectives on the topic. The cornerstones are to elicit essential properties of an economic study to be suitable for decision support. The central questions are the following:

1. Do economic studies either deliver an economic impact assessment or do they consider the proportion of impact and management costs, i.e. how far are they suited as decision aid?
2. If economic studies consider strategies to encounter biological invasions, what type of measures (according to the CBD demand) do they consider?
3. Do they consider external effects which are reflected through the type of data that is used in the respective study?

Additionally (i) the *scope*-problem and the (ii) *uncertainty*-problem as a derivate of the

[39] The chapter is bases on a published article by Born *et al.* (2005).

previous chapters are addressed in terms of questioning (i) "how comprising are the assessment studies along the TEV?", and (ii) how far do such studies take into account that there is uncertainty on various levels?"

The following sections are organised as follows: First, the cornerstones of the review are introduced and explained (Section 5.2). In section 5.3 a general overview of current economic studies on biological invasions is given. Subsequently, in greater detail selected studies are analysed by using a set of criteria that disclose the quality of the present economic studies and their suitability for policy advice (section 5.4). For this analysis studies are selected which disclose quantitative economic results clearly related (more or less) to a spatial and temporal framework of analysis, and that provide a basis for general cost-benefit reflections. In section 5.5 the studies from the in-depth analysis are considered regarding the challenges of the *scope*- and the *uncertainty*-problem. Section 5.6 draws conclusion whether or not and to what extent the investigated studies are suitable for policy advice and meet the *scope*- and the *uncertainty*-problem.

5.2 Cornerstones of the review

Referring to the questions above for the survey central cornerstones are formulated. Concerning the first question two distinct types of economic assessment are highlighted under the cornerstone "**field of application**". For answering the second question the categorisation of the measures along the CBD (1. prevention, 2. eradication, 3. control, supplemented by 4. adaptation) is used and constitutes the cornerstone "**evaluated measure**". Question 3 is subsumed under the cornerstone "**private versus social costs and benefits**" which reflects the two distinct assessment perspectives that are applied in economic studies.

Additionally to answer the questions regarding the challenges of uncertainty and the scope of impacts, two more cornerstones are used which will be discussed in detail in section 5.5. The *scope*-problem is regarded under the cornerstone "**scope of impacts to be considered**", the *uncertainty*-problem considered under the cornerstone "**the role of uncertainty**". The cornerstones are explained in the following.

Cornerstone 1: "Field of application"

Referring to the content of each study the two distinct fields of policy support are considered. As explained in section 4.2 they can either raise general awareness through assessing overall damages or support decision-making concerning the choice of an appropriate strategy.

1. Economic impact assessment

Many studies consider costs of impacts of biological invasions from the point of no political action. The purpose is to raise awareness about the considerable amount of costs generated. Such economic impact assessments account for gross costs of invasive species impacts and support to decide whether any action should be considered at all, and if yes, in which field (Bräuer 2003). It starts from the point where new or revised political action is needed. Details have been explained in section 4.2.3

2. Decision aid

Economic assessment studies that serve as decision aid for policy advice contain account for net benefits (see section 4.2.4). It can have two foci, which can be considered *ex-ante* and *ex-post*:

1. Selection of the optimal measure concerning one species out of a set of alternative measures to identify the most cost-efficient strategy.
2. Comparison of different cost-effective measures for different target species, to reveal the highest net benefits of management strategies for several species.

Cornerstone 2: "evaluated strategy"

Selecting the appropriate conservation strategy depends on the phase of the invasion process as explained in section 2.4. In the review the studies were assigned to one of the four strategies. Additionally the comparison between different strategies where applicable to respond to the first focus of decision aid "selection of an optimal strategy out of a set of alternative strategies" has been marked.

Cornerstone 3: "social versus private costs and benefits"

As explained in section 4.1 biological invasions are characterised through external effects due to the public good nature of the multiple goods and services of ecosystems which are affected through invasion impacts. This should be reflected in the data that are used for economic assessments. Economics usually draws a line between two different types of costs and benefits that enter ecological-economic assessments: social and private costs and benefits. Social costs and benefits request "which strategy has more benefits than costs for society?". Private costs and benefits ask "which strategy is profitable for a private business?". The private assessment of costs and benefits means accounting for expenditures and earnings for the private sector, often for one single business, e.g. a farm. Direct income effects are involved, for example defensive costs arise through the spraying of pesticides and the release of biological control agents. Often it is appropriate to calculate both the costs and the benefits caused by the impacts of invasive species in terms of market prices. Only direct use-values are considered. However, the wide scope of impacts of biological invasions is not taken into account by private costs and benefits. Appropriate data to internalise external effects of biological invasions, however, are social costs, not private costs when accounting for invasive species damages with the aim to provide economic assessments that supports decision-making. Social costs and benefits reflect on all value categories of the TEV and avoid biases of the damage value which happens, if market prices are used. Market imperfections, unemployed resources, taxes, and subsidies can distort such prices.

The use of data is also relevant for the different techniques that are employed for the evaluation of changes in the environment or in the specific case of biological invasions impacts (see section 4.2). For certain techniques, only private costs and benefits are relevant and can be used, such as the effect on production approach. For other techniques, such as the opportunity costs approach, both types of data are applicable, and results can be produced that are relevant for the valuation of changes in the environment.

5.3 Overview of the literature

To obtain an initial overview, the available literature is surveyed with reference to the evaluated CBD strategy and the potential field of application. It contains information about the distribution of the underlying policy support aimed at (decision aid or economic impact assessment), the evaluation perspective (ex-ante or *ex-post),* and about the evaluated conservation strategy (prevention, eradication, control or adaptation).

Table 7 gives an overview of 32 current research studies of costs and benefits of invasive species[40]. It contains information about the distribution of the underlying policy support aimed at (decision aid or economic impact assessment), the evaluation perspective (*ex-ante* or *ex-post*), and about the evaluated conservation strategy (prevention, eradication, control or adaptation).

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[40] The broad overview also entails ecological economic and bio-economic studies and papers. Not all existing studies have been reviewed. The focus was on studies with a rather scientific background, i.e. scientific books and peer-reviewed articles. Due to double mentioning of some studies in several categories, the total number of studies in Table 3 is 36.

Table 7: Economic studies of biological invasions in terms of field of application (aim), content, and evaluated management strategy (according to Born *et al.* 2005)

Aim		Study	Content	Evaluated strategy*				
				Pre	Era	Con	Ada	Comp
Decision Aid	*Ex-ante*	Reinhardt *et al.* 2003 [1]	Economic impact assessment of 20 invasive species, Germany		x	x		
		Zavaleta 2000	C&B** of controlling Tamarix, western USA		x	x		
		Barbier 2001	Bioeconomic model of interspecific competition between invaders and indigenous species			x		
		Cullen and Whitten 1995[1]	C&B** of biocontrol, Australia			x		
		Higgins *et al.* 1996	Ecological economic model of fynbos ecosystem functions and management, South Africa			x		
		Higgins *et al.* 1997	Dynamic ecological economic model for conflict solution in invasive species management, South Africa			x		
		Sharov and Liebhold 1998	Bioeconomic model of C&B** of invasive species' spread rates, USA			x		
		Settle *et al.* 2002	Dynamic model of interaction between exotic invaders (lake trout), native species (cut trout) and human use, USA			x		
		Wit *et al.* 2001	Economic scenarios of black wattle management, South Africa			x		
		Finnoff *et al.* 2005	Bioeconomic model to account for feedback links between zebra mussel and economic systems			x	x	x
		Studies ∑ 10	**Records ∑**	**0**	**2**	**10**	**1**	**1**
	Ex-post	Bertram 1999 [1]	Blueprint for action, impact assessment of pests, New Zealand	x	x	x		
		Anaman 1994	Input-output analysis of secondary impacts of screwworm fly, Australia		x			
		Reinhardt *et al.* 2003[1]	Economic impact assessment of 20 invasive species, Germany		x	x		
		Horan *et al.* 2003	Economic model on risk permits for invasive species management			x		
		Cullen and Whitten 1995[1]	C&B** of biocontrol, Australia			x		
		Le Maitre *et al.* 2002	Invasive alien trees and water resource management, C&B, South Africa			x		
		Odom *et al.* 2003	C&B of control strategies of Scotch broom, Australia			x		
		Tisdell 1990	C&B** of invasive weeds management, Australia			x		
		Buhle *et al.* 2005	Bioeconomic model to determine least costly control option			x		
		Knowler and Barbier 2005	Bioeconomic model of C&B** of introduced and traded Salt Cedar, USA			x		
		Wilgen 2001[1]	Review of economic evaluation studies of invasive species, South Africa			x		
		White and Newton-Cross 2000	Ecological and economic effects of rabbit calicivirus disease, Australia			x		
		Pimentel *et al.* 2002[1]	Economic evaluation of invasive species, Australia, Brazil, British Isles, India, New Zealand, South Africa, USA			x		
		Sharov and Liebhold 1998[1]	Bioeconomic model of C&B** of invasive species' spread rates, USA			x		
		McConnachie *et al.* 2003	Economic evaluation of biocontrol of red water fern, South Africa			x		x
		Hill and Greathead 2000 [1]	Economic evaluation of biocontrol strategies,			x		x
		Headrick and Goeden 2001	C&B** of biocontrol, case studies, USA			x		x

Aim	Study	Content	Evaluated strategy*				
			Pre	Era	Con	Ada	Comp
	Perrings 2005	Model about the optimal strategy to encounter invasive species under uncertainty			x	x	
	Studies ∑ 14	Records ∑	1	3	17	1	3
Economic Impact Assessment	Turpie and Heydenrych 2000	C&B** of invasive species' impacts on fynbos ecosystem, South Africa	-	-	-	-	-
	Engeman Richard M. *et al.* 2004	Economic evaluation of feral swine damage on marsh systems, USA	-	-	-	-	-
	Kasulo 2000	Impacts of invasive species on African lakes	-	-	-	-	-
	McNeely 2001	General impact record of biological invasions	-	-	-	-	-
	Pimentel 2001[1]	Economic evaluation of invasive species, Australia, Brazil, British Isles, India, New Zealand, South Africa, USA	-	-	-	-	-
	Pimentel 2005	Costs of 316 invasive species in US aquatic systems	-	-	-	-	-
	Pimentel *et al.* 2005	Update of global economic evaluation of invasive species (cp. Pimentel 2001)	-	-	-	-	-
	Reinhardt *et al.* 2003[1]	Economic impact assessment of 20 invasive species, Germany	-	-	-	-	-
	Studies ∑ 8	Records ∑	-[2]	-[2]	-[2]	-[2]	-[2]
	Total Studies ∑ 36	Records ∑	1	5	27	2	3

[1] Multi-species study in which several studies/methods are integrated.
[2] Per definition are no strategys evaluated.
(*Pre = prevention; Era = eradication; Con = control; Ada = Adaptation; Comp = comparison; **C&B = costs and benefits)

5.3.1 General character of the studies

In general it has been difficult to draw a firm boundary between studies aiming at decision aid or at an economic impact assessment, since the field of application the considered papers initially aimed at, has been rarely documented.

The review shows that the majority of the investigated studies follow the format of decision aid in the choice of a strategy. *Ex-post* studies evaluate all four management strategies or compare them, whereas *ex-ante* studies concentrate on control strategies.

5.3.1.1 Decision aid studies

The objective of decision aid studies is to ascertain the most efficient mitigation strategy by comparing the efficiency of different options and accounting for net benefits. Decision aid studies are divided into a group that accounts from an *ex-ante* view, a second group from an *ex-post* perspective.

10 studies evaluate possible strategys, i.e. *ex-ante*. They largely employ ecological-economic models to provide different management regimes (Higgins *et al.* 1997; Higgins *et al.* 1996; Settle *et al.* 2002; Sharov and Liebhold, 1998). Economic-ecological models offer the advantage of economic evaluation without a restriction to the status quo. Expected

developments can be outlined and included in the calculations (Barbier, 2001), which is useful for the choice of a proper conservation strategy. Settle *et al.* (2002) and Finnoff *et al.* (2005) model feedbacks from the economic and ecological system. The interaction of an invasive and a native species is shaped, taking into account human intervention in terms of its use or its management. A scenario approach is the one chosen by Wit *et al.* (2001) comparing two strategies: "do nothing" and "mitigation". Reinhardt *et al.* (2003) as well as Cullen and Whitten (1995) evaluate different mitigation strategies. However, mitigation mostly refers to control strategies in all studies. A classical cost-benefit analysis is conducted by Zavaleta (2000) who considers impacts of *Tamarix* in the western US and respective strategies of eradication and subsequent restoration. She compares the status quo of foregone benefits of local water use due to *Tamarix*, with the expected business costs for eradication and a subsequent restoration programme. This ratio shows that long term considerations of costs and benefits of an invasion often reveal the usefulness of management efforts.

There are 18 studies with an *ex-post* view which constitute the vast majority of decision aid studies (see Table 7). Three of them compare different strategies. A comparison of strategies for an individual species is conducted by McConnachie *et al.* (2003), who compare a set of alternative control strategies (mechanical, chemical and biological control). Hill and Greathead (2000) review several classical biological control programmes, and Headrick and Goeden (2001: pp. 249) provide two case studies of biological control and its general role to "have the best chance for success in ecosystem management".

Economic as well as bio-economic models present a considerable share of decision aid studies. Some deal with the application of market incentives, such as risk permits to prevent invasive species introduction or the use of market instruments for invasive species management (Horan and Lupi 2005), others model the choice of the optimal strategy in terms of economic costs and benefits of each strategy (Buhle *et al.* 2005; Knowler and Barbier 2005; Perrings 2005). Other studies quantify total costs due to invasive species *ex-post*, however distinguishing total costs for invasive species management and due to its impacts. The quantification of border control costs *ex-post* is possible by accounting for certain cost types, such as labour and material costs. For example, Bertram (1999: 69) investigates New Zealand's measures of "pest surveillance & response, vector control, pest control, conservation, and other biosecurity activities" plus total impact costs in terms of production loss from 1991 until 1999. Cullen and Whitten (1995) report on different studies which have done *ex-post* as well as *ex-ante* evaluations of biological control strategies reviewing efficiency of biological control strategies. Aim of the report is to explore the benefits (potential as well as real) of research in the field of biological control.

To sum up: So far economic decision aid studies concentrate on the evaluation of strategies already undertaken, as reflected in the high percentage of *ex-post* analyses, rather than to assess new mitigation strategies in form of *ex-ante* analysis.

5.3.1.2 Economic impact assessment studies

There are eight studies dealing with economic impact assessment. These studies only record gross costs of specific invasive species (see section 4.2.3). While doing so, they assume a business as usual where no political action has yet been implemented which corresponds to economic impact assessment without *any* policy (see for example Turpie *et al.* 2003, Engeman *et al.* 2004). Such studies consider only the costs of biological invasions impacts without strategies taken against them (see Figure 13). There are also economic impact assessments of the second type, where the baseline includes *some* political action, but a *revised* policy is necessary, because the former policy did not result in a total impact reduction to zero and a revision of management strategies has been called for (see Figure 14). This second type of studies accounts for costs of biological invasions estimated through impacts (e.g. agricultural production loss (Pimentel *et al.* 2001; Wilgen 2001) plus costs of applied mitigation strategies. Costs of *Ambrosia artemisiifolia* in terms of treatment expenditures of the induced allergic reactions by Reinhardt *et al.* (2003: pp. 18) illustrate one example for the second type of economic impact assessment. Another example is the study by Pimentel *et al.* (2001) that include control costs to the gross costs of invasive species impacts. They quantify the extra-costs for control of alien weed species as the proportion of alien to non-alien weeds multiplied with the financial costs for weed control.

What is typical for economic impact assessment studies is that they take into account impacts as well as responding strategies in a static view. They account for the total sum of damages plus management costs for the moment, but neglect the feed back loops between effects of strategies and targeted impacts. Reducing effects on impacts of applied strategies that will occur over a certain time period and that are usually considered as net benefits are refrained. Therefore such studies do not concern with efficiency, i.e. with benefits of management options that are necessary for policy advice as explained in section 4.2.3.

5.3.2 Evaluated strategies

With regard to strategies of biological invasions, the surveyed economic analyses focus mainly on control strategies, and here primarily on biological control[41]. This results from intensive agricultural and sylvicultural efforts in biological control. Due to the "sustainable and self-renewing nature" (Hill and Greathead 2000: 222) of such strategies, they use to turn out to be cost-effective and have been employed since 1888. Once the final invasion phase is reached, control strategies are often more reasonable than eradication, which can be explained by the ongoing irreversibility during the invasion process. As a result (i) eradicating a population is more expensive than mitigating and keeping the invasive population at an acceptable level (Sharov and Liebhold, 1998), and (ii) ecological studies indicate a high probability of eradication failure because of missed steps to reduce post-eradication susceptibility to re-invasion (Bertolino and Genovesi 2003; Zavaleta *et al.* 2001).

This explains why economic literature on eradication strategies is limited to three studies. Anaman (1994), Bertram (1999), and Rheinhardt *et al.* (2003) make *ex-post* analyses of successful eradication measures. For instance, Anaman (1994) calculates the costs of a successful programme combating the screwworm fly in Australia with a dynamic bio-economic model.

Even though prevention is the approach preferred by the CBD, it is the least economically investigated by one study (Bertram 1999). Only prevention activities of border control and quarantine in New Zealand are assessed *ex-post* and have been part of the national prevention program for many years (ibid).

5.4 *In-depth analysis of selected studies*

The quality and hence the credibility of the studies' results is a crucial point for their suitability as policy advice. In the following analysis, the focus is on studies that quantify the costs and benefits of biological invasions in detail and that provide sufficient information. Providing sufficient information means to give information about a set of seven criteria, which have been developed in order to elicit the character of each study and to show how far

[41]Biological control is defined as "the purposeful introduction and permanent establishment of exotic natural enemies of pests and weeds, with a view to permanently suppressing their abundance within a prescribed region or country" (Hill and Greathead 2000: 208). Confusingly, several studies subsume control and eradication efforts under "management". In this survey it is differentiated between these two management options as far as possible, in line with the original papers.

each is suitable for policy advice. The seven criteria are: (i) geographical focus, (ii) affected sectors and relevant stakeholders, (iii) evaluated conservation strategies and the level of assessment, (iv) methods, (v) type of data, (vi) total costs and (vii) uncertainty. Each criterion is explained below. Although uncertainty will be explored in detail, it seems useful to already depict in the analysis how a study deals with it. Out of the above described studies a set of twelve meet these demands for a detailed survey. The full analysis is shown in Table 8.

5.4.2.1 Geographical focus

This criterion provides details about the occurrence of the biological invasion (continents, countries, and ecosystems), site-specific conditions, and the spatial distribution of studies on invasive species throughout the world.

Concerning the distribution of studies, most research has been undertaken outside Europe. Economic studies are mainly conducted in South Africa, America, Australia and New Zealand. Studies from Asia and South America are not included as relevant analyses are not available.[42]

5.4.2.2 Affected sectors

If economic analysis is to support decision-making, it is important to take into account the given economic, social and political structures. As far as policy advice is concerned, it is therefore necessary to identify affected sectors and relevant actors that are exposed to damaging impacts of biological invasions[43]. The following sectors with relevant actors are distinguished: agriculture, fishery, forestry, health care, nature conservation, municipality, tourism.

A lot of studies focus on the analysis of impacts on the agricultural sector (11 times). Other land use sectors, such as forestry (6 times) and fishery (once) are also represented. Furthermore health care (4), tourism (3 times), and municipalities (7) are identified to be affected. Surprisingly, impacts on nature conservation have only been considered twice.

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[42] Owing to the lack of data on social costs and benefits on these countries, it cannot be concluded that biological invasions do not pose a problem on these countries (Marambe *et al.* 2001; Vitousek *et al.* 1996).

[43] The systematic of sectors subsumes policy fields, such as health or nature conservation, economic sectors, such as fishery, agriculture and forestry, but also the institutional level of municipalities.

5.4.2.3 Object and level of assessment

As object of the studies the CBD strategies are used again (see section 2.4): prevention (**Pre**), eradication (**Era**), control (**Con**). Adaptation is neglected in that analysis as none of the studies referred to that strategy.

The level of assessment describes the spatial reference of analysis and allows drawing conclusions on the appropriateness of data. A distinction is drawn between the regional and national level. For both levels data on social costs and benefits are necessary. As it will become clear data on private costs and benefits has also been used.

Table 8 shows that control strategies prevail again (75% of surveyed studies). Prevention strategies are assessed once, namely by Bertram (1999); a detailed analysis of eradication strategies is not available. Total costs, such as described as problematic for decision aid due to a lacking confrontation of costs and benefits are found in one third of all studies examined. This means, although such studies do refer to costs of management options, they often do not provide a costs-benefit ratio.

5.4.2.4 Methods

Monetary positive (benefits) and negative (costs) aspects of impacts and management options can be assessed with different methods, and can be based on data of private as well as social costs and benefits. Using the methods described in section 4.2, the following methods have been identified in the survey[44]:

1. *Opportunity costs* (**opp.c**) to reflect the costs for society when costs for the regulation of biological invasions could alternatively have produced market goods. Production loss (direct use-value) is measured via forgone benefits in the agricultural or forestry sector. The *effect on production approach* (**prod.c**) has been also used to illustrate effects on the business level. Production costs can also simply add up the direct (e.g. cost of labour, pesticides, machinery) and indirect (e.g. medical treatment) costs of measures. However, the distinction is often not very clear.

2. *Defensive expenditures* (**def.ex**) "measure the effect of an environmental externality on production possibilities, often by measuring the expenditures which individuals are willing to undertake to avert damage" (Bertram, 1999: 47).

[44] Hedonic pricing, as well as the travel cost method have not been found in any economic study and, hence, are not used in this survey.

3. The *contingent valuation method* (**CVM**) is the only stated preference method employed to assess relevant values of nature conservation aspects.

4. Some studies use information derived from *literature* (**lit**).

The mentioned valuation techniques are all different because they use different approaches (see section 4.4). These different approaches restrict the comparability of results and hence, the choice of the used technique like production or opportunity costs has direct effects on the studies results and their suitability for decision-making. The following aspects point them out exemplarily:

(i) Many control strategies tend to fail or are ineffective (Curnutt, 2000). Therefore, measuring the costs and benefits of such strategies is not the right approach for assessing the costs and benefits of biological invasions. As we know, management costs are often subsumed under general costs due to invasive species (see 4.2.4). On the one hand it implies an overestimation of impacts if there are more effective alternatives, and on the other it may lead to underestimation due to the imperfect assessment of external effects.

(ii) Normally, the higher production costs caused by invasive species will not vanish completely after appropriate mitigation strategies. For example, Pimentel *et al.* (2001) assume that 73% of US weeds are invasive species and that therefore 73% of weed costs are caused by invasive species. Few invasive species may cause major costs due to their ubiquitous predominance in ecosystems. However, their niche will be occupied by other species after their eradication or control. Therefore, costs would probably arise at any rate because native weeds cause the same effects. Distinguishing between different species with their respective share of the total amount of costs seems necessary in terms of the effectiveness of different strategies. Additionally, it might be more cost-effective to just reduce the population size and keep it at a low level instead of enforcing the eradication of the species (Sharov and Liebhold, 1998).

5.4.2.5 Type of data

The analysis is supposed to clearly indicate (i) whether the data used is unchanged data on private costs and benefits (see section 5.2) or data that has been adapted or obtained using the method of cost-benefit analysis, and (ii) the level of assessment. In the majority of cases it will not be appropriate to scale up regional data to a national level (as is done for instance by McConnachie *et al.* (2003)).

The following difficulties are identified in respect to the source of data and its importance regarding the political relevance of the results:

(i) One general problem is the data itself due to lacking or reduced data on social costs and on external effects. It is still difficult to identify (much less monetarise) external effects. One result is an extreme reduction of data. For example, Pimentel *et al.* (2001) reduce so-called "ecological costs" to ecological damage of (1) purple loosestrife and (2) aquatic weed control. There are certainly more control costs for other species. Furthermore, ecological costs cannot be reduced to control costs. In this context, the external effects of ecological impacts of invasive species are totally neglected.

(ii) Subsidies distort market prices. Mitigation strategies in the European agricultural sector will apparently turn out to be more expensive if subsidies under the Common Agriculture Policy (CAP) are subtracted (which has been neglected e.g. by Reinhardt *et al.* (2003). Providing policy advice has to take such distortions into account. Subtracting subsidies, however, would also decrease the opportunity costs, for example of cultivating *Quercus rubus,* an invasive species in German forests. One effect would be the encouragement of respective eradication programmes. Vague explanations on the source of data and their generation create suspicion of double-counting: whether damage costs arise with or without a certain control strategy (McConnachie *et al.* 2003) or whether governmental funding is subtracted from total expenditure (Bertram 1999).

(iii) It is questionable whether aggregated data on private costs and benefits are representative. Very often data collected for a certain area are extrapolated, which disregards different site conditions. Invasive species impacts may differ, as may the success of a certain strategy. It might be helpful to consider the range of costs of minimal and maximal scenarios.

(iv) Data on private costs and benefits neglect external costs. In such analyses "there has been substantial (economic) considerations (on measures), but no quantitative costing of a whole set of 'externalities', e.g. reduced hazards to users and the public from reduced chemical usage, lack of undesirable residues and decreased disruption of the environment and effects on non-target species", as Cullen and Whitten (1995: 272) remark in an economic valuation study on biological control.

When comparing type of data and level of assessment, it becomes obvious that there is often a discrepancy between the aim of the studies and the data used. Many studies aim to provide nationally relevant information but use unaltered data on private costs and benefits, implying shortcomings on the methodological side.

In general one will notice that few data on social costs are used. This is mainly due to the lack of figures available on economic costs and benefits caused by invasive species. However, respective studies also do not even mention the monetary value of changes of indirect and non-use values. Turpie and Heydenrych (2000) present the only study conducting a willingness-to-pay analysis regarding the loss or alteration of the option and existence values due to biological invasions. No other direct or indirect assessment approaches are found.

5.4.2.6 Total costs

The column "total costs" in Table 8 represents the total costs determined in each study. In addition, the benefit-cost ratio (**B/C**) is noted in brackets as stated in the original papers. (For an overview of all abbreviations, see below the table).

Although in some cases we cannot reconstruct the total costs, we at least illustrate them. It turns out that costs rise as the extent of a study increases. Two (or more) studies and their figures cannot be compared even if information is available about many of the criteria surveyed within the studies. For example, stating that the impact of biological invasions on New Zealand is exactly twelve times higher than in South Africa would be not sound. The studies by Bertram (1999) and Turpie and Heydenrych (2000) are too different in their figures to be compared owing to the different influential spheres of the respective invasive species, the different assessment methods used, and the different foci of the studies.

Table 8: Detailed analysis of quantitative studies on the costs and benefits of biological invasions (source: Born *et al.* 2005)

Source	Geographical focus /*IS*	Economic sector	Object/ Level of assessment	Method	Type of data			Total costs (B/C)
					Analyses of private costs and benefits (C&B)		Analyses of social C&B	
					i) Direct C (B)	ii) Indirect C (B)	iii) C (B)	
Bertram, 1999	New Zealand *weeds*	Agriculture Fishery Forestry Health care Nature cons.	Con, Pre Era/ National	i)-iii) prod.c, opp.c, def.ex. CVM	C of production loss (NZ$400 million)		C for defensive expenditure NZ$ 440 million)	**NZ$ 840 million/year (=1%of GDP)**
Turpie and Heydenrych, 2000	South Africa (Fynbos)	Agriculture Tourism Municipal	ImpA/ Regional	i) prod.c, opp.costs ii) opp.c iii) opp.c, CVM, lit.	C for harvest loss of: wild flowers • thatch • sour figs • tea • medicine • honey (= US$1–25/ha)	C of less: • water supply • (US$7-163/ha) • less pollination by fynbos bees • (US$8.3–114.6/ha)	C for loss of: • genetic resources • (US$ 80-700 million) • existence value • (1,35 US$/ha).	**US$65 million**
Wit *et al.* 2001	South Africa *Black wattle tree* (Fynbos)	Agriculture Forestry Municipal	Con (Biol. Control)/ National	i)-iii): opp.c, prod.c, lit	B of IS harvest (US$528 million)	C of increased fire hazards (US$1 million)	B of IS carbon sequestration (US$24) C of reduced surface stream flow (US$1425)	**US$ 552 million (2.6:1)**
Zavaleta, 2000	Western US *Tamarix spec.*	Agriculture Municipal	Era, Con/ Regional	iii): opp.c., proc.c			B due to reduced water loss (US$ mill) : • municipal (26-68) • irrigation (39-121) • Hydropower (16-44) • Flood control (52) Restoration + eradication cost (US$): 7420/ha	**133-285 US$ Benefits; 7429 US$/ha management costs**
Engeman *et al.* 2004	Florida, USA Feral swine (*marsh land)*	Nature cons.	ImpA Con/ Regional	iii) lit			Mitigation costs: 247,742 – 807,226/ha	**US$ 238,710 – 4,036,130**

Source	**Geographical focus /*IS***	**Economic sector**	**Object/ Level of assessment**	**Method**	**Type of data**			**Total costs (B/C)**
					Analyses of private costs and benefits (C&B)		**Analyses of social C&B**	
					i) Direct C (B)	**ii) Indirect C (B)**	**iii) C (B)**	
McConnachie *et al.* 2003	South Africa *Azolla filiculoides*	Agriculture Tourism Municipal	Con National	i) prod.c	1. Control C: Herbicides, Labour (US$1308) 2. Biolog. control C: Salaries infrastructure, survey (US$1511) 3. Damage C: Pumps, miscellaneous, livestock, alternative water facilities (US$7940)			**US$1511/ha (2.5:1)**
Wilgen, 2001	South Africa (Fynbos)	Agriculture Forestry Tourism	Con (Biol. Control) National	Review with CBA approach			US$20 million labour costs counted as benefits	**Over US$ 11.75 billion**
Reinhardt *et al.* 2003	Germany 20 IS	Nature cons. Agriculture Forestry Health Municipal	ImpA, Era, Con,; (Biol. Control)/ National	i) – ii) prod.c, opp.c	C of: • measures against IS (labour, material) • economic damages (yield loss, infrastructure damage)	C of: • increased demand of sustaining infrastructure, medical treatment		**€160 million/year**
Cullen and Whitten, 1995	Australia *Rubus fructicosus, Echium plantaginetum*	Agriculture	Con (Biol. Control)/ National	i) prod.c, opp.c	C of: • production loss • control costs			**AS$ 0,7-2 million (20-42:1)**
Pimentel *et al.* 2002	USA 50,000 IS (incl. beneficial species)	Agriculture Forestry Health care Municipal	ImpA/ National	i)-iii) lit	C of: • production loss • control	C of: • environmental damage • indirect damage		**US$137 billion/year**

Source	Geographical focus /*IS*	Economic sector	Object/ Level of assessment	Method	Type of data: Analyses of private costs and benefits (C&B) i) Direct C (B)	ii) Indirect C (B)	Analyses of social C&B iii) C (B)	Total costs (B/C)
Tisdell, 1990	Australia Project 1: *4 weed species* Project 2: *Echium*-species	Agriculture	Con (Biol. Control)/ National	i) prod.c, opp.c	C of: • reduced crop pollination • implementation costs • income loss by apiarists B of: • reduction of chem. control • increased wheat production (income) • increased livestock grazing			**Project 1: A$33 million (1.5:1)** **Project 2: A$17 million (8.7:1)**
Pimentel *et al.* 2001	World (UK, India, USA, South Africa, Brazil) 120,000 IS (incl. beneficial species)	Agriculture Forestry Health care Municipal	ImpA; Comp/ Global	i)-iii) lit	Production loss in: • crop: US$216.1 billion/year • pasture: US$7.5 billon/year • forest: US$4.2 billion/year Health: US$6984.7 million/year	C of: • environmental damage		**US$336 billion/year**

Used abbreviations

Geographical focus	Object of assessment		Method	Total costs
Country, region (region) Harmful invasive species (*IS*) (Affected landscape)	Prevention (Pre) Eradication (Era) Control (Con)	Decision Aid	Opportunity costs (opp.c) Production costs (prod.c) Defensive expenditure (def.ex)	Benefit-Cost ratio (B/C)
	Impact (ImpA)	Economic Impact Assessment	Contingent Valuation (CVM) Literature analysis (lit) (i), (ii), (iii) refers to "Type of data"	

Ad 1: Remarks on non-use values of populations, but no figures available due to lack of data. Intermediate goods comprise the purchase of livestock, feed and grazing, animal health.

Ad 2: Employment of the TEV concept. Indirect use-values are assessed by data on private costs and benefits: the value of water is derived from municipal water supply costs, pollination values of fynbos vegetation from on gate farming prices for honey and trough less fruit and crop production. Estimations about the potential value of genetic resources vary from US$80 million to US$700 million. The aggregation to the total value of R455 million is unclear (R7 = US$ 1).

Ad 3: Analysis of 8 scenarios with different benefit-cost ratios. Ecological impacts are identified by a questionnaire survey. Impacts are evaluated via ordinal values without quantitative numbers. Benefits are: nitrogen fixation, possible medicinal use of astringents and styptics, combating erosion. Cost are: increased erosion after increased fire intensity, destabilisation of river banks, loss of recreational opportunities, aesthetic costs, nitrogen pollution, loss of grazing potential, loss of biodiversity. Fire hazards, water supply and carbon sequestration are quantified indirect values. The last two come from data on social costs and benefits. Fire hazards are evaluated in terms of increased fire management costs. No remarks on whether data of private or social costs are used.

Ad 4: Zavaleta conducted a classical Cost-Benefit Analysis, comparing "do-nothing" with "eradication and restoration". In fact data on social costs and benefits was applied as it account for forgone alternative use of the water for public hydropower, irrigation, municipal facilities and fold control.

Ad 5: Accounted costs stem form willingness-to-pay studies that are transferred, i.e. the study employed a utility transfer of other studies results.

Ad 6: Two cost scenarios: (1) current mechanical and chemical control costs of US$ 1308, and (2) biological control costs of US$1511 at 2000 prices. Biological control is supposed to be more cost-effective due to the extinction of *A. filicoides* populations plus damage costs are avoided. No remarks on whether damage costs occur with control to keep the status quo of invested areas, or whether damages are avoided by control. Miscellaneous costs mainly cover loss of property prices in housing estates bordering on infested water bodies. Social costs have a high standard deviation.

Ad 7: Stated costs of US$11.75 billion are not reproducible. The eradication programme "working for water" creates income in a region with high unemployment. Wit *et al.* account for this investment as a benefit of US$20 million. It is unclear whether the amount corresponds with a utility transfer for a programme financed by the government. Survey of economic consequences of South African invasive species infestation without detailed analysis of private and social costs and benefits.

Ad 8: Surveys of economic and ecological impacts and qualitative remarks on both. The need for willingness-to-pay studies is addressed. Health costs are economic costs.

Ad 9: Potential double-counting within analysis of 2 projects. Project 1: impacts of orchard mites, *Skeleton weed, Sirex* wasps and *Chondrilla juncea.* Project 2 depicts economic effects of *Echium*-species.

Ad 10: "Beneficial" species are agricultural crop species as well as domestic plants and animals. Figures based on 1975 prices, discount rate 10%. Indirect damage includes fouling damage caused by *dreissena polymorha*, outages by *Boiga irregularis*.

Ad 11: Potential benefit-cost ratio analysis of an ex-ante scenario of preventive mitigation. No remarks on diminishing effects of control.

Ad 12: Quantification of environmental costs mainly match control costs. "Beneficial species", see Ad 10.

5.5 *Studies addressing the scope- and the uncertainty*-problem

Throughout the previous chapters two problems have gained major interest because they are especially relevant for economic assessments of biological invasions. It is the *scope*- and the *uncertainty*-problem. Both have been mentioned to influence and determine the quality of economic assessment results in their suitability for decision support. Hence, they will be explored in detail in the following.

5.5.1 Scope of impacts to be considered

As explained in section 4.2 the number of impacts of biological invasions is large and each is different and affects human well-being in a different manner. Many economic studies concentrate on direct impacts which are easy to quantify, such as production losses in agriculture or fisheries. Indirect effects, i.e. indirect-use and non-use values, are often neglected as they are not directly reflected in markets and have to be measured separately. The majority of societal effects occur indirectly.

As a result of the previous chapters one challenge of every economic assessment of biological invasions lays in the scope of impacts to be considered. This means an assessment ought to cover all value categories of the TEV but also primary values have to be regarded. This has been referred to as the *scope*-problem.

In section 4.2 the TEV has been introduced to show that the concept covers all values of ecosystem properties. Within this survey the TEV was used to illustrate the comprehensiveness of current studies for such instrumental values. Table 9 shows how the surveyed studies take the different TEV categories into account[45].

Economic studies emphasise on direct use-values. This can be explained by the availability of quantitative data on production decrease, weed and pest control within land use sectors. The more we leave the field of palpable use-values, the more such values become disregarded. Although indirect use-values such as decreased pollination functions, water availability and fire hazards within ecosystems, and the carrying capacity of pasture land were considered by the studies by Turpie and Heydenrych (2000), Bertram (1999), and others, a whole string of possible alternations in indirect use-values and option values were neglected, such as

[45] Although the TEV has been criticised in the thesis, the TEV has exclusively used in this survey because it concentrates on economic valuation of biological invasions and not on the physical assessment of such.

alternative landscape functions for tourism (recreation by hiking, bird-watching).

Table 9: Assessed value types of the studies surveyed (own source)

Source	Use-Values			Non-Use Value
	Direct Use-Value	Indirect Use-Value	Option Value	Existence / Bequest Value
Bertram, 1999	X	X	X	
Turpie and Heydenreich, 2000	X	X	X	X
Wit *et al.* 2001	X	X		
Zavaleta, 2000	X	X		
Engeman *et al.* 2004		X		
McConnachie *et al.* 2003	X			
Wilgen, 2001	X	X		
Reinhardt *et al.* 2003	X	X		
Cullen and Whitten, 1995	X			
Tisdell, 1990	X			
Pimentel *et al.* 2001	X	X		
Pimentel *et al.* 2002	X	X		

Reinhardt *et al.* (2003: 19) for instance accounted for health costs caused by *Ambrosia artemisiifolia.* They considered the costs of medical treatment to combat allergic reactions in humans. However, there are more values to be considered due to the loss of well-being by having asthma, and the costs would certainly rise by adding such results.

In section 4.5 additional values outside the typical scope of the TEV, i.e. primary values have been mentioned. Surveyed studies do not mention such primary values explicitly which are necessary for the self-organising capacity of ecosystems and enable the flow of usable goods, although some studies acknowledge that there are values that are "incalculable", such as the loss of an endemic species or a unique ecosystem (Engeman *et al.* 2004: 4).

5.5.2 The role of uncertainty

As already explained in chapter 3, uncertainty surrounds the invasion issue. On the one hand it refers to the outcomes biological invasions can cause. On the other hand it shapes the success of a strategy. With the review it has been analysed that none of the studies either reflected on ecological uncertainty stemming from the species development itself or respective environmental conditions or on the insecure success a strategy. It turns out that uncertainty does not play a vital role in the analysed economic studies. During the course of classical cost-benefit analyses, in the majority of cases a sensitivity analysis was conducted, either by employing different discount rates (Cullen and Whitten, 1995; Pimentel *et al.* 2001; Tisdell 1990) or by comparing different management regimes (McConnachie *et al.* 2003; Wit *et al.* 2001) to slightly acknowledge that there is uncertainty regarding the outcomes of an applied strategy. However, the sensitivity analysis does not refer to ecological behaviour. Conducted sensitivity analyses neither considered the continuing dispersal of species into adjacent ecosystems nor reflected the possible degradation of ecosystem functions in the course of further impacts or re-invasion after eradication. Bertram (1999: 49) remarks on New Zealand's "expenditure on border controls and quarantine services which is akin to the payment of insurance premiums against catastrophic events [...] as means of confronting risks and uncertainty". This means, economic studies are often aware that there is uncertainty regarding the impacts of invasive species. However, to refer to them explicitly and to detect them where they occur in the assessment process, has often been disregarded.

5.6 *Conclusions from the survey*

The review confirms the impression Wilgen *et al.* (2001) had in their overview of South-African studies, namely that "attempting an objective analysis and summary of the studies (of economics of biological invasions) that have been done is frustrating, as every study has used a different approach, making an accurate assessment of aggregate impacts impossible" (Wilgen *et al.* 2001: 154). However, with a systematic approach it is possible to present an overview of current studies with some general features to be discussed. Next to the heterogeneity of approaches the analysis illustrates also methodological weak points of economic valuations of biological invasions. The results of the analysis of studies of the costs and benefits of biological invasions can be summed up as follows:

- There is an imbalance in all examined studies: they focus on *ex-post*-evaluation, on control strategies, on few countries, and on agriculture. Clearly missing are *ex-ante* studies, evaluation of prevention strategies, and valuation results from a larger range of countries.

- Studies regarding impact assessment by monetising impacts and respective strategies mainly focus on the agricultural and sylvicultural sector and in most cases assess biological control strategies. A comparison of evaluation studies (far less an aggregation of their results) is not meaningful, as the range of methods, TEV categories assessed, the spatial framework and the number of invasive species differ too much.
- Economic studies mostly do not mention the hierarchical three-stage approach contained in the CBD.
- Even though many studies investigate a single case of invasion in detail, total cost figures should be used with caution, due to uncertainty and the existence of values which are difficult to assess in monetary terms. Hence they should not provide the sole basis for decision-making.
- None of the decision-aid studies considers all relevant economic effects in either a comprehensive (with respect to the TEV) or a methodologically sound way (e.g. by considering only data on social costs and benefits). To consider all TEV-categories does not mean they have to be evaluated in any case, it means, that at least a qualitative remark that there are changes or not should be done. Furthermore primary values have not been mentioned. None of the studies realises the whole scope of impacts, and therefore the *scope*-problem can be confirmed for the studies at hand.
- The *uncertainty*-problem can be also confirmed for the surveyed studies. Uncertainty, especially arising in the ecological context of the invasive process, is not explicitly considered or dealt with. Sensitivity analysis is the way economic valuation studies deal with uncertainty in the majority of cases. This only happens after the valuation process in CBA (Pearce *et al.* 2006: pp. 60). One explanation why uncertainty has been neglected is the restricted ecological data. If even ecologists are not able to predict an invasion, and results are very single specific it is even more challenging to gain data on social costs and benefits that is so often missing. However, it has been stated in section 4.5, that even if incomplete knowledge lead to an incomplete evaluation, it has to be made explicit what we do not know.

What are options out of the problem to recognise that no systematic approach for a valuation of impacts and management options of invasive species currently exists?

One might be tempted to demand more comprising evaluation procedures. However, this would imply that general problems of any evaluation of invasive species are neglected. The problems are closely linked with a missing analysis structure for all such studies when using the TEV. These problems have been explained in section 4.5.4.3 and refer to the second, the valuation step of economic assessments. Review also shows that the role of a physical assessment always remains unclear. Some studies do mention that they use the TEV, but in general they do not explicitly address of a preliminary description and systematic record of impacts on ecosystem properties by invasive species. The missing analysis structure results in the non-comparability of the studies mentioned above, as each study focuses another set of ecosystem properties that are changed by biological invasions. It also explains why it has been difficult to draw a firm boundary between economic impact assessment and decision aid studies. A missing systematic base which explains the objects of assessment, i.e. whether and to which impacts, and whether and to which management options monetary costs and benefits are assigned, make it even more difficult to make the distinction.

So, if the review confirms that economic studies on costs and benefits not only tend to focus on direct use-values and therefore neglect non-use values, also refrain from taking primary values into account, and only touch the problem of uncertainty, one might wonder how such challenges can be improved.

One distinct option to approach that problem would be the search for a suitable analysis structure that can guarantee that both, the *uncertainty*- and the *scope*-problem can be included. Searching for such a solution means to go back to the very first step of any economic assessment, to the physical assessment. This development of an analysis structure for the physical assessment will be elaborated in chapter 6.

6 Enhancing economic assessments with the concept of ecosystem services

6.1 The concept of ecosystem services

Previous chapters have shown that economic assessments of biological invasions face several problems. Among them the *scope*- and the *uncertainty*-problem have been mentioned in particular. To approach these problems, in the first place, it is necessary to focus on the entire assessment procedure, and not merely on the economic evaluation part of the process. As was already mentioned in the first chapters of this thesis, the physical assessment is a crucial step of any economic assessment, as it determines the quality of the following, economic-evaluation step. However the literature review in Chapter 5 demonstrates that current literature on the economics of biological invasions generally overlooks the necessary step involving a distinct physical assessment. This results in an unstructured and incomplete analysis of impacts, often dealing with only some selected effects of biological invasions.

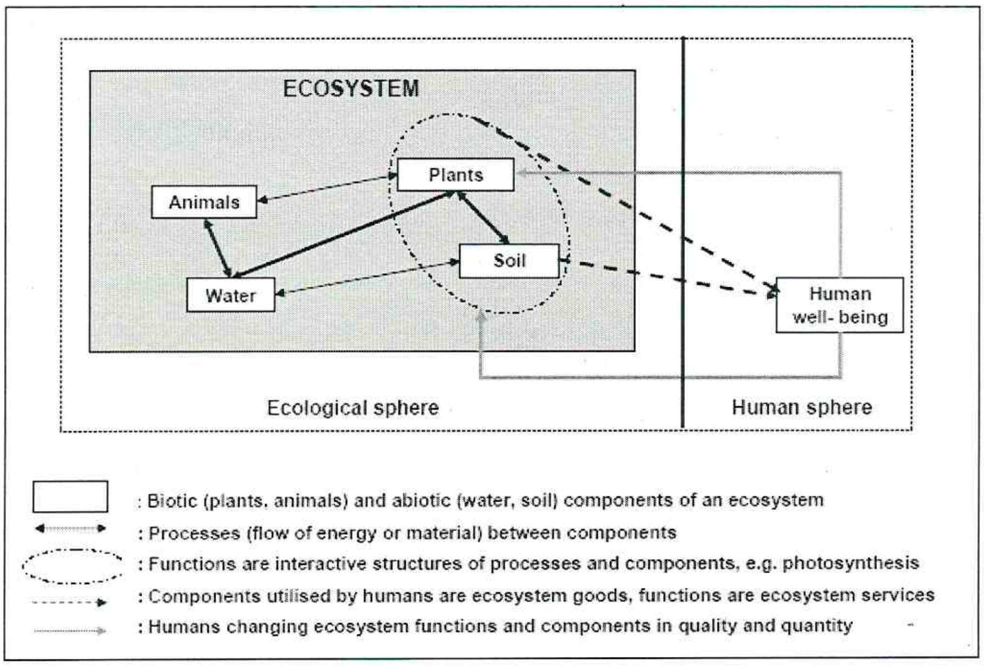
As a consequence, it is necessary to consider the *scope*- and the *uncertainty*-problem already at the physical assessment stage, and at the same time to search for a conceptual approach that will improve the procedure at this early stage of any economic assessment with the purpose of achieving a systematic analysis. For this, the concept of ecosystem services will be introduced here. This concept offers opportunities to improve the basis for an economic assessment of biological invasions and thus provides important improvements regarding the identified problems with *scope* and *uncertainty*. In the end these improvements contribute to enhance the physical assessment of the impacts of a biological invasion.

In the following paragraphs, the extent to which the concept delivers a conceptual contribution to improve the physical assessment of biological invasions' impacts, as a basis for an economic assessment of biological invasions, will be explained. First, in section 6.1 the concept will be introduced, and its origin and current relevance will be described. As a part of this section its relation to the concept of biodiversity is explained. In section 6.2 the concept of ecosystem services is linked with the phenomenon of biological invasions, i.e. impacts of invasive species are illustrated parallel to the list of ecosystem services. Section 6.3 illustrates advantages of using ecosystem services for an economic assessment of biological invasions. First of all, different conceptual contributions to tackling the problem of *scope* will be demonstrated, then advantages of ecosystem services to addressing the *uncertainty*-problem will be discussed, while finally the additional opportunity of a stakeholder analysis connected to ecosystem services will be presented. Section 6.4 aggregates the findings from the previous sections.

6.1.1 Origins of the concept

Ecosystem services are understood as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life. They maintain biodiversity and the production of ecosystem goods, such as seafood, forage and timber [...]." (Daily 1997: 3). The underlying basic assumption of the concept is the existence of a strong linkage between ecosystems and human well-being, as ecosystems provide goods and services which generate benefits to humans. The concept explicitly addresses the fundamental dependency of humans on the functioning of ecosystems, because without these the vital flow of goods and services that constitute the foundations for human life ceases, and human well-being decreases. Figure 17 depicts the basic assumption of the concept: each ecosystem is considered as a complex system of ecological structures, containing ecological processes, and biotic (living organisms) and abiotic (physical and chemical) components (Limburg *et al.* 2002).

Figure 17: Ecosystem functions and components providing goods and services to humans (own source)



Processes are interactive structures between biotic and/or abiotic components, e.g. the flow of matter or energy. A function, in turn, is a complex of components and processes, e.g. if plants use light for photosynthesis. A function is characterised by the interdependency of components and processes. Ecosystems are regarded as a "fundamental ecological unit" (Lindemann 1942: 415) where the "biotic community cannot be clearly differentiated from its abiotic environment", as the specific complex processes between the biotic community and the abiotic environment shape each different ecosystem (ibid). This ecological unit is also characterised by a spatial boundary. Within the spatial area the specific ecosystem components, functions and processes are prevailing. Ecological functions and components in various ways contribute to human well-being (see below). If ecological function are considered in their benefiting character to humans one talks about ecosystem services; if ecosystem components are beneficial for humans, they are called ecosystem goods.

Human well-being, in the definition given in the context of ecosystem services, has various components. They comprise the basic material for a good life, freedom and choice, health, good social relations, and security (Millennium Ecosystem Assessment 2003: 12). How well-being, or poverty as the deprivation of well-being, is experienced depends on the context, i.e. on the physical environment, and on social and personal factors, such as gender, age, and the recipient culture. No matter the degree of complexity with which human well-being is conceptualised, ecosystem services form a fundamental basis for it.

The roots of the concept of ecosystem services are ancient. Already Plato knew of the relation between the deforestation of Attica and the ensuing soil erosion and drying of springs (Daily 1997: 11). Today, the concept is founded on the scientific base of ecosystem ecology. The explicit discussion about the role of ecosystems dates back to the middle of the last century (Daily 1997: pp. 23; de Groot *et al.* 2002).

Starting in the 1950s, studies about the modulating role of ecosystems in regulating abiotic components, such as nutrients, sediments and water, illustrated the complex interdependencies of ecosystem functioning. Internal systemic feedbacks underline such dependencies (Odum 1953). Examples for ecosystem functions are food chain interactions where trophical dependencies between species are shaped by given physical constraints in a given ecosystem, such as soil or water availability.

The full meaning of the concept develops with the transformation of ecological functions into goods and services that are object of human consumption and perception (de Groot *et al.*

2002: 394). Ecosystem components, such as freshwater or plants, are goods if utilised by humans and thus contributing to human welfare. Ecosystem functions are considered as services if humans benefit from them. For instance in the case when humans benefit from the erosion-preventing capacity of vegetation cover on slopes. The explicit relation between humans and ecosystems, their ecological functions, and their components are the remarkable subject of the concept. The linkage between both, ecosystems and human-well-being, are the ecosystem goods and services provided by the ecosystem. Although humans are not an integral part of ecosystems they are an integral part of the concept, as ecosystem properties are of special interest due to their capacity to become beneficially utilized by humans. The concept explicitly addresses the linkage of ecological or social and human systems. Humans gain direct or indirect benefits from ecosystem functions or components. The regulation of nutrients, sediments and water in terms of litter and waste treatment is important for the production of food and wood. Microbial organisms in the soil enable the transport of nutrients and water by treating litter and waste and by recycling biomass. These processes enable the production of new organic matter, which, in turn, is food for other organisms, including humans. Interactions of biotic and abiotic components provide the basis for sustaining not only human life, but all life on earth. Ecosystem services in general yield a diverse flow of vital services (Chee 2004). They comprise basic life-supporting processes, such as oxygen production, freshwater regulation or the generation and renewal of soil fertility. The benefits derived from ecosystems are manifold. They include the production of goods, such as food, timber, seafood and fuel, but also non-material benefits are provided, as they carry information and socio-cultural values, necessary for spiritual and intellectual inspiration and for human enjoyment.

6.1.2 Current relevance

Humans are an integral part of the concept of ecosystem services. They do not only receive ecosystem benefits, but also drive the ongoing degradation of such. Human overuse causes the deterioration of ecosystem services and goods, both in quantity and quality. This dual role of humans makes the concept of interest for politicians, but also for scientists. The research community working on biological diversity employs the concept to show the relevance of the various dimensions of biological diversity (e.g. Hooper *et al.* 2005). But the concept has also been used in the field of ecological economics, linking ecological and social systems (Bingham *et al.* 1995), as the concept provides an explicit link between the ecological and the socio-economic sphere. The comprehensive approach enables the reflection on ecological, economic, as well as social aspects of ecosystems and their provision of services and goods. Ecological economics research is concerned with the valuation of ecosystem services and the challenge that is linked to it due to its various dimensions[46].

The relevance of the concept in environmental politics has increased in the last few years, as the motivation has grown to use the concept to guide human decision-making in the face of the rapid deterioration of the natural environment. There is a great demand for support to decision-making regarding the management of natural resources. The concept can be helpful not only to decision makers. It also identifies the need of scientists to provide information by translating complex structures and ecosystem components into a more limited number of ecosystem functions, and to explain why and to what extent such functions enable the flow of ecosystem services fundamental for human well-being. This information can be used for the planning and management of ecosystems.

The most prominent example in which the concept has been applied in the context of environmental politics is the UN-report about the status of the world's ecosystems, the *Ecosystem Millennium Assessment*[47]. This report addresses the need for sustainable management of the different ecosystems of the world and calls for an integrative approach to decision-making concerning the use of ecosystem services, in which decision makers understand the multiple effects any management or policy change can have on an ecosystem. Individuals, as well as public and private decision makers on all institutional levels (local, municipal, provincial, national, as well as international level), are recognized as drivers in the processes of change affecting ecosystems and their services. The report shows how differently humans are dependent on specific ecosystem services, and that all societies of the world face a strong degradation of these.

The explicit link of humans with ecosystems, on the one hand as drivers of degradation, on the other hand as perceivers of benefits, makes the concept of ecosystem services also interesting in the context of biological invasions. Humans have been recognised as drivers of the invasion process, but also as perceivers of damages (see section 2.2). How the concept can work in the context of biological invasions and how the different types of damages can be linked with the different ecosystem services and goods will be shown in the following.

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[46] For an overview of ecological economic literature dealing with the valuation of ecosystem services, see for instance de Groot (1992); Daily (1997); Goulder *et al.* (1997); Costanza *et al.* (1998); Villa *et al.*(2002).

[47] See: http://www.millenniumassessment.org/en/Reports.aspx

6.1.3 The classification of ecosystem services

The concept of ecosystem services states that ecosystems provide goods and services to humans. However, which are the kinds of goods and services supplied by ecosystems? The following chapter explains the different ecosystem services in order to elicit the variety of ecological properties and assets that are beneficial to humans.

A variety of classifications and categories of ecosystem services have been referred to in literature (Goulder *et al.* 1997; Costanza *et al.* 1998; de Groot *et al.* 2002; Limburg *et al.* 2002; Chee 2004). Some authors place emphasis on ecosystem functions, i.e. they focus on ecological structures of ecosystems that are beneficial to humans; others emphasise the flow of services and goods provided by such functions.

All literature, however, aims to illustrate the linkage between ecosystems and their beneficial functioning for humans. In the following work there is a strong reference to the nomenclature of the Millennium Ecosystem Assessment. This nomenclature has been chosen due to its suitability to show the problem of biological invasions[48]. Some slight deviations, however, have been made. Ecosystem services in this context comprise both the flow of services and the provision of goods.

The Millennium Ecosystem Assessment (2003) works with four major categories of ecosystem services. They are explained in the following.

6.1.3.1 Sustaining services:

Sustaining services are the basis for all other ecosystem services. They are different from all the other ecosystem services because their potential to benefit humans occurs indirectly and over a long time period. They comprise soil formation and retention, the global cycles of nutrients and water, the primary production and the provisioning of habitats.

Soil formation and retention means the accumulation, renewal and build up of organic matter by autotrophic organisms interacting with abiotic components in the soil, such as substrate and ground water. Primary production via *photosynthesis* is the first step of a production chain that, in the end, supplies a variety of viable biomass, either usable for other organisms, such as secondary producers or predators, or for human consumption. The global *nutrient* and *water cycle* are of long-term relevance and happen on the global scale. In the study of both

[48] Within the Millennium Ecosystem Assessment biological invasions are addressed to be one key issue in the degradation of ecosystem services (Millennium Ecosystem Assessment 2003: 86).

kinds of cycle, the interaction of terrestrial and atmospheric sources and sinks of nutrients and water are considered. The *provisioning of habitats* is strongly linked with the crucial role of biodiversity (see Section 6.1.4). Habitats are suitable living space for animals and plant species and have the function of being simultaneously both refugium and nursery. The provisioning of habitats is important for the maintenance of biological and genetic diversity, as habitats provide for instance suitable space for reproduction and are therefore crucial for vital stocks of harvestable ecosystem goods such as fish, livestock, and fruits. Such conditions are also important for non-usable species, such as microbes in the soil, which need space to live, grow and reproduce.

6.1.3.2 Provisioning services:

Provisioning services are the products that are obtained from ecosystems. This group of services embraces "goods" of ecosystems in the classical sense as they are tradable on markets.

They comprise the range of food products derived from plants, animals, and microbes, but also *fibre materials*, such as wood, jute, hemp, silk and many other natural materials. Natural sources of energy, such as dung, oil or wood, provide ecosystem *fuel* goods. *Genetic resources* include the genes and genetic information used for animal and plant breeding and biotechnology. This includes the maintenance and evolution of genetic material. *Biochemicals, natural medicines*, and *pharmaceuticals* can supply biocides or food additives such as alginates. *Ornamental* resources comprise animal skins, shells or flowers that are used by humans. *Freshwater* in the sense of a provisioning service provides a good usable for consumption.

The provisioning service of freshwater shows that certain ecosystem components or functions can have more than one role: freshwater availability is linked to the global water cycle (as a sustaining service), but also to the regulating service of water, in terms of the steering capability of ecosystems for run-off and infiltration, and can be considered as a consumptive good.

6.1.3.3 Regulating services:

Regulating services are the benefits obtained from the regulation of ecological processes in ecosystems. They are essential for sustaining life-supporting systems that indirectly sustain human well-being.

Climate regulation includes the maintenance of air quality, as ecosystems contribute chemicals to, and extract chemicals from, the atmosphere. This phenomenon occurs on the local as well on the global scale and is driven by precipitation and temperature. Climate regulation also comprises the regulation of greenhouse gases, as ecosystems enable the sequestration of these. *Erosion control* is possible due to the vegetation cover that prevents for example landslides in mountainous ecosystems. The *regulation of water*, i.e. the availability, run-off and renewal of water, but also the mitigation of droughts and floods, are examples for a regulating service. *Water purification and waste treatment* are based on the role of vegetation and biota to filter and decompose organic compounds in all ecosystems. *Pollination* as ecosystem service underpins the role of biota in the movement of floral gametes, pollen and seeds. Pollination is essential for the growth of every plant, and therefore the cultivation of harvestable crops, fruits or wild plant species are dependent on it. *Biological control* means that natural trophic-dynamic relations act upon species abundance in all ecosystems. Therefore natural enemies can be used for the regulation of weeds and pests by humans, but they also create a natural equilibrium of species in an ecosystem without human interference. Ecosystems can determine the abundance of pathogens and can function as a natural *regulation of human diseases*. *Storm and flood protection* as a regulating service provided by the ecosystem is predominantly relevant for coastal ecosystems, as mangroves can provide a buffer function for these coastal areas in the case of flood events.

6.1.3.4 Cultural services:

Cultural services are non-material benefits that people can obtain from ecosystems. Ecosystems are the source of a variety of information relevant for humans and carry religious, cultural, as well as scientific or historical objects.

Ecosystem properties shape and influence social relations, as the availability of natural resources determine their use by the recipient society. Fishing societies and nomadic herding ones present huge contrasts concerning social structures and the organisation of work and sustenance, for instance. The diversity of ecosystems promotes *cultural diversity*. Landscapes, single natural objects or ecosystem structures and processes provide information for *education*, and even for *science*. The examination of natural properties reveals certain chemical, ecological or physical causalities of processes and structures, and is therefore the object of many scientific investigations. Ecosystems, because they carry information, shape our current cultural knowledge systems, but also influence educational and *cultural heritage* *values*. As the concept of ecosystem services comprises semi-natural as well as natural ecosystems (Farber *et al.* 2002), certain ecosystems that are a relict of historical land-use present properties of semi-natural ecosystems with a high historical value, but often also with high *aesthetical value*, which can serve as source of *inspiration*.

Clear-cut boundaries between the different non-material benefits of ecosystem services are often difficult to draw, as religious habits will play a part in a society's cultural heritage. However, what becomes clear with these services is the high textual value in which ecosystem services have been used for years.

Table 10 summarises all ecosystem services and explains ecological processes and functions within ecosystems that enable the flow of goods and services.

Table 10: Ecosystem services, related processes and functions, and their relevance for humans (own source, derived from Millennium Ecosystem Assessment 2003: 52)

Ecosystem services	Related ecosystem processes and functions	Example/relevance for humans
Sustaining services		
Soil formation and retention	Accumulation of organic matter, retentive role of root matrix and soil biota	Maintenance of arable land
Nutrient cycling	Biota that enables nutrient storage and re-cycling, steering the global nutrient cycle	Maintenance of productive soils
Primary production and photosynthesis	Capability of biota and plants to uptake nutrients, basis for the production of biomass	Production of biomass, e.g. food
Provisioning of habitats	Suitable living space for animals and plants for reproduction, nursery	Maintenance of commercially harvested stocks of species
Water cycling	Global water cycle steered by ecosystems through filtering, retention and storage of fresh water	Natural irrigation through precipitation, medium of transport
Provisioning Services		
Food and fibres	Production of food, natural materials, precursors to synthetic products	Spices, forage, crops, fodder for animals, oil, dyes, rubber, timber, fibres
Fuel	Production of natural energy sources	Wood, dung, oil
Genetic resources	Evolution and maintenance of genetic material	Domesticated animals, cultivated plants, resistance to pests and diseases
Biochemicals, natural medicine, pharmaceuticals	Biological materials usable for medicinal products, biochemicals and pharmaceuticals	Biocides, food additives, drugs
Ornamentals	Ornamental use of plants and animals	Animal skins for fashion

Ecosystem services	**Related ecosystem processes and functions**	**Example/relevance for humans**
Fresh water	Provisioning of water for consumptive use	Drinking water
Regulating services		
Climate regulation	Regulation and balance of bio-geochemical cycles	Purification of air
Erosion control	Buffering erosion through ecosystem structures, e.g. vegetation cover	Forest and land cover on steep slopes, buffer stripes along rivers
Water regulation	Regulation of water availability, run-off	Drainage of soils
Water purification and waste treatment	Decomposition of organic matter in water, removal of nutrients and compounds	Decomposition of litter
Pollination	Animals as vector of gametes	Pollination of crops for the generation of food
Biological control	Population control through trophic-dynamic relations	Natural balance of pests
Regulation of human diseases	Natural regulation of number, vectors of pathogens, toxic substances	Non-existence of cholera-bacteria in dry ecosystems
Storm and flood protection	Buffering disturbance through ecosystem structures in ecosystems	Natural flood retention areas, e.g. mangroves, riparian forests
Cultural services		
Cultural diversity	Ecosystems and their functions influence recipient cultures	Different cultural habits adapted to ecosystems
Aesthetical values	Landscape, scenery having aesthetic value	Tourists enjoy "scenic roads"
Inspiration	Natural objects, places as source of inspiration for culture & art	Artistic motives for painters
Spiritual and religious values	Spiritual enrichment, nature carrying religious values	Worship of holy forests on the part of indigenous people
Scientific and educational values	Information for and object of study and education, ecosystems influence types of knowledge systems in different cultures	Excursions into unexplored nature
Recreation and ecotourism	Variety of landscapes with recreational uses	Landscapes for spending leisure time
Cultural heritage and historical values	Compartments of ecosystems (species, landscapes) with historical value	Natural historical monuments

6.1.4 The status of biodiversity in the concept of ecosystem services

With the various services and goods the concept of ecosystem services offers an opportunity to illustrate the manifold modifications brought on nature by humans. Biodiversity is a holistic concept as well. It has been developed by scientists in order to elicit the complex functioning of nature. One of its meanings is to highlight the various threats to the natural environment (Takacs 1996: pp. 41). In the nineties of the last century there has been growing interest in the role of biodiversity in supporting the functioning of ecosystems and in providing benefits to humans (e. g. Daily 1997: 16; Hooper 2005). Biodiversity includes diversity within and between species and diversity of ecosystems. Diversity has been recognised as "a structural feature of ecosystems and the variability among ecosystems is an element of biodiversity" (Millennium Ecosystem Assessment 2003: 9). Thus, both concepts are closely related (Millennium Ecosystem Assessment 2003: 8). Since the very beginning of research on that topic, the importance of biodiversity for humans has been emphasised and its role has been discussed in the context of ecosystem functioning (e.g. Ehrlich and Mooney (1983)). Whether the loss of biodiversity deteriorates the functioning of ecosystems or not, whether higher diversity stabilizes ecosystem functioning, and whether the productivity of ecosystems depends on its diversity, are all questions which have not been answered to date (Muradian 2001). At the moment there is no clear answer to the relation between ecosystem functioning and biodiversity (Hooper 2005). However, the dependency of ecological processes and various components of ecosystems have been acknowledged to be essential for biodiversity. This is why biodiversity has an outstanding role within the Ecosystem Millennium Assessment (Ecosystem Millennium Assessment 2003: pp. 60). Biodiversity is considered to be the basis for all the services, and in turn, is supported by all of them. Biodiversity affects all forms of ecosystem services, depicted in the four categories in section 6.1.3. Among them are key ecosystem processes, mainly in terrestrial ecosystems, e.g. the production of biomass (via photosynthesis), nutrient and water cycling, as well as soil formation and retention. It is also essential for other services. An example is pollination, which is the base for plant-derived ecosystem services such as crop or fruit production. Species composition, their interactions e.g. in food chains, but also their relative abundance are all aspects of biodiversity. Therefore, the concept of ecosystem services covers the entirety of aspects of biodiversity, just from the perspective of ecosystems and their beneficial role to humans.

The various impacts of biological invasions have often been linked with the loss of biodiversity (see section 2.3). Invasive species have been acknowledged to be a major threat to biodiversity (Oecd 1996: 47). Hence, the Millennium Ecosystem Assessment explicitly addresses biological invasions as one of the five human-made drivers to ecosystem change which are most important (Ecosystem Millennium Assessment 2005: 97; Ecosystem Millennium Assessment 2005a: 68). The concept of ecosystem services is well suited to show

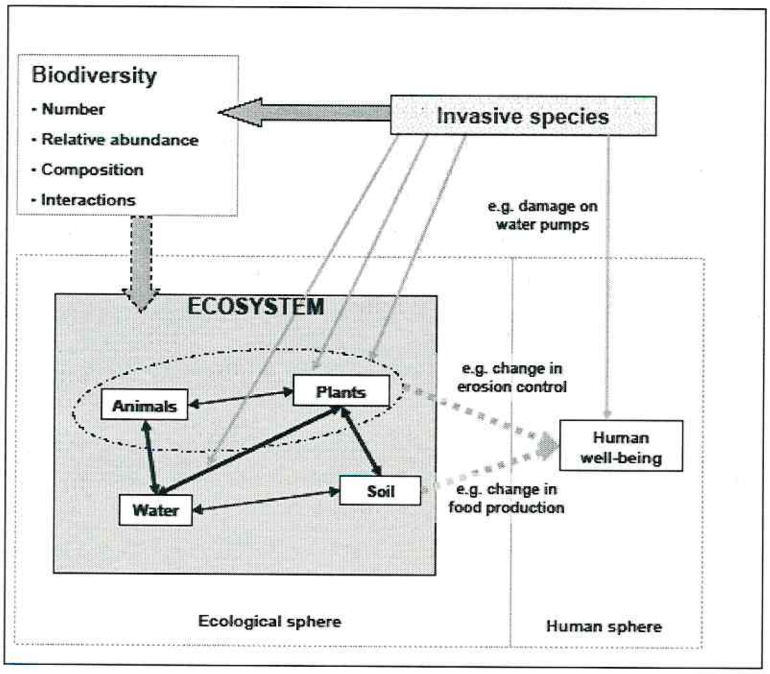
how manifold impacts of biological invasions are. By using the various ecosystem services the magnitude of different impacts can be depicted. They will differ in their sensibility to damages of invasive species, i.e. some ecosystem services are more prone to impacts of biological invasions than others. Biological invasions as major driver of ecosystem change have yet been described only in a general way. A detailed description of how biological invasions impact the well-being of humans has been neglected in the report so far. To consider the concept of ecosystem services in the context of biological invasions and its capacity to contribute to the *scope*- and the *uncertainty*-problem has been missing. This gap will be closed in the following. This means that the magnitude of impacts in the course of biological invasions is reflected in their effect of changing ecosystem services. This investigation aims to find out to which extent the concept is suitable for ecological-economic assessments of biological invasions. This will be explained in the following paragraphs.

6.2 Impacts of biological invasions on ecosystem services

The mere illustration of impacts of biological invasions by using ecosystem services has been done already (see for instance Binimelis *et al.* 2007; Charles *et al.* 2007). Within this thesis, however, not only a description of the different impacts on the various ecosystem services will be shown, its conceptual contribution to enhance economic assessments will be the focus, with major regard on the *uncertainty*- and the *scope*-problem. A profound investigation of the concept of ecosystem services regarding its suitability to meet both challenges of ecological-economic assessments is new in this context.

Usually impacts of invasive species are referred to the concept of biodiversity and to changes of such, see chapter 2. Changes in vertical and horizontal food chain processes, but also on ecosystem processes are often subsumed under shifts in biodiversity. Impacts touch inter-species relations and their functioning role in ecosystems, which has, according to the understanding of the concept, consequences on human well-being. This is different to the concept of biodiversity which illustrates impacts of biological invasions via their shifting property on all levels of biodiversity. By changing ecosystem processes, components and functions, invasive species alter the flow of ecosystem goods and services (see Figure 18). A shift of species composition in the vegetation cover of steep slopes might lead to a change of the regulating service of erosion control. A decrease in soil fertility will affect the provisioning service of food production. Thus, by using ecosystem services, the various impacts of biological invasions can be illustrated by grouping the different impacts along the different ecosystem services. Figure 18 depicts how impacts of biological invasions can be linked to ecosystems and related goods and services.

Figure 18: Invasive species altering ecosystem services by influencing ecosystem functions and components (own source)



Impacts of invasive species can be either illustrated via changes of biodiversity, which are often referred to species interactions, their relative abundance or number in the recipient ecosystem, or the specific species composition of the ecosystem at hand. They can be also directly related to the different biotic and abiotic components of an ecosystem and the specific processes between the components. The link with ecosystem services is, if changes in ecosystem functioning have an influence on human well-being, for instance, if the natural capacity of an ecosystem to regulate erosion or of crop production diminishes due to invasive species.

The scale on which impacts of biological invasions become obvious is normally the regional or local scale, owing to the spatial extent of an ecosystem. Nevertheless, there are also effects due to invasive species that are of global relevance, such in the case of species extinction which are irreversible (Nelson and al. 2006). Extinction implies that out of the total number of species in the world, one more get lost. To show how invasive species impact the different ecosystem services the four categories of Table 10 are taken up. Table 11 shows which impacts of biological invasions are relevant on the local, regional or global scale and how they affect each ecosystem service.

Table 11: Impacts of biological invasions on ecosystem services, processes and components (own source)

Ecosystem services	Impact on ecosystem process or component	Example of impact	Scale
Sustaining services			
Soil formation and retention	Increase/decrease in erosion, alternation in soil fertility, salinisation	Salinisation resistance of *Tamarix*-species on floodplains, USA[1]	Regional, local
Nutrient cycling	shifts in nutrient dynamics, uptake in the soil	Nitrogen accumulation of *Acacia*-species in the Fynbos vegetation, South Africa[2]	Global, regional, local
Primary production and photosynthesis	Shifts in primary production	Increase of algae due to the invasive predator (*Salmo trutta*) that decreases herbivores lead to increased eutrophication[3]	Global regional
Provisioning of habitats	Change in species composition within habitats	Threat of extinction of the Kakapo, New Zealand[4]	Global, regional, local
Water cycling	(see water regulation)		Global
Provisioning services			
Food & fibre	Decrease in crop, fish, wood production due to invasive species, increase/decrease in material resources	Decrease in native fish yields due to Nile perch, Lake Victoria[5]	Regional
Fuel	Increase/decrease in energy material	Increase in biomass production for energy source, South Africa[6]	Regional, local
Genetic resources	Increase in hybridisation, introgression	Hybridisation of wildcat and housecat[7]	Global
Biochemicals, natural medicine, and pharmaceuticals	Decrease in genetic pools due to biodiversity loss	Extinction of species with potential pharmaceutical value[8]	Global
Ornamentals	Change in number of ornamental plants and animals	Impact of the ginger strain *Ralstonia* on ornamental ginger, Hawaii[9]	Regional
Freshwater	Alternation in (fresh) water availability, quality	Improved water quality due to filtration function of *Dreissena polymorpha*, USA[10]	Regional
Regulating services			
Climate regulation	Micro-climatic alternation due to changing vegetation structure	Changing radiation on bare rocky positions through *Pseudotsuga menzesii*, Europe[11]	Regional, local
Erosion control	Buffering of disturbances through ecosystem structures, e.g. vegetation cover preventing erosion	Stabilising effect of *Fallopia*-species on riverbanks[12]	Local
Water regulation	Changes in local water	Decease in local fresh and ground water	Regional,

Ecosystem services	Impact on ecosystem process or component	Example of impact	Scale
	households	due to high evapotranspiration of *Tamarix*-spec., USA[1]	local
Water purification and waste treatment	Alternation in nutrient and compound, biomass removal	Sophisticated digestive diet of Possums causing changes in fallen canopy leaves, New Zealand[13]	Regional
Pollination	Pollinators as vectors, changes in pollinator functions	Fig wasp pollinators as vectors of exotic *Ficus* spp., USA[14]	Regional,
Biological control	Increase/decrease in invasive species population, IS as control agents	biological control of invasive species and as control agents, Australia[15]	Regional, local
Regulation of human diseases	Increase of human diseases mediated by invasive species or due the species itself	Skin injuries due to phototoxic substances of *Heracleum mantegazzianum*[15]	Regional, local
Storm and flood protection	Changes of riparian forests as flooding buffer	*Tamarix*-spec. dominating native riparian forest and changing its capacity to buffer flood events, USA[1]	Regional, local
Cultural services			
Cultural diversity	Differences in ecosystem features, such as species composition, decrease due to homogenisation of species by invasive species	*Senecio inaequidens* as ubiquitous plant at rail roads, decrease in region specific flora[16]	Global, regional, Local
Aesthetic values	Increase/decrease in aesthetic inspiration	Yellow flowering Oxalis pes-caprae for aesthetic inspiration, South Africa[17]	Regional, local
Inspirational values	See "aesthetic values"	See "aesthetic values"	Regional, local
Spiritual and religious values	*	*	Regional, local
Scientific and educational values	Change in observable natural phenomena	Invasive species as object of scientific research	Regional, local
Recreation and ecotourism	Shifts in recreational value of landscapes	*Rhododendron ponticum* as attraction for tourist in Wales[18]	Regional, local
Cultural heritage and historical values	Alternation in parks, home gardens, "cultural landscapes"	Increase in exotic plants for horticulture[19]	Regional, local

[1](Ellis *et al.* 1998; Bell *et al.* 2003); [2](Le Maitre *et al.* 2002); [3](Townsend 1996); [4](Clout and Pimentel 2002); [5](Kasulo *et al.* 2000); [6](Higgins *et al.* 1997); [7](Rhymer and Simberloff 1996); [8](Perrings *et al.* 2002); [9](Anderson and Gardner 1999); [10](MacIsaak 1996); [11](Klingenstein 2004); [12](Kowarik 2003 : 17); [13](Nugent *et al.* 2001); [14](Richardson *et al.* 2000); [15](Hill *et al.* 2000); [16](Lopez-Garcia and Maillet 2005); [17](Damanakis 1976); (Dehnen-Schmutz *et al.* 2005); [19](Kowarik 2003).
* For this ecosystem service no example has been found

Impacts of invasive species listed in Table 11 are now explained in detail and grouped along the four categories of sustaining, provisioning, regulating and cultural services.

6.2.2.1 Impacts on sustaining services

Impacts of invasive species on long-term and global ecosystem services, such as the global water or nutrient cycles, are very difficult to detect, as effects due to invasive species cumulate with other environmental problems (in the case of nutrient cycling, e.g. with emissions of trade and transport). The approximate proportion of invasive species effects on general changes of such ecosystem services can therefore be hardly established. Changes of such ecosystem services due to invasions' impacts become apparent on smaller scales, and will have an effect on a certain ecosystem. It can be assumed that effects of biological invasions on globally pertinent services will happen gradually and would only be recognisable in the long run. Whether or not they will affect human use of ecosystem services cannot be determined yet, as long-term studies are necessary to reveal such interdependencies. Shifts caused by invasive species might have too small effects on sustaining services on the global level up to now. To show, for instance, the contributing alteration of the invasive *Acacia*-species in the fynbos region to the global nutrient cycle or to the global water cycle is very difficult. However, on the regional scale, i.e. a bio-geographical region, a shift of nutrient contents in the soil has been identified which causes cascading effects such as changes in species composition in all of the ecosystem (Le Maitre *et al.* 2002). Therefore, impacts of invasive species on these services are considered on the local level.

Ecosystems supply habitat, refuge and reproduction space for species. Invasive species are mostly considered to be harmful due to their impact on the sustaining service "provisioning of habitats". Impacts of invasive species on this ecosystem service refer to the diversity of genes and species. Changes of species composition in the course of biological invasions are known for almost every ecosystem of the world. Certain countries, such as Australia, New Zealand or Hawaii, are highly vulnerable to such impacts, due to their isolated character. These islands flora and fauna experienced a strong co-evolution for millions of years, without human disturbance (see Section 2.3.1). Forty percent of New Zealand's terrestrial birds, for instance, have been lost due to invasive species, and over 40% of its remaining bird species are classified as under threat (Clout 2002). Breed disturbance is the main problem, caused by preying feral housecats and rats (ibid). The extinction of species is one possible aspect of biodiversity loss, which is accelerated through biological invasions. An example where a native, even endemic species is threatened due to invasive species is the endemic Kakapo (*Strigops haprobtilus*) on New Zealand, with a remaining world population of 62 individuals (Elliott *et al.* 2001). Feral cats and rats have had a severe impact on the population. But not only islands are affected. Alien species have also been recognised as problematic for countries such as the United States of America, in which an estimated 500 introduced plants have become pests. General mechanisms that determine the competitive advantage of invasive species over native species have been explained in Section 2.3.

6.2.2.2 Impacts on provisioning services

Biological invasions have impacts on provisioning services in various ways. They often affect the land-use sector. Effects vary from reduced timber harvest in forestry (Reinhardt *et al.* 2003: pp. 29) to decrease in native fish species yields (Hall and Mills 2000; Kasulo *et al.* 2000). But also the overgrazing of pasture sites caused by invasive rabbits proved to be "economically and socially catastrophic for farmers" in Australia (White and Newton-Cross, 2000: 118). The case of Lake Victoria, in which the Nile perch was introduced in the 1950s for reasons of sport fishing, aquaculture, creation of a new fish industry, and for controlling water pests (Kasulo, 2000:185), is just one of 18 large lakes in five countries where the introduction of alien species, that became invasive soon after its introduction, has been causing problems since and deteriorating the provisioning service of food (Hall and Mills 2000). Shifts in ecosystem structures impacted fish populations that are a significant source of food and income in the region of Lake Victoria. Interesting in this context is that the introduction of exotic species can have positive as well as negative effects. On the one hand, the increase in Nile perch yield poses a benefit On the other, the collapse of a fish community of more than 4000 native species constitutes a major damage (Hall and Mills 2000).

Concerning the supply of material and energy, again both negative and positive aspects have been recognised. The South African fynbos example is again an illustrative one. On the one hand, 15 invasive species are responsible for the decline in freshwater availability and generated repercussions on vegetation and human land-use in the region. On the other hand, invasive tree species constitute a reliable source of timber and firewood for the local communities (Higgins *et al.* 1997).

Impacts and alterations in the provisioning service of genetic resources are linked with the service of biochemicals, natural medicine and pharmaceuticals. This service should also be considered in the context of genetic loss (as another typical aspect of biodiversity loss). The maintenance of a pool of genetic resources is considered one of the tasks in preserving biodiversity. Maintaining genetic diversity serves to guarantee the potential use-value of species that might be discovered to have pharmaceutical agents (Perrings and Gadgil 2003). The extinction of species via hybridisation[49] and introgression[50] due to invasive species is one

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[49] Hybridisation is defined "as interbreeding of individuals from what are believed to be genetically distinct populations, regardless of the taxonomic status of such populations (Rhymer and Simberloff, 1996:84)".

issue that diminishes the variety of organisms and life forms (Rhymer and Simberloff 1996). The supply of ornamentals is a service that is negatively as well as positively affected by biological invasions. On the one hand there are examples where invasive species expel native species that are used as ornamentals or have an effect on the cultivation of these. On the other hand certain invasive species are a source of inspiration and carry aesthetical values due to their colourful blossom. An example for the positive consideration of an invasive species is the invasive shrub *Rhododendron ponticum* in Wales. Although the species affects coastal ecosystems, it has been highly appreciated by tourists that visit the coastline of Wales (Dehnen-Schmutz *et al.* 2004).

One of the most important provisioning services of ecosystems is the supply of freshwater. Humans are dependent on it in terms of drinking water, but also for irrigation, and production space of freshwater species. One well-known invasive species is the zebra mussel (*Dreissena polymorpha*). It has impacts on the quality of freshwater by intensively filtering water. The effect is on the one hand a decrease in native bivalves, e.g. in the Great Lakes in the US, but on the other hand a reduced amount of phytoplankton increases the water quality (Johnson and Padilla 1996). The zebra mussel, however, also poses a problem in water bodies, since it has the capacity to block pipelines that carry raw water through industrial facilities (Aldridge *et al.* 2004), and can therefore directly affect human-built facilities.

6.2.2.3 Impacts on regulating services

Many cases of biological invasions causing impacts on the regulating services, i.e. on regulating processes of ecosystems that indirectly benefit humans have been documented. Among them are effects of about 15 species (Australian *Acacia*-, *Eucalyptus*- and *Hakea*-species, and European and American *Pinus*- and *Prosopis*-species) on the South African fynbos vegetation, a bio-geographical region that has been already referred to (see Section 4.5.3). Some of the invasive species were introduced for reasons of timber and firewood supply (Higgins *et al.* 1997). This relatively small number of species is estimated to use 30-70% of the region's runoff (Le Maitre *et al.* 1996). Thus, the invasive species of the fynbos-region have an impact on water regulation functions. In the long run *Acacia*-species will also have an impact on nutrient availability in the soil, particularly on its nitrogen content. *Acacia*-species, as members of the plant family *Fabaceae*, enable nitrogen fixation through symbiotic

[50] Introgression is "gene flow between populations whose individuals hybridize, achieved when hybrids backcross to one or both parental populations" (Rhymer and Simberloff 1996: 84).

myccorhiza at the rhizomes of the plant (Kowarik 2003: 34).

Other impacts on soil conditions are known for *Tamarix*-species in America, where the species occupy riparian sites and where it is drought-resistant to sinking groundwater levels (Lesica and Miles 2004). The dense stands of the tree, its successful seed dispersal strategy, and its low habitat requirements, have caused it to replace native cottonwood and willow communities and lead to changes in the species composition of the natural vegetation structure. Salinisation occurs as a result of the higher evapo-transpiration rate of *Tamarix*-species in relation to that of former vegetation (Kowarik 2003: 34). Altered litter production and decomposition in riparian ecosystems due to changed moisture and leaf type are another result of *Tamarix*-invasions (Ellis *et al.* 1998).

Pollination is an essential ecosystem service for maintaining seed dispersal and therefore the constant regeneration of plants in general. In the case of invasive species, pollinators become important vectors for invasive species seed dispersal, as in the case of *Fig*-wasp pollinators that disperse exotic *Ficus*-species in America (Kaufmann *et al.* 1991) In Tasmania, recently introduced pollinators from New Zealand, honeybees (*Apis mellifera)* and bumblebees (*Bombus terrestris*), are expected to promote the invasion of so called "sleeper weeds" in the case that both species support the seed dispersion of such plants. These plants are currently not problematic due to them lacking suitable pollinators (Stout *et al.* 2002). Some studies claim these species have a problematic potential (ibid).

In the context of invasive species' impacts, biological control has two aspects. On the one hand a variety of exotic control agents are applied in the field of biological control as a strategic measure to address pests and weeds. The introduction of myxomatosis in Australia to reduce the explosive number of rabbits and the significant damages caused by them (White *et al.* 2000) is just one case where biological control agents have been able to decrease the variety of adverse impacts caused by an invasive species. On the other hand invasive species might experience an evolutionary process to adapt to other food sources, and the release of exotic insects entails a risk to native plants and communities. There are examples of phytophagous species that have been properly adapted to invasive plants (Jordon-Thaden and Louda 2003).

The Japanese Knotweed (*Fallopia japonica*) is a plant initially introduced to Germany in the beginning of the 19th century. It has been used for fodder and planted as ornamental in parks, but also for the stabilisation of embankments and for greening industrial sites (Kowarik

2003: 215). The plant, then, was considered valuable for several reasons. A negative effect of its introduction is that the dense stands of *Fallopia* hinder water run-off after floods in riparian areas. An invasion of *Fallopia* on forest sites furthermore restricts the re-growth of understorey (Kowarik 2003: 221). The Japanese Knotweed, therefore, is an example where an invasive plant can have both benefiting (stabilising embankments) and damaging (prevention of water run-off) effects concerning humans.

6.2.2.4 Impacts on cultural services

Invasive species also influence the non-material benefits of ecosystems. Ecosystems supply unique aesthetic properties which can be of high cultural value for the people in the recipient country. Cultural identity in Australia for instance is inevitably linked with wilderness areas (Russell 1989), whereas in Europe semi-natural landscapes developed by humans over hundreds of years carry cultural identity for many European societies (Eser 1998: pp. 115). Next to such differences in cultural values, landscapes carry historical information and belong to the cultural heritage of a nation. It can inform about former types of land use and its special feature, e.g. remains of the spatially small cultivation fields in Central Europe during the Middle age (Wegener 1998: 32). Invasive species have been changing the cultural services of ecosystems. Typical sandy regions of Eastern Germany with natural stands of pines have been changing in some parts through the invasive shrub *Mahonia aquifolium*, which grows in the undergrowth of pine forests. Impacts of invasive species are also known for National Parks and wilderness areas. The frequent occurrence of seedlings and small saplings of *Pinus radiata* in the Myall Lakes National Park, Australia, are just one example where visitors consider such species as disturbing the aesthetic experience of such places (Brown 2001).

Cultural services of ecosystem functions comprise also objects of scientific interest, for which biological invasions are significantly important. Research considers biological invasions as "biological tracers from which we (the scientific community) can extract valuable information on the dispersal of established species or future invaders" (Johnson and Padilla 1996: 25). A whole community of conservationists and scientists focuses on biological invasions to explain traits of species, in order to find patterns of dispersion, and find approaches that can predict future invasions. A great number of invasive species are of extraordinary scientific value.

Another cultural service is the use of ecosystems for recreation and ecotourism. Ecosystems carry values that are attractive to people which motivate them to visit such places. Invasive species affect these ecosystem properties. On the one hand, there are examples of species that are highly attractive to tourists, such as the example of dense stands of *Rhododendron ponticum* at the coastline of Wales (Dehnen-Schmutz 2005). People even travel to places just to watch and visit these species. On the other hand, some species are harmful to human health, which can lead to the avoidance of infested recreation sites. Pathogens or toxic substances of invasive species, such as the phototoxic secretions of the invasive Giant Hogweed, are a reason why people might avoid visiting places where the species grows. These examples show that invasive species can be both positive as well as negative incentives for people to visit places and thus, can have an increasing but also a reducing effect on the ecosystem service of recreation and tourism.

To summarise, the previous section has underlined how manifold impacts of invasive species are. Changes of ecosystem services have been recognised all over the world and for all different services. The overview also showed that changes of ecosystem services are not necessarily damaging. Some of the impacts of invasive species have been considered to be an increase of the respective ecosystem services, and hence positively affect this ecosystem service. The growing research field on biological invasions is an example for the latter kind of impacts. The illustration is well structured as impacts are assigned to each ecosystem service. How far this contributes to an improvement of economic assessments of biological invasions will now be explained.

6.3 Advantages of the concept of ecosystem services for economic assessments of biological invasions

As shown in the previous sections, the concept of ecosystem services can, in a first step, help to illustrate the magnitude of the biological invasion's impacts. This is not new (see for instance Binimelis *et al.* 2007; Charles and Dukes 2007). A description of the impacts of a biological invasion along the different ecosystem services is only one aspect of how the concept can be used. It also offers conceptual advantages for economic assessments. The concept of ecosystem services can contribute to address (i) the *scope*-problem and (ii) the *uncertainty*-problem, which have been located in the stage of the physical assessment.

Although ecosystem services have yet been mentioned to provide a systematic typology and comprehensive framework for integrated assessments, the aim was always to support economic valuation of ecosystem functions (de Groot *et al.* 2002). In the following, the perspective on ecosystem services to support decision-making is different. The focus will be on the concept's capacity to deliver a conceptual contribution to improve the physical assessment of biological invasions' impacts, not only the economic valuation. To elicit the conceptual advantages of the concept for the physical assessment, section 6.3 is structured as follows: section 6.3.1 demonstrates how the concept of ecosystem services can help to meet the *scope*-problem. It explains how ecosystem services can serve as analysis structure to supplement the TEV. Section 6.3.2 explains how far the *scope*-problem can be approached with the help of the concept of ecosystem services and how it provides an enhancement compared to other economic assessments. Section 6.3.3 presents an additional advantage of the concept of ecosystem services. It is an attached stakeholder analysis. With the help of such an analysis recommendations for the management of invasive species can be given.

6.3.1 The concept of ecosystem services contributing to the *scope*-problem

6.3.1.1 Ecosystem services functioning as analysis structure

What was presented in section 6.2 looks like a mere description of impacts of biological invasions. However, it is more than that. While describing impacts along the various ecosystem services, the concept structures the available information. A structure that supports the analysis of biological invasions' impacts and enables a systematic catalogisation of affected ecosystem services for a physical assessment.

The TEV has been suggested to be used for the economic valuation of goods and services provided by ecosystems (Millennium Ecosystem Assessment 2003; eftec 2005). However, it has been claimed that a physical assessment and therefore a defined analysis structure, is indispensable as the precondition for the monetarisation of impacts with ecosystem services. Section 4.5.4 showed the TEV to be unsuitable to serve as an analysis structure for a physical assessment, as it focuses only on economic valuation. Although the TEV accounts for a total of values, i.e. the concept requires the economic valuation of *all* ecosystem properties relevant for humans, it has come under criticism for being too abstract to identify which properties, goods, or services are actually relevant for the economic valuation, and which ones fall outside the scope of the TEV (see Section 4.5.4.2). It became clear how crucial it is to describe the attributes of goods that are considered within an economic valuation along the TEV categories as it only tells the usability of them. As this exclusive economic concept does not address the physical assessment, the concept of ecosystem services can complement it. The concept of ecosystem services can depict the specific character of the ecosystem goods and services. The list of ecosystem services, grouped into four categories can serve as analysis structure, i.e. for each ecosystem service an impact of an invasive species can be recorded and analysed. Absolute completeness regarding all potential effects of biological invasions, of course, cannot be guaranteed with a total of 26 different ecosystem services. The crucial advantage of the concept, however, is that an encompassing overview on all dimensions of impacts of invasive species can be given and thus the vast majority of impacts is covered by the mentioned ecosystem services.

To work with the concept of ecosystem services also means to use a list which allows a standardisation of the physical assessment when the various impacts of an invasive species have to be considered and decisions have to be taken whether and which management is appropriate. The basic structure of the analysis is always the same as all assessment refer to the same determined set of goods and services. The concept, furthermore, helps to show to a certain degree interactions between ecosystem components. In the case of sustaining services, for instance, it acknowledges their fundamental role for maintaining all other services[51]. Specific interactions between single ecosystem services are mostly unknown. In this context the inner systemic feedbacks and cascading effects due to invasive species' impacts are also often unknown. The advantage of using the concept of ecosystem services is the possibility to identify for each ecosystem service how much is known about a certain impact of an invasive species. To elicit the degree of knowledge for each ecosystem service has been referred to as the *uncertainty*-problem.

Thus, with the determined set of objects to be recorded (namely the specific ecosystem service changed by an impact of an invasive species) the concept enables a systematic and well-structured analysis of impacts of biological invasions in the stage of the physical assessment. It helps to reveal that one species can influence several ecosystem services, for instance in the case of the invasive *Tamarix*-species which affects the local water regulation, but also soil formation and retention due to its high evaporation rate (see Table 11). The focus can also be on a region, i.e. for a certain spatially defined area one can record and describe the different impacts of invasive species that occur within this target region on the different ecosystem services. Such systematically recorded impacts can now enter the economic valuation procedure.

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[51] This has implications for the economic valuation of ecosystem services, as double counting states a problem (see for instance Hein *et al.* 2006). This, however, will not be addressed here as it is outside the scope of the thesis.

6.3.1.2 Sustaining services as primary value

In addition to the necessity of an analysis structure for the goods to be valued, the TEV has been criticised for neglecting the primary value which is outside the scope of economic valuation (see 4.5.3). Although this value is outside the scope of the TEV, there is the option to take it into account in the physical assessment after all. Namely, since the concept of ecosystem services refers to this value, it can also be included in a physical assessment. In the first stance the primary value will be recognised as a change of the sustaining services. A quantitative measurement, however, will be difficult to make. The visualised alterations in such ecosystem services seem to be important information to take into consideration when taking decisions, especially when we know that invasive species impact the provisioning of habitats for threatened species.

As explained above, sustaining services constitute the basis for all other goods and services. The ecosystem services concept thus shows interdependencies between several goods and services because it acknowledges that certain processes are substantial for the generation of most of the other goods and services mentioned in the concept. Photosynthesis, for instance, provides the very basic condition for the production of any kind of organic matter. Substitutability between the sustaining services that create benefits to humans is therefore not possible. Furthermore, the whole array of ecosystem services listed is necessary for the well-being of humans, and it is therefore assumed that as soon as the availability or quality of one service starts to change, well-being as a whole is affected. Sustaining services make it clear that there are ecosystem services that do not correspond to any TEV-category, as they are non-instrumental and outside the scope of the TEV.

To date, there are actually aspects of indirect use-values of sustaining services that have been included in economic valuations. An example is the provision of nutrients in the soil for the production of agricultural goods. However, there are also aspects of sustaining services that are outside the TEV nomenclature in general. A good example is the irreversible shift of a habitat driven by an invasive species that results in the extinction of a native species. This irreversible change of the sustaining service "provisioning of habitats" is not amenable to marginal valuation, which not necessarily leads to the consequence that the impact is left outside a physical assessment, even more so as it is one of the most important and serious impacts invasive species can have. In the case of "nutrient cycling", there is also awareness of the fact that nutrients are a fundamental constituent of organic matter and thus essential for the maintenance of every form of life in an ecosystem. Sustaining services therefore cover the idea of primary values, which implies that certain properties of an ecosystem are essential for the self-organizing capacity of the system as a whole. They are the glue for all ecosystem components and processes (Gren *et al.* 1994).

The idea of primary values, however, goes beyond sustaining services. In the understanding of primary values, not only sustaining services are essential for the self-organizing capacity of ecosystems: it is actually the completeness of all structures that sustains an ecosystem as a whole. Referring to the concept of ecosystem services, this now means pointing out the complex interdependencies between all other ecosystem service groups as well, i.e. between provisioning, regulating and cultural services. The multidimensionality of processes, their feedbacks and interdependencies between the different ecosystem services, have been rather neglected by the concept up to now. Such shortcomings are results from the difficulties and remaining questions regarding the detection of such relations. To understand the complexity of ecosystem structures, components as well as processes, which build up an ecosystem and which enable the flow of goods and services, will require much further investigation (Limburg *et al.* 2002).

To summarise, the previous sections indicate the possibility to use the concept of ecosystem services for a physical assessment of invasive species′ impacts. It thus supplements the concept of the TEV as it provides a clearly defined framework for every economic assessment. This allows a standardisation of the procedure. As the concept also touches the idea of primary values, impacts on these services, which are neglected by an economic evaluation, can be taken into account in the physical assessment of biological invasions. With the encompassing analysis structure of the 26 ecosystem services, one additional advantage can be pointed out. It is the option to present uncertainty regarding impacts of biological invasions.

6.3.2 The concept of ecosystem services contributing to the *uncertainty*-problem

6.3.2.1 Detecting uncertainty with the help of ecosystem services

When the concept of ecosystem services has been mentioned to offer the possibility to structure available information about impacts of invasive species along each ecosystem service, incomplete knowledge was also meant. It can be also depicted at the same time. If available information is scarce, we have to deal with uncertainty, a fact that can be visualised via the list of ecosystem services. To use the concept of ecosystem services also in the context of the *uncertainty*-problem has been found out to be the great advantage since uncertainty is high regarding (i) biological invasions` impacts and (ii) the success of the management (see chapter 3). It helps to depict what type of knowledge for each ecosystem service is available. This means the degree of knowledge of the impacts by an invasive species can be determined for each ecosystem service. The optimal situation is when available information allows of the definition of all possible outcomes of an invasion with all attributed probabilities. In such a situation we are in certainty (see chapter 3). Such a situation only occurs if an invasive species has already become invasive in the target region and the actual decision is concerned with the management of impacts ex-post the invasion. In such a case both quantitative and qualitative data are available about the impacts. Usually an economic assessment records only those impacts that are apparent to the decision maker, and only obvious impacts constitute the analysis structure of an economic valuation of impacts, as proven by the literature survey (see chapter 5). However, there are many more impacts of invasive species that are not certain, as our knowledge is incomplete. How the concept also offers the opportunity to detect when we are dealing with incomplete knowledge regarding both, impacts of invasive species and the ecological effectiveness of a management strategy will be explained in the following.

6.3.2.2 Situations of uncertainty regarding impacts of invasive species

It has been noted that with the help of the concept an embracing view on many different aspects of ecosystems that contribute to human well-being can be described. The different impacts of invasive species on the great number of ecosystem services can be illustrated as each impact corresponds to a change of one ecosystem service. This is in line with the definition of an outcome, namely *any event that occurs due to an alien species or its regulating management, in the course of the species development to become an invader.*

How much in detail an impact on a certain ecosystem service can be described depends on available knowledge. Different situations of uncertainty have been distinguished before. They are amenable to a systematic analysis along the set of ecosystem services. This means, next to certain impacts, situations of risk or of uncertainty i.n.s. regarding impacts can be identified.
Situations of *risk* are similar to a situation of certainty because the impact on the ecosystem service at hand can be clearly indicated. What distinguishes both situations is the target region for which a management decision has to be taken. Risk exists regarding an invasive species that is expected to become invasive in a target region but has not appeared yet, whereas in the

situation of certainty the species has already become invasive there. Available information on former impacts will be transferred to the target region. This is possible as ecological and anthropogenic factors in the target region are similar to the ecosystem which has been already invaded. If this is the case, impacts on the target ecosystem are very likely.

Uncertainty i.n.s. means that less information is available on the impacts of biological invasions than in the situation of risk. Impacts are no longer foreseeable due to the missing opportunity to find similar ecosystem properties of which knowledge can be transferred to the target region. As they cannot be derived from impacts that occurred somewhere else before, they can only be assumed. *Uncertainty* i.n.s. occurs due to two reasons:

i. If species which are alien to a target region have not yet turned invasive. Uncertainty i.n.s. exists for alien species' impacts *ex-ante* an invasion, i.e. for the phases of introduction, establishment and naturalization.

ii. If cascading effects of an impact occur ex-post an invasion. This is the case for impacts triggered by an initial impact. In such a situation all trigger effects of the initial impact can only be vaguely anticipated, e.g. if the known impact changes the nutrient content in the soil, it can only be surmised whether this impact will also affect the natural rate of soil renewal or the availability of freshwater.

Ignorance in the case of biological invasions implies that an impact of an invasive species is outside the set of expected outcomes of its invasion. While working with the concept of ecosystem services all possible outcomes are defined via the list of ecosystem services. An outcome is accordingly a change of any of the 26 services. An unexpected outcome in terms of ignorance means that an impact of an invasive species happens, which is not addressed by one of the 26 ecosystem services and which is outside this fix set of goods and services. As a consequence only the categories of risk and uncertainty i.n.s. can be considered by the concept. Thus ignorance cannot be revealed with the concept of ecosystem services although predominant in several invasion phases. Ignorance exists regarding biological invasions due to randomness in the invasion process. We have to acknowledge that there are always situations for which no knowledge at all is yet available. This is a remnant of ignorance that is irreducible (Amendola 2002). We can never be sure whether or not one day there will be an outcome that will be outside the set of expected or anticipated outcomes, due to random changes or new evolutions in the ecological or social system (Faber *et al.* 1992). Since the situation of ignorance is inherent to every development in the future and cannot be captured with the concept of ecosystem services, it will be outside further investigation. Approaches

for reducing communal and personal ignorance have been mentioned in section 3.1.
Table 12 summarises the different situations according to available knowledge about the impacts of biological invasions when working with the set of ecosystem services.

Table 12: Identified situations of knowledge for impacts of invasive species by using ecosystem services (own source)

Situation of knowledge	Impact occurring yet	Occurrence in target region
Certainty	Yes	Yes
Risk	Yes	No
Uncertainty i.n.s. **a) for alien species** **b) for cascading effects**	No (just alien) No	Yes Yes

The different situations can be briefly described as:

- If a species is already invasive in a target region and impacts are recorded, as a result we are in a situation of certainty.
- A situation of risk exists, if impacts have been recorded, but not for the region under consideration.
- A situation of uncertainty i.n.s. exists for species that are alien but not invasive in a target region and in the case when a species is occurring in a target region but possible cascading effects of impacts are unknown.

6.3.3 Addressing stakeholders for affected ecosystem services

Next to the contributions to approaching the *scope*- and the *uncertainty*-problem, there is another property of the concept that is of great meaning for decision-making. In the following it will be elaborated to what extent the concept of ecosystem services can be used for the identification of stakeholders which are involved in the process of biological invasions. Humans substantially verify the phenomenon of biological invasions, either by introducing invasive species or by managing them. Unconscious behaviour in habits and conscious decisions about the appropriate management of invasive species decisively determine the rate and speed of biological invasions. Thus, analysing how people are involved in biological invasions is important for their appropriate management and the design of effective strategies. To depict stakeholders these in biological invasions is useful, as those that drive the process are seldom confronted with the damaging impacts of invasive species. Using the concept of

ecosystem services helps revealing drivers and recipients of damages of invasive species.
To show how the concept enables the identification of relevant stakeholders, a three step stakeholder analysis is conducted. In order to bring it in the context of ecosystem services stakeholders are identified and subsequently assigned to the various ecosystem services. Regarding biological invasions, respective stakeholders can be addressed by attributing the different impacted ecosystem services to the different stakeholders.

6.3.3.1 Stakeholders involved in biological invasions

The following analysis of different perspectives of actors on the invasion problem has an affinity to approaches in New Political Economics (Gawel 1995: pp. 55; Hansjürgens 2000: pp. 152). Stakeholder analyses have been also applied in business administration (Brade 2005: pp. 47).
A stakeholder in the following context is understood as a single person or an organised group of people which pursues a certain interest or goal. It represents an analytical unit (Freese and Steinmann 2005: 34)[52]. The different interests of stakeholders can be politically motivated, e.g. if agricultural actors persuade and foster agricultural political interest in the public. Thus stakeholders are to a large degree overlapping with political sectors from section 5.4.2.2.
A stakeholder analysis generally intends to identify relevant groups of people or single persons that influence the decision-making process regarding the appropriate management of an environmental problem. In this thesis it is the problem of biological invasions. The identification and involvement of stakeholders has been noted to be crucial for the development of assessment results that foster the solution of environmental problems (Turner *et al.* 2000; Howarth and Farber 2002; Wilson and Howarth 2002; Townsend 2003). In order to structure such an analysis in a systematic way, stakeholders involved in the problem of biological invasions are determined and analysed in the following steps:

1. Listing relevant stakeholders
2. Characterising stakeholders
3. Determining their relevance for the decision-making process.

The last step, of course, can only be implemented in a concrete decision context and for a certain invasive species. The exercise of all three steps for a concrete example is given in

[52] Synonym for a group of people which pursue an interest would be interest group, see for instance Brade (2005: 50).

chapter 7. Step one and two, however, are exercised in the following.

Step one: Listing relevant stakeholders

In the first step different stakeholders have to be identified. Stakeholders who play a role in every invasion are the following: agriculture, fishery, forestry, health care, nature conservation, municipalities, and tourism. They are noted in the first column of Table 13.

Table 13: Stakeholders involved in biological invasion (own source)

Stakeholder	Impact recipient	Type of impact	Impact driver	Driver function
Agriculture	X	• Production loss (crops, fibres, vegetables, wine) in food production • Loss of genetic diversity	X	• Increase in species diversity • Utilisation of alien species • Unintentional introduction through seedlings, propagules, packing material • Biological control
	X	• Loss of yield (life stock, meat, wool) in livestock breeding/fishery	X	• Increase in species diversity • Trade with alien species
	X	• Production loss in fodder production • Loss of genetic diversity	X	• Increase in species diversity • Utilisation of exotic species • Biological control
Horticulture	X	• Decrease in native species diversity • Hybridisation of native and alien species	X	• Breeding of species => increase in species diversity, but also decrease of native species diversity • Trade with exotic species • Counter measures for invasive species
Forestry	X	• Loss in yield of timber production • Loss of genetic diversity	X	• Increase in species diversity • Utilisation of alien species • Counter measures for invasive species
Municipality	X	• Damages in public "green" places (parks, roads, gardens) • Overgrowth of public facilities • Impeding of public transport	X	• Planting exotic species • Public transport and trade as vector for propagules • Counter measure against invasive species
Tourism	X	• Decrease and increase of touristic attractions (cultural landscapes, recreation areas parks, landscape gardens,) • Acceptance of changing landscapes • Historic and cultural meaning of exotic species	X	• Accidental and deliberate introduction of invasive species through travel and transport
Health care	X	• Increase in allergic pollen, phototoxic substances of plants, pathogens • Medical treatment of health problems due to invasive		

Stakeholder	Impact recipient	Type of impact	Impact driver	Driver function
		species		
Conservation	X	• species loss • Loss of genetic diversity • Decreasing number of native species • Management of nature reserves		• Counter measures against invasive species, mostly control and eradication
Private households	X	• Diseases • Damages to buildings, gardens • Avoidance and restricted access to infested areas	X	• Dispersal of invasive species propagules • Planting and release of alien species • Control of invasive species
Research	X	• Invasive species opening a broad field for scientific exercise	X	• Provision of knowledge for an improved management
Educational institutions			X	• Provision of information about invasive species
Media			X	• Provision of information to the public about problem with invasive species

Stakeholders from agriculture, horticulture and forestry are subsumed under the term land-use if all of them are being referred to. Next to those stakeholders who have the background of a political sector, the list has to be complemented by private households, research, educational institutions and media.

Private households are highly involved in biological invasions, both in terms of driving the problem but also of being affected by them. How they contribute to biological invasions will be explained in the next step.

Research perceives biological invasions as an increase of potential object of investigation. Results can be used for an improved management of invasive species.

Educational institutions and media are two stakeholders that drive the process of invasive species negatively (see below), as they supply information about problems with invasive species.

Listed stakeholders are, of course, preselected and further stakeholders might be added for a certain invasive species. This will become clear in chapter 7. Mentioned stakeholders are distinguished in primary stakeholders, i.e. they are *directly* involved in the invasion problem, and secondary stakeholders who are *indirectly* involved in the invasion process. An example for a secondary stakeholder is media. Owing to the fact that the present stakeholder analysis intends to outline its usefulness for economic assessments, a detailed analysis comprising all possible stakeholders has been neglected so far.

Step 2: Characterising stakeholders

Table 13 not only lists different stakeholders of impacts of biological invasions. It also reveals the general role stakeholders can have in terms of describing their activities and how they are affected by impacts of biological invasions. An essential distinction has been drawn between those stakeholders that receive impacts of biological invasions and those that drive the process. Stakeholders that receive impacts are called "recipients" and those that drive them "drivers". Drivers can have a positive and a negative promoting function. Promoting drivers have the potential to foster the dispersal and introduction of invasive species. Negative drivers have the capacity to reduce impacts of biological invasions and to employ counter strategies for biological invasions. One stakeholder, of course, can be both, driver and recipient at the same time. Agriculture is an example for a stakeholder with a dual role: it is affected in terms of receiving damages in production yield, but also drives invasions as it provides pathways for the introduction and dispersal of alien or invasive species. As a negative driver function, agricultural stakeholders can employ biological control, eradication strategies and prevention if they use, for instance, non-foreign seedlings for cultivation.

For the identification of relevant stakeholders, one has to consider the whole invasion process, especially because drivers are relevant in the beginning of an invasion, during the introduction phase. Affected stakeholders only emerge during the invasion phase when impacts of biological invasions become damaging and apparent to the public (see Section 2.2). In the following, the focus is on the driver function of stakeholders, as the majority of damaging impacts has already been explained in Section 2.3.

Actors from *land-use*, i.e. from agriculture, forestry, horticulture as well as hunting and fishing, are on the one hand damaged through invasive species, and on the other hand main drivers of the process. Exotic species and products, e.g. seedlings or propagules, hybrids of feral animals as well as fodder plants, are examples where alien species are introduced for their direct utilization and their beneficial character to humans. Such goods are introduced intentionally and offer a pool for secondary release (see section 2.2.2.1). Horticulture, for instance, represents an area where the cultivation and trade of alien plants has been commonly practiced for hundreds of years. Due to this, public as well as private gardens are one of the main pools for alien species' secondary release today. Next to the intentional introduction of species there are also unintentional pathways. Packing material, timber, soil or other properties without direct use carry a variety of alien species or its propagules. Places of ship loading or rail stations are spots where invasive species have been increasingly recognized.

Biological control is an example how agricultural stakeholders can negatively influence, i.e. restrict a biological invasion.

Municipalities also have both roles: being drivers as well as being recipients. The utilization and appreciation of alien plants for stabilising rail roads, dykes and highways, promote the introduction of alien species, e.g. when instead of locally adapted plant species alien species are preferred due to lower prices. Planting alien species on public places supports the dispersion of seeds and gametes of invasive plants, as public transport is the most important vector (Ortner 2005). As stated above, public places such as parks, gardens, or arboreta and greenhouses contain a considerable amount of alien species that can in time become invasive if they manage to escape into semi-natural or cultural landscapes and ecosystems.

For stakeholders from the *tourism* sector, invasive species' impacts can have both positive and negative aspects. On the one hand, areas with high abundance of invasive species can decrease the attractiveness of a tourist area, such as the fynbos region in South Africa. On the other hand, biological invasions, such as in the case of *Rhododendron ponticum* in Wales, can increase the touristic value of an area and characterize a whole landscape. These stakeholders are relevant as drivers as they foster an unintentional introduction and dispersion of alien invasive species by vessels, aircrafts or other means of conveyance for tourists.

For stakeholders from , invasive species, such as *Ambrosia artemisiifolia* and *Heracleum mantegazzianum* are of great concern. These stakeholders have to deal with invasive species if they contain toxic substances or allergic pollen that cause severe health problems. Invasive species impose increasing costs for this political area in terms of the increase of medical treatment costs.

Conservationists are main recipients of invasive species problems, as they concern themselves with ecosystem changes such as loss in species or genetic diversity. The responsibilities of maintaining nature reserves and managing invasive species appropriately are often mainly attributed to these stakeholders. It is important to note that neither the health sector nor the conservation sector is a special driver. Both are primarily recipients of impacts, although in the case of conservation stakeholders they are not directly affected in terms of loss in yield or production. However, owing to their great interest in nature conservation they are important stakeholders that foster the management of biological invasions.

Private households are also important stakeholders due to their dual role of driving and perceiving damages of biological invasions. Intentional behaviour, such as the purchase of alien plants for private gardens, but also the release of alien species, e.g. when aquariums are emptied into natural water bodies, can initiate a biological invasion. Unintentional behaviour can also foster invasive species, for instance if private households use goods that contain invasive species propagules, such as *Ambrosia artemisiifolia*-seeds in bird fodder, or parts of *Fallopia*-rhizomes in garden soil which has been brought to other places.

6.3.3.2 Attribution of stakeholders to ecosystem services

As explained above, identified stakeholders can be assigned to the various ecosystem services. Table 14 shows stakeholders assigned to each ecosystem service.

Table 14: Impacts of invasive species attributed to sectors (own source)

Ecosystem services	Impact on ecosystem process or component	Relevant stakeholder	Impact recipient	Impact driver
Sustaining services				
Soil formation and retention	• Increase/decrease in erosion, alternation in soil fertility, salinisation			
Nutrient cycling	• Shifts in nutrient dynamics, uptake in the soil			
Primary production and photosynthesis	• Shifts in primary production			
Provisioning of habitats	• Shift in biodiversity	• Conservation	x	
Water cycling	• (see water regulation)			
Provisioning Services				
Food and fibres	• Decrease/increase in production	• Agriculture	x	x
Fuel	• Increase/decrease in energy material	• Agriculture	x	x
Genetic resources	• Increase in hybridisation, introgression • Loss of genetic diversity	• Land use • Conservation	x x	x
Biochemicals, natural medicine, pharmaceuticals	• Decrease in genetic pools due to biodiversity loss	• Industry • Private households	x x	
Ornamentals	• Production loss	• Agriculture • Horticulture	x x	x x
Freshwater	• Alternation in (fresh) water availability, quality	• Agriculture • Municipality • Private households	x x x	
Regulating services				
Climate regulation	• Micro-climatic alternation due to changing vegetation structure	• Land use • Municipality • Private households	 x x	x
Erosion control	• Loss of structures of ecosystems	• Municipality • Land use	x x	

Ecosystem services	Impact on ecosystem process or component	Relevant stakeholder	Impact recipient	Impact driver
Water regulation	• Alternation in water regulating systems (embankment, retention sites)	• Municipality • Private households • Land use	x x x	x x
Water purification and waste treatment	• Alternation in nutrient and compound, biomass removal	• Land use	x	
Pollination	• Pollinators as vectors, changes in pollinator functions	• Municipality • Private households	x x	
Biological control	• Increase/decrease in invasive species population, IS as control agents	• Land use • Municipality	x x	
Regulation of human diseases	• Increase of human diseases mediated by invasive species or due to the invasive species itself	• Health • Private households • Tourism	x x	 x
Storm and flood protection	• Change of riparian forests as flooding buffer, retention site	• Municipality • Private households	x x	
Cultural services				
Cultural diversity	• Differences in ecosystem features such as species composition, decrease due to homogenisation of species by invasive species	• Municipality • Private households • Tourism	x x	x
Aesthetical values	• Increase/decrease in aesthetic inspiration	• Tourism • Municipality • Private households	x x x	
Inspiration	Increase/decrease in aesthetic inspiration	• Municipality • Private households	x x	
Spiritual and religious values	*	*		
Scientific and educational values	• Change in observable natural phenomena	• Research • Educational institutions	x x	
Recreation and ecotourism	• Shifts in recreational value of landscapes	• Tourism • Private households	x x	x
Cultural heritage and historical values	• Alternation in parks, home gardens, "cultural landscapes"	• Land use • Municipality • Private households	 x x	x

*due to lacking examples there have no stakeholders assigned

As one can see, usually more than one stakeholder influences the change of the recipient ecosystem service. According to the distinction from above, a stakeholder can be both recipient of the change of an ecosystem service due to invasive species, i.e. the stakeholder is confronted with the changes of an ecosystem service due to an impact of an invasive species, and driver of the change, i.e. the stakeholder fosters the introduction and dispersion of an invasive species. The changes of an ecosystem service are mentioned in terms of an impact of an invasive species or on an ecosystem process or component. The relevant stakeholders are now sorted for the recipient political sectors. Table 15 also underlines the fact that often recipients of impacts and drivers are not the same. An example is the ecosystem service of cultural heritage and historical values. Land use stakeholders introduced the alien plant *Solidago canadensis* for commercial utilisation and for reasons of providing a rich source of pollen for honey bees (Kowarik 2003: 147). The plant, however, developed into an invader. Now it has been growing almost everywhere, such as home gardens or urban fallows. Recipients of the invasion in this case are municipalities and private households.

From the above it can be conclude that biological invasions' impacts often affect several stakeholders. By using the ecosystem services concept involved stakeholders can be identified. This information can be used for the design of management strategies. As the stakeholder analysis in its completeness makes only sense in a concrete example, the full meaning of the analysis becomes obvious in the case study of *Ambrosia artemisiifolia* in Chapter 7.

6.4 Conclusions: the suitability of ecosystem services for economic assessments of biological invasions

In this chapter a variety of characteristics of the concept of ecosystem services have been explained. In the following those properties are summarised that are relevant in the context of biological invasions and their suitability to provide improvements for economic assessments. Major advantages have been referred to as contributions to (i) the *scope*-problem, and (ii) the *uncertainty*-problem, and (iii) the possibility of a stakeholder analysis.

1. Contributions to the *scope*-problem

Section 6.2 shows biological invasions can have, and in fact do have, impacts on all ecosystem services of the world. Although the existing examples refer to different ecosystems in the world, the visualisation of impacts by using ecosystem services underpins the magnitude of effects and their far reaching character. The list of 26 ecosystem services is encompassing, and by using the list the number of impacts that have to be taken into account during an assessment is determined. Thus, the concept of ecosystem services can serve as analysis structure for a physical assessment of biological invasions. Accordingly it offers a kind of checklist for the impacts that have to be recorded during that stage of economic assessments. Additional advantage of such a checklist is the standardisation of the assessment procedure. All physical assessments now refer to the same base, namely the entire list of ecosystem services. If this would become common practice, comparability of results would be possible. With such a framework at least a well-structured analysis of impacts of biological invasions is possible.

The information about the impacts for each ecosystem service shows that invasions are not always of damaging nature. There are also examples where invasive species have a positive effect, as is the case for many cultural services. However, it is evident that the majority of impacts are damaging and decrease human well-being. The concept allows consideration of the changes of ecosystem services without an implicit positive or negative connotation.
By providing an analysis structure the concept has a strong complementing character for the TEV. With the help of ecosystem services also impacts are considered according to their change due to an invasion which is outside the scope of the TEV. Such impacts, however, can only be described and not quantified which is necessary for a monetarisation in the economic valuation step, for which the TEV is used for.

2. Contributions to the *uncertainty*-problem

If the concept helps to structure available knowledge, also incomplete knowledge about impacts of a species can be disclosed. This means that within the analysis structure of ecosystem services, not only those impacts can be recorded for which complete knowledge is available and that are of major interest to relevant stakeholders, but that also those ecosystem services have to be taken into account for which information is not complete. Regarding uncertainty, different situations can be divided according to the degree of information. This means that in a physical assessment of biological invasions' impacts, situations of certainty, risk and uncertainty i.n.s. concerning the occurrence of an impact can be considered[53]. An encompassing economic assessment demands the analysis of all services, no matter if impacts are expected or can be clearly identified. This particular characteristic of a physical assessment of biological invasions along ecosystem services is a strong improvement compared to standard economic assessment bases. To make uncertainty about impacts already

[53] Why ignorance can only be included in terms of reducing personal ignorance has been explained in section 6.3.2.

explicit at the physical assessment stage is very helpful, as it forms an integral part of any economic analysis from the very beginning. Usually considerations of uncertainty are addressed after the evaluation process (e.g. via sensitivity analyses in cost-benefit analysis, see for instance Pearce *et al.* (2006: 60)). The ecosystem services concept allows doing so before the evaluation itself. This implies that uncertainty can be taken into account in the same way as available information can be. The decision maker then knows the base on which the decision is made. This will have implications on the management, as a decision which relies on a base with high uncertainty might be more cautious than one that has considered mostly certain facts.

Identifying ecosystem services for which we can vaguely expect some change due to an invasion offers the opportunity to invest further research efforts in understanding whether the species will be driven by ecological as well as anthropogenic factors to become invasive at a certain point in time.

3. Stakeholder analysis

By attributing stakeholders to the different ecosystem services, impacts can be also assigned to them. Such an analysis shows that various stakeholders are involved in the problem with invasive species, all in a different manner. This means they can be affected by impacts, i.e. are recipients, or they can be drivers of the issue at hand; they also can be directly or indirectly involved. Usually those that drive the process are not the ones that perceive the damages. As each stakeholder has different opportunities and an own suitable set of strategies how to approach biological invasions, the various management options can be depicted. For a successful management consorted action is recommended. The fact that the concept refrains from the typical point of view that invasive species are linked with the loss of biodiversity is also helpful. This perspective used to address only the environmental political arena for which it is object of conservation strategies. With the help of the mentioned stakeholder analysis, however, it became clear that many more stakeholders are concerned with biological invasions than merely conservationists.

Concerning the subsequent economic evaluation it has to be noted, that recording the different impacts of invasive species on ecosystem services leaves open the question of whether this will be expressed in monetary units or not. The concept leaves open how to evaluate those impacts. While the TEV is fairly narrow and each value has to be transferred and represented in a monetary unit, there are possibilities to depict changes of an ecosystem service also in other terms. The detailed information that is available about all the different kinds of ecosystem services come from the different dimensions of ecosystem services. This implies that the numerous services can have numerous units of measurement. For instance for the majority of cultural services it will become difficult to transfer an ecosystems value into monetary terms. People will express their appreciation about the cultural, aesthetical and educational value in a descriptive and qualitative manner. Other ecosystem services can be easily expressed in monetary terms, e.g. ecosystem goods that are traded on markets, such as timber. Implications of the mentioned findings on an economic evaluation will be discussed in the last chapter of this thesis.

6 Enhancing economic assessments with the concept of ecosystem services

7 Biological invasions in Germany and the case study of *Ambrosia artemisiifolia*

At the end of the previous chapter the contribution of the concept of ecosystem services to the *scope*- and the *uncertainty*-problem of economic assessments of biological invasions has been explained. This chapter now applies the concept (i) to illustrate problems with invasive species in Germany in general, and (ii) to show in a case study of the invasive short ragweed the contributions to economic assessments. Until now contributions to enhance economic assessments of biological invasions have only been theoretical. With the case study, the practical advantages of the concept of ecosystem services are now tested and its application reveals that not only uncertainty can be taken into account already at the physical assessment stage, but also can relevant actors be identified at this stage. In the end, conclusions can be drawn for an appropriate management of *Ambrosia artemisiifolia.*

The chapter is structured as follows: in section 7.1 the situation of invasive species in Germany is presented. For this, impacts of different invasive species are explained on the list of ecosystem services. In section 7.2 the case study of *Ambrosia* illustrates what the impacts on ecosystem services of that species are occurring in Germany. Furthermore available management options are explained and to what extent they are successful. In this context, uncertainty regarding potential impacts of the plant short ragweed as well as concerning the success of management strategies is examined. A stakeholder analysis identifies relevant drivers and recipients of the problem and how they can contribute to a management of the species. In the last part of section 7.3 conclusions are drawn about how the contributions of the concept of ecosystem services work in practice to enhance economic assessments of the invasive plant.

7.1 Impacts of invasive species on ecosystem services in Germany

7.1.1 The special situation due to Germanys evolutionary history

The European situation concerning biological invasions is different compared to that in many other regions of the world. Owing to the middle European evolution a specific fauna and flora developed in Germany. After the last ice age, species moved through natural corridors and are today highly adapted to current environmental conditions. They have been witnessing wide-ranging and frequently changing anthropogenic land use systems since the Neolithic, i.e. since

6500 years ago (Kowarik, 2003: 16). This evolutionary path of European species stands in contrast to that of other species on other continents, like America or the southern Hemisphere, where human activities did not influence evolutionary processes for thousands of years, and where constant conditions have existed for a much longer time period. Species in middle Europe have had time to properly adapt to prevailing conditions (Kowarik, 2003: 22). This evolutionary difference explains why the European definition for 'introduced' and therefore 'new' species includes a first appearance after 1492 (see section 2.1.)

Today, the middle European climate provides conditions that exterminate several frost-sensitive species. As a consequence, a high number of subtropic and tropic plants are not able to survive and to establish vital populations. Also characteristic for the situation in Germany is the high number of intentionally introduced species, as the utilisation of non-native species in horticulture and many forms of land use systems has been common for hundreds of years. Such species sometimes even have a high cultural and historic meaning. Several parks and landscape gardens have been cultivating a wide range of alien shrubs, like *Rhododendron*-species, coniferous trees and flowers for ornamental reasons since the 15th century. Some of these species escaped into the wild and are today so-called "home-made" invasions (Kowarik, 2003a). Remarkably, no extinction of a native species due to an invasive species has yet occurred (Kowarik 2003: 288). This fact has been often explained with the adaptation of species over hundreds of years, which gives native species a distinct advantage over invasive species. Lacking extinction, however, does not imply that invasive species are not problematic in Germany. Profound ecosystem changes due to invasive species are obvious and have been increasing. Additionally, there is a high number of non-native species that might wait for the day to come to surpass the time-lag and to disperse invasively. Examples from Germany, for which scientists worry about future species development and the ongoing dispersion into undisturbed areas are *Douglasie* (originally a forestall tree) and *Elodea* (originally from botanical aquaculture) (Kowarik 2003: 185).

7.1.2 Examples of invasive species changing ecosystem services in Germany

Changes of *sustaining services* have been mentioned to be difficult to record because these services are of long-term relevance, act on the global level, and are often changed by a combination of several factors that make the identification of the changing factor "invasive species" difficult. Nevertheless, there are changes of these services. Impacts occur on the

local, regional and even national level. Especially regarding the change of the ecosystem service "provisioning of habitats", numerous examples of invasive flora and fauna are known that have changed habitat conditions for native species, see for instance Kowarik (2003: 37). The invasive plant *Impatiens glandulifera*, for instance, provides a new trophical source for insects and therefore influences horizontal food chain interactions, which in turn are important to the constitution of an ecosystem's species composition (see section 2.3.1.4).
Also *provisioning services* are affected, e.g., by species initially introduced for reasons of utilisation. Tree-species, such as *Prunus serotina* and *Quercus rubra*, both originally from North America, were once intentionally introduced in forestry, and are now invasive plants (Reinhardt *et al.* 2003: pp. 29). Their high growth rate has been appreciated in timber production. Soil improvement and reinforcement are characteristics that make them preferable to other, native species (ibid). *Pseudotsuga menzesii, Acer negundo* and *Castanea sativa* are other examples of invasive species formerly used in forestry. Today their dispersal into nature reserves is uncontrolled. Apart from conservation aspects, *Prunus serotina* causes a lot of damage in forestry. It can build very dense shrubby stands on sandy sites which restrict optimal yields from native timber species (Kowarik 2003: 175). Similarly to *Acacia*-species in South Africa, the *Miscanthus*-species provide new sources of material for energy use. The species´ high biomass production offers an alternative to native species in providing energy. Large-scale plantings in some regions of Germany are initiated for commercial production (Kowarik 2003: 142). Although an invasive dispersal tendency has not been discovered yet, this plant may be suspected to possibly become invasive one day (Barney and Ditomaso 2008). Genetic resources are linked with genetic diversity as one level of biodiversity (see section 6.1.4). Genetic resources, relevant for the utilisation of old crop and feral species, decrease due to the tendency to use hybrids of exotic species in agriculture, horticulture and forestry. Municipalities planting highways often do not use local or native species, even though they are supposed to have economic advantages because such plants and seedlings are properly adapted to local climatic conditions and thus, the mortality rate is lower than of foreign seeds. Several invasive species are used as ornamentals today. Examples of such a positive impact on this provisioning service are Policemen's helmet (*Impatiens glandulifera*) or the Giant hogweed (*Heracleum mantegazzianum*).

Regulating services are highly influenced by species, such as *Fallopia*-species, *Robinia speudoacacia*, or *Impatiens*-species. They all affect natural erosion control; either by building thick stands that hinder erosion, and thus have a positive effect on this ecosystem service, but may also promote erosion by their unfavourable rhizome system (Kowarik 2003: 217, 159, 169). *Fallopia*-species for instance, influence local water regimes. On the one hand, their rhizomes have a positive, stabilising effect on riverbanks; on the other hand these plants limit the runoff after flood events. Another essential ecosystem service is biological control. It is related to predator-prey relations in natural and semi-natural systems. A variety of exotic and genetically modified species are preferred to native species in agriculture. This kind of impact refers to the genetic diversity aspect of species. Indirect effects are noticed, e.g., concerning *Chrysoperla carnea* (Schrader 2003), which is a typical native biological control agent in agriculture. Genetically modified maize releases toxins that increase the mortality rate of *Chrysoperla carnea*, which means such an effect damages agricultural productivity through lacking native control agents.

Aesthetical values are also positively influenced by biological invasions if species are used as ornamentals. In this context they have a beneficial impact on society. Such species can also have historic and cultural meaning. They give evidence about former cultural styles in horticulture and gardening. In contrast to that, there are also invasive species that threaten middle European landscapes that were shaped by traditional land use systems. The occurrence of invasive species deteriorates the original meaning of such a landscape. An example is the widely appreciated natural monument *Kalksinterquellen* near Asse (Germany), that has drastically changed shape as due to thick stands of *Heracleum mantegazzianum* (Kowarik 2003: 213). Next to the historic and cultural meaning of invasive species, they are also of interest to the scientific community. Several research projects investigate ecological traits, disturbance effects and their meaning on other, native certain species. Invasion ecology is a great field of research in Germany.

Table 15 summarises impacts of invasive species and their impacts on ecosystem services in Germany. It can be concluded that biological invasions did not yet lead to extinctions of a native species. This is supported by the fact that a wide range of alien species is naturally excluded by its deficient frost tolerance.

Such a natural exclusion, however, does not imply that biological invasions are not of relevance in Germany. As described above, for almost all invasive species have impacts that cause changes in almost every ecosystem service. Not only can the different ecosystem services illustrate the magnitude of impacts of different species in Germany, they can also helps to systematise the impacts for that one species causes.

Table 15: Impacts of invasive species on ecosystem services for Germany (derived from Kowarik 2003[54])

Ecosystem services	**Example of impact on ecosystem process or component**
Sustaining services[1]	
Soil formation and retention	*Spartina anglica* increases sediment accumulation at the coastline of the Wadden Sea (p. 235)
Nutrient cycling	*Robinia speudoacacia* accumulates nitrogen in nutrient poor soils (p. 157)
Primary production and photosynthesis	*Senecio Canadensis* with competition advantage compared to other perennials due to a longer time period of photosynthesis (p. 148)
Provisioning of habitats	*Impatiens glandulifera* providing additional trophic sources for insects (p. 204)
Water cycling	*Mimosa nigra* changing local water regimes (p. 34)
Provisioning Services	
Food and fibers	*Prunus serotina* causing less timber production in the undergrowth (p. 175)
Fuel	*Miscanthus*-species usable as renewable energy resource (p. 132)
Genetic resources	*Mahonia aquifolium* as a hybrid of North America *Mahonia aquifolium* and *Mahonia repens*[1]
Biochemicals, natural medicine and pharmaceuticals	*
Ornamentals	*Fallopia*-species used as ornamentals in parks and landscape gardens (p. 215)
Fresh water	*
Regulating services	
Climate regulation	*Prunus serotina* changing radiation in forests (p. 175)
Erosion control	*Fallopia*-species have a stabilising effect on riverbanks (p. 217)
Water regulation	*Fallopia*-species limiting runoff after flood events (p. 217)
Water purification and waste treatment	*
Pollination	*Mahonia aquifolium* supporting the evolution of new pollinators [1]
Biological control	*Chrysoperla carnea* in genetically modified maize has an indirect effect to through toxins (p. 294)
Regulation of human diseases	*Ambrosia artemisiifolia* with highly allergic pollens[2]
Storm and flood protection	*Lupinus polyphyllus* used for erosion prevention on dams (p. 154)
Cultural services	
Cultural diversity	*Rhododendron* increasing the number of alien trees in horticulture (p. 300)
Aesthetical values	*Impatiens glandulifera* provides aesthetical value by colourful blossom (p. 104)
Inspiration	See *aesthetical values*
Spiritual and religious values	*
Scientific and educational values	Invasive species in general as the largest "outdoor" experiment of evolution
Recreation and ecotourism	*Heracleum mantegazzianum* infested areas to be avoided by children (p. 214)
Cultural heritage and historical values	*Solidago*-species as succession barrier on nutrient poor and species rich pastures at the *Schwäbische Alb*(p. 149)

[1] Examples of impacts on sustaining services refer to a regional or local occurrence
* For this ecosystem service no example has been found
[1] Auge and Brandl (1997); [2](Reinhardt *et al.* 2003)

In order to show how the concept of ecosystem services supports decision-making, a specific invasive species will be chosen as a case study, which will reveal further advantages of using

[54] Other sources of literature are marked by additional footmarks.

the concept of ecosystem services.

7.2 The case of Ambrosia artemisiifolia in Germany

In section 6.3, the conceptual advantages of using ecosystem services in economic assessments of biological invasions have been theoretically described. Major focus was on the first step, the physical assessment of invasive species. Therefore, the following sections intend to highlight these conceptual contributions in practice in the case of *Ambrosia artemisiifolia*, in the following referred to as simply *Ambrosia*. This invasive plant is an especially interesting case because it is of its ongoing march into Germany and the current debate by a lot of stakeholders on how to approach the starting invasion.

For the physical assessment of *Ambrosia*, not only scientific sources were accessed. The scientific sources are limited due to the recent invasion in Germany. In order to gain as much knowledge as possible and to enable an all-embracing physical assessment, also electronic and personal information, as well as grey literature has been used. Personal information by experts has been gained by attending several workshops for a nationwide initiative for establishing an information network about *Ambrosia*[55], and through personal communication with people that deal with the *Ambrosia*-problem in their work.

The case study is structured as follows: in section 7.2.1 the plant and its situation in Germany are characterised; in section 7.2.2 impacts of the species are considered with the concept of ecosystem services; in section 7.2.3 current management options are explained; in section 7.2.4 uncertainty regarding impacts of the species and regarding the success of current management strategies is discussed; in section 7.2.5 a stakeholder analysis reveals relevant stakeholders involved in problems with the plant, and finally, in section 7.2.6 advantages of using the concept of ecosystem services for economic assessments of the species are summarised.

7.2.1 Characteristics of the *Ambrosia*-problem in Germany

7.2.1.1 *Ambrosia* in Germany

The common ragweed, (*Ambrosia artemisiifolia*), has been found in Germany in 1863 for the very first time (Schrader 2003). It is a well known invasive plant species in countries in Eastern and Southern Europe. Although already occurring in adjacent countries, such as

[55] The workshops were in Braunschweig in November 2005 and December 2006 at the Federal Research Centre for Cultivated Plants (formerly Federal Biological research Centre for Agriculture and Forestry; *German*: BBA.

Hungary, Austria, Czech Republic, Slovakia, Switzerland, Italy and France, the plant has not played a vital role in Germany until few years ago, neither regarding conservation matters, nor as part of the general public debate about invasive species (Dechamp and Penel 2002; Schrader 2003). Within the last years, however, there has been growing concern about the species, because of its increasing presence in Germany. Its occurrence today is still patchy in Germany, but since the 1950's *Ambrosia's* spread has been increasingly recognised. Only a few sites with higher dense of the plant have been reported for the last recent years. Pollen traps in the Frankfurt region, the whole Rhine-Main area and in Stuttgart have recorded a high pollen count of *Ambrosia*. This fact has been explained by the warmer climate of these regions (Schrader 2003). Since temperature is assumed to be a key factor in the plants dispersion (Rabitsch and Essl 2006), a triggered invasion of *Ambrosia* coming from the southern and warmer countries of Europe is increasingly expected (Taramarcaz *et al.* 2005). Rising temperatures due to global climate change are only one aspect of the ongoing invasion of *Ambrosia* in Europe and especially in Germany. The increasing availability of nitrogen sources (e.g. through public transport and the use of industrial fertiliser) is another consequence of global change, and has been acknowledged to lead to a dramatic increase in pollen production of the plant (Townsend 2003). This fact provides additional support for expectations of a rising frequency of *Ambrosia*-occurrence in Germany.

7.2.1.2 Ecological characteristics of the invasive plant

Ambrosia is native to North, Central and South America. The plant taxonomy assigns it to the family of *Asteraceae* and to the genus *Ambrosia*, which covers 40 species. Three *Ambrosia*-species are known for Europe (*Ambrosia artemisiifolia*, *Ambrosia coropifolia* and *Ambrosia trifida*), but only *Ambrosia artemisiifolia* will be the object of the following investigation. The common ragweed is a summer annual plant and grows mainly on idle sites, such as way sides or rail roads, dump sites or fallows, which are areas typically anthropogenic disturbed. Uncovered soil provides conditions that allow the species to progress rapidly from germination to blossom (Brandes and Nitzsche 2006). Such conditions are also present in arable fields, where *Ambrosia* has been mainly found in fields with peas, soya bean and sunflowers, as well as in tomato and maize stands (Brandes 2005). Suitable conditions for the plant are characterised by high water availability, warm temperature and little competition for light. The plant depends on sufficient water availability as its rhizomes are small. This explains why *Ambrosia* has been mostly found in regions with summer rain and is absent from continental areas without high humidity in summer time (Nitzsche 2005).

Due to a short juvenile phase and a long phase of blossom, the plant is able to produce seeds within a few weeks from germination. The amount of seeds can be enormous. The time of blossoming is normally between August and October (Reinhardt *et al.* 2003: 17). The average number of seeds per plant is estimated at about 3000 to 4000 seeds, but can reach up to 60,000 seeds (Zwander 2001; Schrader 2003; Bohren *et al.* 2005). Longevity of seedlings is very high and can be up to 40 years in the soil, which is one crucial problem for the management of the species (see the management options for *Ambrosia*, section 7.2.3). Apart from human-related spread, the increasing rate of introduction of the plant is supported by the dispersal of seeds by wind and insects.

In general the plant has been characterised as "opportunistic" as it is very robust and able to survive under difficult conditions until they change and become favourable for the plant (Bohren *et al.* 2005). Such favourable conditions exist after harvest and when light availability increases. The plant can then grow in size and number very rapidly. This opportunistic character is additionally supported by the plants ability to regenerate quickly. The plant often is able to grow and to produce seeds after cutting, even when cutting occurs as late as September.

7.2.1.3 Pathways of invasion

Next to a natural dispersion by wind and insects, there are several accidental introduction path ways of *Ambrosia* by humans. A high proportion of *Ambrosia*-seeds become introduced as bird fodder with sunflower seeds (Zwander 2001), see Figure 19. Additionally, various grass-seed mixtures are highly polluted with *Ambrosia*-seeds (Schrader 2003). Plant imports for gardening and the transport of soil and substrates have been recognised as further pathways for the dispersion of *Ambrosia*-seeds (Taramarcaz *et al.* 2005). Especially the two both latter are typical pathways for secondary release.

Introduction pathways of *Ambrosia*-seeds that has not quantified yet, supposedly are public transport and trade (Schrader 2003). Along, roads, highways and rail roads the plant has become more and more present, especially in border regions in Germany with Austria, the Czech Republic and Switzerland.. Table 16 summarises mentioned characteristics of the plant.

Figure 19: Bird seeds with *Ambrosia*-pollen

Table 16: Characteristics of *Ambrosia* (own source)

Characteristics	
Name, (plant family)	• Common ragweed (*Ambrosia artemisiifolia*), (Asteraceae)
Synonyms	• Common ragweed, *Ambrosia elatior* Linnaeus, *Ambrosia elata* Salisbury, *Ambrosia panicolata* Michaux, *Ambrosia media* Rydb., *Ambrosia monophylla* (Walt.) Rydb.
Phaenology	• Annual plant, 20-150 cm growth height • wind and insect dispersal; wind pollination • seed production: up to 3000 seeds per plant, seed survival up to 40 years • blossom: July until October
Growth sites	• dry, nutrient rich, also saline disturbed sites • typical spots of occurrence: dump sites, way sides, hedgerows, acres, fallow
Home range	• USA, Canada
Occurrence in Europe	• Hungary, Slovakia, Czech Republic, Austria, Switzerland, France, Italy
Main Problems	• allergies due to pollen (asthma, rhinitis) • loss in agricultural yield (cultivation of sunflowers, soya bean, peas, maize)
Pathways	• unintentional introduction through pollution of seeds, crop and soil; mainly in bird fodder, with sunflower seeds, also in grain; plant imports for greening parks, international trade and transport

As the main focus of this thesis is to contribute to enhancing economic assessments of biological invasions, in the following the advantages of the concept of ecosystem services are tested in practice. This means it will be explored how the concepts contributes to (i) the *scope*-problem, (ii) the *uncertainty*-problem, and (iii) how the concept suits for the analysis of relevant stakeholders all in the context of the *Ambrosia* problem.

7.2.2 Contributions to the *scope*-problem: ecosystem services used as analysis structure for impacts of *Ambrosia*

The concept of ecosystem services used in the case of *Ambrosia* can serve in a first stance as an analysis structure for a physical assessment of the impacts the species. It thus contributes to the *scope*-problem. Using the list of ecosystem services helps to structure available information and determine whether we are in a situation of certainty, i.e. we are fully informed about an impact of a biological invasion, or a situation of uncertainty, i.e. we do not completely know whether, when and how the impact might occur. With the help of the concept of ecosystem services impacts of *Ambrosia* can be attributed to and determined for each ecosystem service, comprising both factual and potential impacts. Thus, decision support for how to deal with the plant is especially important due to the ongoing dispersion of *Ambrosia* throughout Germany. In order to determine how the concept of ecosystem services can provide decision-supporting information already at the physical assessment stage, in the following first factually occurring impacts are considered.

In the case of *Ambrosia* the *scope*-and the *uncertainty*-problem are inevitably linked to each other. Scarce information on the plants factual impacts indicate how much available knowledge is restricted yet. This means the list of ecosystem services used as analysis structure for all impacts of the invasive weed, not only impacts on ecosystem services which are fully known, but also those about which we are not fully informed. Regarding those impacts that are completely known, we recognise that it is only a small number of ecosystem services that have been recorded to be influenced by *Ambrosia* until now. Well-known and properly recorded impacts of the species are depicted in Table 17.

Table 17: Impacts of *Ambrosia* on ecosystem services (own source)

Ecosystem services	Impacts of *Ambrosia*	
	Type	**Description of impact**
Sustaining services		
Nutrient cycling		
Primary production and photosynthesis		
Provisioning of habitats	(-)	Shift in species: dominance in segetal flora of arable land
Provisioning Services		
Food and fibers	(-)	Loss of yield: production loss in peas, soy beans, maize, sunflowers
Fuel		
Genetic resources		
Biochemicals, natural medicine, pharmaceuticals		
Ornamentals	(-)	Loss of yield: production loss of harvestable sunflowers for bouquets
Fresh water	(-)	Loss of water: Competition with native plants for water
Regulating services		
Climate regulation		
Erosion control		
Water regulation		
Water purification and waste treatment		
Pollination		
Biological control		
Regulation of human diseases	(-)	Health problems: increase in asthma, hey fever, rhinitis due to the allergic reactions on *Ambrosia*-pollen
Storm and flood protection		

Ecosystem services	**Impacts of *Ambrosia***	
	Type	**Description of impact**
Cultural services		
Cultural diversity		
Aesthetical values		
Inspiration		
Spiritual and religious values		
Scientific and educational values	(+)	Object of research: research activities on *Ambrosia*-traits, ecological, economic effects
Recreation and ecotourism		
Cultural heritage and historical values		

Those ecosystem services for which knowledge is incomplete are discussed in section 7.2.4. Next to the description of the impact, the table shows the impact's character, i.e. whether it has a positive or negative influence on the respective ecosystem service. This is shown in the column "type" of impact.

Referring to the four groups of ecosystem services, the most well-known impacts of the invasive species on ecosystem services can be distinguished:
With respect to *sustaining services,* impacts of *Ambrosia* on (i) *soil formation and retention*, and (ii) *provisioning of habitats* have been recorded. As stated above, sustainable services happen on temporally and spatially large scales. Nevertheless, these services are locally supplied which enables the consideration of invasive species´ impacts with a local effect.
Impacts on the nutrient availability in the soil have been recorded and can be classified as a change in the ecosystem service of *soil formation and retention* (Nitzsche 2005). Although *Ambrosia* uses nutrients with an equivalent of 700 to 800 kg fertiliser per ha (Schrader 2003), the plant has been observed for not being nitrophilous (Brandes and Nitzsche 2006). Influences on the nutrient content in the soil, however, seem to be rather location-specific and their relevance for changing global and long-run sustaining services can be determined only with great difficulty.
Changes of local segetal flora are classified as changes in the service of *provisioning of habitats*. The invasive plant has been acknowledged to change the native weed associations of agricultural ecosystems in Hungary due to high dominance of *Ambrosia*-stands (Pál 2004). This shows that *Ambrosia* affects typical conservation matters because it changes the native composition of species and therefore also affects biodiversity on the species level.
With respect to *provisioning services*, impacts on (i) *food and fibers*, (ii) *ornamentals*, and (iii) *fresh water* have been detected. Impacts on the production of peas, soy beans, maize or

sunflowers refer to changes in the provisioning services of *food and fibers*, but also in the provision of *ornamentals*, if sunflowers are used for bouquets (Dernovici *et al.* 2006). Arable fields with peas, soy beans, maize and sunflowers can contain large stands of *Ambrosia*, but these large stands occur mainly in regions where *Ambrosia* is already quite problematic, such as Hungary and the Czech Republic (Pál 2004; Bohren *et al.* 2005). For cultivations of rape and wheat it has been recognised that the plant establishes dominant stands only after harvesting (Bohren and Delabays 2005). The high abundance of the plant results in the loss of yield of the recipient crop. In Hungary, the short ragweed has been recorded as the most frequent and most widespread invasive plant in every type of agricultural area (Pál 2004). Direct damages to arable crop happen due to competition for space, water and nutrients. Next to these direct impacts, *Ambrosia* is also host for several pathogens, e.g. *Plamopara halstedii* (problematic in sunflower cultivation), *Phyllachora ambrosiae* or alternative host for *Meliodyne arenaria*, *Erysophe cichoracearum*, or *Protomyces gravidus*, and thus indirectly influences crop production (CABI 2007).

The impact on *fresh water* stems from the fact that the plant consumes a lot of water and therefore generates competition for fresh water for other plant communities (Nitzsche 2005). It has been estimated that the plant uses about 2,000 tons of water per ha for an average *Ambrosia*-population. Water can be the limiting factor for the growth of the invasive weed (Nitzsche 2005). This impact, however, is difficult to declare as damage, as native weed species might also consume this amount of water in competition with crops.

Regarding *regulating services*, impacts of *Ambrosia* on (i) *regulation of human diseases* are known. These impacts are perceived as the most harmful effects of the species. The pollen of *Ambrosia* is responsible for a number of health problems. Asthma, rhinitis, conjunctivitis, hey fever, even states of anxiety, are reactions of people that are affected by the pollen (Gergen *et al.* 2000; Zwander 2001; Reinhardt *et al.* 2003: pp. 19; Schrader 2003; Genton *et al.* 2005). The long period of blossom during which the plant blossoms, can last until November. This enlarges the time of exposition to allergy inducing pollen and the allergic incapacitating risk period for patients (Dechamp and Penel 2002). Additionally, increases in (i) the high allergic potential of the plants pollen, and (ii) cross allergies with Artemisia, but also with Apple, Camomile, Melon and Celery have been recognised[56]. For about 50% of all asthma patients, the Asthma-inducing potential of *Ambrosia*-pollen are two to five times higher than for other allergic factors, such as grass pollen (Alberternst and Nawrath 2005) and is therefore feared

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[56] http://www.blumenboersen.ch/archiv/monatspflanze-ambrosie.htm 2004

for its long lasting and severe effects. The short ragweed pollen ranges in America, for instance, at the top of allergic outdoor factors (e.g. Gergen *et al.* 2000; Lanphear *et al.* 2001; Ownby 2002). About 15 Million US residents suffer from asthma (Weber *et al.* 2002), in Hungary every fifth person is assumed to have respiratory allergies (Török *et al.* 2003), and the short ragweed pollen has been found to be a significant cause of all these health problems. There are only a few quantified results that show the proportion of allergies induced by rag weed in comparison to other health problems. For instance, for the Rhône-Alpes region, it was determined that about 12% of the regions residents suffer from Asthma due to *Ambrosia* (Bohren 2005), and for the Swiss Rhine-valley about 10% of examined people (at the age of 15) have shown symptoms of a typical *Ambrosia*-allergy (Bohren 2005). Approximate estimations for absolute health costs range from € 17,000 to € 47,000 in Germany (Reinhardt *et al.* 2003: 20), for the Rhône-Alpes region about CHF 250,000 annually (Taramarcaz *et al.* 2005), and for Canada about US$ 49 Million annually, which include health and prevention costs (ibid).

Concerning *cultural services,* impacts on *scientific and educational values* are known, but as a positive effect. This positive effect refers to the high scientific interest in the species. Researchers have investigated its ecological traits. Its economic impacts are scientifically examined in order to show the role on human and ecological systems (see, for instance, issue 11/2006 of the "Nachrichtenblatt des Deutschen Pflanzenschutzdienstes"). The positive impact on this ecosystem service stems from interest in the plant from medicine, e.g. (Gergen *et al.* 2000), conservation research, e.g. Alberternst *et al.* (2006), as well as in biology (Brandes and Nitzsche 2006). It is the only positive change of an ecosystem service attributed to the short ragweed until now so far.

As we can see impacts on several ecosystem services have been recognised. Some impacts are very serious, such as impacts on human health, whereas others are of minor relevance due to their limited capacity to change the ecosystem service, as in the case of fresh water consumption. For a reduction of impacts of *Ambrosia* several management strategies are at hand. What strategies to encounter *Ambrosia* impacts exists, are explained in the following section.

7.2.3 Management options for *Ambrosia*

There is growing concern for the management of the species (see for instance www.ambrosiainfo.de). Prevention has been taken into account as far possible as one strategy that addresses the root of the problem. The success of this management strategy, however, is limited (see, section 3.4.1). Other management strategies, such as eradication and control are also possible. How far and to what extent the management options are at hand is now described.

The majority of currently available strategies in order for facing impacts of *Ambrosia* are derived from agricultural practice and from the health sector.

In contrast to available management options of the health sector, which only can target health impacts without reducing the population of *Ambrosia* directly, agricultural practice provides several measures that directly have an effect on the population of the invasive plant. (For an overview of the four general management strategies with different targets, see section 2.4).

7.2.3.1 Prevention, eradication, control: management options aiming directly at *Ambrosia*

A political demand for early detection of the plant and *prevention* has been articulated in Germany due to the ongoing dispersal of *Ambrosia* (e. g. Starfinger *et al.* 2005). Next to eradication and control, prevention aims directly at the population of an invasive species (see 2.4). To implement prevention is problematic due to Germany's location: its borders with countries that already face *Ambrosia*-problems and an unrestricted flow of good and services without custom border limit the possibility of successful prevention options. One preventive effort are so-called black lists, e.g. in Switzerland or the USA. Such lists show certain plants, among them the short ragweed, of which public trade is forbidden (Bohren *et al.* 2005). This kind of governmental effort to manage invasive species, can lead to a decrease as well as to an early detection of *Ambrosia*. Another approach that seems to be fruitful is the dispersion of information about the plant. Press and media, as well as certain political institutions, such as the Federal Agency for Nature Conservation (German: *Bundesamt für Naturschutz*) promote public awareness through information campaigns. To what extent information will prevent further incoming of the plants seeds into Germany remains questionable, especially when taking into account that prevention often fails (see section 3.4.1).

The vast majority of management options, however, aim directly at the invasive plant. They are common practice in agriculture, which favours *control* as a management option for

Ambrosia. But also *eradication* actions, in terms of pulling, plugging, weeding and burning have been recommended (Brandes and Nitzsche 2006). One reason for the lack of success of eradication strategies is the difficult identification of the plant in early stages for lay people (Essl 2005, personal communication). This fact hinders the realisation of appropriate local eradication as a management option.

The success of all eradication and control measures is limited because of the appropriate time for interfering in the growth process. This point in time has to be carefully detected: on the one hand the plant has to be interrupted in its growth before blossoming so that no viable seeds can be produced; on the other hand it must be guaranteed the plant does not have time to recover and produce seeds again. Optimal cutting time has been observed between 10th and 15th of September (see Table 18). This time period will probably vary with annual differences in the seasons (Bohren *et al.* 2005).

As the majority of well-known and already applied management strategies is control, in the following different types of this option are described:

Mechanical control has been recognised as most successful management option (Bohren *et al.* 2005). Pulling, cutting, mowing and plugging can be also used for control of *Ambrosia*. Mechanical measures are only recommended for small areas of infestation (Department of Environmental Protection 2005).

For larger areas, *chemical control* with herbicides is suggested. On cultivated land, the optimal time window to combat *Ambrosia* is after harvesting and during the different cultivation phases until the end of the vegetation period. The application of herbicides shows success, but if not used at the right point in time of the vegetation phase of the plant, it often fails and the plant sprouts again (Dernovici *et al.* 2006). Direct control with herbicides seems to have no long term effects (Schrader 2003). One has to deal with the same problems, as with those of mechanical measures: the appropriate point in time is most important. Furthermore, experience from other countries shows that populations of common ragweed can easily develop resistance to several herbicides (triazines, ureas and glycines (Dernovici *et al.* 2006)).

Biological control comprises the application of natural enemies, or competition with other plants for light so that the population of *Ambrosia* decreases. Biological control using natural enemies, however, has not been successful yet. More than 30 insects have been examined as potential biological control agents (Dernovici *et al.* 2006). This relates to the generalistic character of any potential enemy (such in the case of *Epiblema sternuana*). Its application would probably have effects on the cultivated crop, e.g. sunflowers, between which *Ambrosia* grows as a weed (Dernovici *et al.* 2006). Covering idle land with grass or harmless plant has been considered as the best available prevention strategy at the moment (Rhône-Alpes 2000).

Main reasons why ecological effectiveness of measures is usually below the target:

- Prevention in the majority of cases is not possible due to the unintentional dispersion of seeds via soil, substrates, and transport und traffic.
- Eradication is mostly not successful due to longevity of seeds in the soil. Thus re-invasion is common. Exceptions exist for the sand dune in Lower Bavaria (personal communication, Zahlheimer 2004).
- Control usually fails due to the growth properties of *Ambrosia* as a typical segetal plant with opportunistic habits and the resistance to many herbicides.

Table 18: control options for *Ambrosia*-management (own source)

Measure type	Measure	Description	Effect	Sources
Mechanical	Pulling Cutting Mowing Plugging	Each of the mechanical measures should get the plant near the root, measures should be done before August 1st; best right before the plant blossoms	Reducing effect depends strongly on exact point in time of intervention; no 100% eradication observed, data not quantified	Rhone-Alpes (2000); Bohren *et al.*(2005); Department of Environmental Protection (2005)
Chemical	Contact herbicides	Application of Round Up (1.2% solution); Atrazin, other herbicides	No complete reduction of abundance (not quantified); suitable dose effects on oats	Schrader(2003); Nitzsche (2005)
		Application of herbicides (Triazin, Imazethapyr, Linuron)	Reduction of abundance observed, not quantified, partly resistance	Brandes and Nitzsche (2006), Bohren *et al.* 2005)
	Salt	Spraying of salt on stem + leaves	Salt tolerance of *Ambrosia* restricted, reduction of abundance not quantified	(Nitzsche 2005)
Biological	Biological control	Application of control agents: • *Zygogramma saturalis* • *Puccinia xanthii*; *Ophraella communa* • *Epiblema sternuana*	➢ ineffective ➢ effects also on sunflower cultures ➢ not selective enough	Reznik (2000) Dernovici (*et al.* 2006) Schrader (2003)
	Soil coverage	Competition with other crops (sugar beet) or option to keep idle land covered with vegetation (grass)	Loss in yield of sugar beet no reliable indicator for success	Rhône-Alpes (2000), Department of Environmental Protection (2005); Nitzsche(2005)

To sum up, all kinds of control mechanisms do not work in terms of their ecological effectiveness. No matter if cutting or spraying has been applied, it can be only part of a strategy to combat *Ambrosia* (Bohren *et al.* 2005). Always a combination of strategies promises success (Taramarcaz *et al.* 2005). In the following adaptation as further management strategy will be explained.

7.2.3.2 Adaptation: a measure that aims indirectly at *Ambrosia*

In contrast to the direct measures applied in agriculture, there are other measures that aim to reduce impacts indirectly. Such options correspond to adaptation as explained in section 2.4. Local communities, such as the Rhône-Alps region provide a lot of information about the invasive plant in terms of information campaigns and brochures to the public. In Switzerland communal house owners inform their tenants about the plant and its problems (Grünig 2005). Next to the preventive character, the strategy of informing people also offers the opportunity for society to adapt to impacts of the invasive plant. This type of management raises public awareness, and appropriate handling of the plant can be the result. An example is the provision of instructions for private households. If they find *Ambrosia* in home gardens they ought to wear masks when approaching the plant in order not to breathe the pollen and to avoid direct contact. The instructions also demand home owners to put the plant in the rubbish and not into the compost to avoid further dispersal of the seeds.
Another indirect management option has been suggested for handling polluted grain and seeds. Cleaning seeds can offer an opportunity to reduce further transport and dispersion of the long viable seeds (Brandes and Nitzsche 2006).

To summarise, although there are efforts to manage the plant and to reduce the plants harmful impacts, but the success of all strategies is limited. The limited ecological effectiveness of management options for invasive species has been mentioned to be a matter of uncertainty (see section 3.4). Incomplete knowledge regarding the success of management strategies has to be supplemented by the fact that only for certain ecosystem services are impacts already known. They have been mentioned in section 7.2.2. How the *uncertainty*-problem regarding management success and occurrence of impacts of *Ambrosia* can be dealt with will be shown in the following section.

7.2.4 Contributions to the *uncertainty*-problem: structuring incomplete knowledge with the concept of ecosystem services

With the description of well-known and documented impacts of *Ambrosia* in Table 17, it has become clear that for many ecosystem services we do not have any information whether or not there is an impact. Other impacts are well-known but have not yet been reported in Germany. Regarding management strategies limited success of already known management options has been recognised. According to these facts, we are in a situation of uncertainty. How this fact can be dealt with will be explained in the following.

7.2.4.1 Uncertainty regarding impacts of *Ambrosia*

As illustrated above, only few impacts of *Ambrosia* have been known for sure and are illustrated in Table 17. Some of them, however, have been only recorded outside Germany. Thus, how likely are they to occur in Germany? What about all the other ecosystem services for which no impacts have been recorded yet? Do we have to expect changes of some of them anytime in the future? Or are impacts on certain ecosystem services very unlikely? Using the nomenclature of uncertainty of chapter 3, we can identify different situations of incomplete knowledge, which tell us, to what extent we can expect impacts of *Ambrosia* on ecosystem services. As learned from section 6.3.2, the concept of ecosystem services does not apply to situations of ignorance as this implies an outcome, i.e. an impact of an invasive species that cannot be addressed by one of the 26 ecosystem services. Thus, in the following ignorance is neglected. In section 7.2.4.3 science as an approach to reduce uncertainty, i.e. also ignorance will be discussed.

Now we turn to situations of incomplete knowledge that are within the scope of the concept of ecosystem services will be discussed. Modifying Table 12 from Chapter 6.3.2.2 the different situations of uncertainty regarding impacts of *Ambrosia* can be addressed.

Table 19: Different situations of knowledge for impacts of *Ambrosia* to happen in Germany (own source)

Situation of knowledge	Impact yet occurring	Occurrence in Germany
Certainty	Yes	Yes
Risk	Yes	No
Uncertainty i.n.s.	No	No

Depending on the fact whether an impact of the invasive plant has first already been occurring and second whether or not occurring in Germany, we have different situations of knowledge, namely certainty, risk and uncertainty i.n.s..

Certainty reflects a situation when an impact of *Ambrosia* on a specific ecosystem service has been already occurring in Germany.

R*isk* is analogous to a situation of certainty. The impact of *Ambrosia* on a specific ecosystem service can be clearly indicated however has not been identified in Germany yet. The literature suggests that such impacts are very likely to occur in Germany as well, as environmental conditions are similar to those of the infested regions (e.g. Taramarcaz *et al.* 2005).

Uncertainty i.n.s. exists if possible impacts of *Ambrosia* can be expected, but have not happened before elsewhere. Thus no precise description of the impact is possible at the moment. This kind of impact often refers to cascading effects due to an initial, well-known impact. An example is the expectation that *Ambrosia* not only affects agricultural, but also natural ecosystems that provide similar ecological conditions. Probabilities for such an outcome to occur are not available, however.

Table 20 illustrates the different situations of knowledge for the various impacts of *Ambrosia* on ecosystem services in Germany, showing whether the impact is perceived as a damage (-) or as a benefit (+).

Table 20: Impacts of *Ambrosia* and related situations of knowledge (own source)

Ecosystem services		**Impact of *Ambrosia***	**Situation of knowledge**		
	Type	**Description of impact**	**Certainty**	**Risk**	**Uncertainty i.n.s.**
Sustaining services					
Soil formation and retention	(-)	Change in nutrients in the soil	x		
Nutrient cycling					
Primary production and photosynthesis					
Provisioning of habitats	(-)	Shift in species: dominance in segetal flora of arable land		x	?
Water cycling					
Provisioning Services					
Food and fibers	(-)	Loss of yield: production loss in peas, soya bean, maize, sunflowers		x	
Fuel	(-)	Loss of yield			x
Genetic resources	(-)	Shift in genetic resources through introgression or hybridisation			?
Biochemicals,	(-)	Loss of yield			x

Ecosystem services		**Impact of *Ambrosia***	**Situation of knowledge**		
	Type	**Description of impact**	**Certainty**	**Risk**	**Uncertainty i.n.s.**
natural medicine, pharmaceuticals					
Ornamentals	(-)	Loss of yield: production loss of harvestable sunflowers for bouquets		x	
Fresh water	(-)	Loss of water: Competition with native plants for water	x		
Regulating services					
Climate regulation					
Erosion control					
Water regulation					
Water purification and waste treatment					
Pollination	(-)				?
Biological control	(+)	Application of research results for biological control		x	
Regulation of human diseases	(-)	Health problems: increase in asthma, hey fever, rhinitis due to the allergic reactions on *Ambrosia*-pollen	x		
Storm and flood protection					
Cultural services					
Cultural diversity					?
Aesthetical values					?
Inspiration					?
Spiritual and religious values					?
Scientific and educational values	(+)	Object of research: research activities on *Ambrosia*-traits, ecological, economic effects	x		
Recreation and ecotourism	(-)	Avoidance of invested areas		x	
Cultural heritage and historical values					?

For those impacts which occur already in Germany we are in a *situation of certainty*. These impacts are on the following ecosystem services:

1. *soil formation and retention* due to changes in nutrients in the soil
2. *fresh water* in terms of loss of water due to a high water consumption rate
3. *regulation of human diseases* because of health problems due to the allergic pollen of *Ambrosia*
4. *Scientific and educational values* due to the various scientific research activities concerning the plant

Out of the mentioned impacts only the impact on human health as a negative impact and the positive effect of being an object of scientific are of major relevance (see section 7.2.2).

Situations of risk exist regarding the following impacts on ecosystem services:

1. An impact on the *provisioning of habitats* is quite likely to happen in adjacent ecosystems due to the high dispersion rate supported by the wind. Although not recorded yet, this impact is very likely to happen in Germany. It has already been recorded e.g. for Hungary (Pál 2004).
2. Also an impact on the ecosystem service *food and fibers* and *ornamentals* is very likely as crops, such as sunflowers, are also cultivated in Germany. For this ecosystem service an impact that results in the reduction of crop yields is very probable due to similar conditions in such cultivations. It has already been recorded yet e.g. for the USA (Dernovici 2006).
3. A positive impact has to be expected in using research results of ongoing scientific activities for biological control. Although no appropriate biological control agent have been found yet, other methods, such as the approach to cover the soil with vegetation, will probably be elaborated, e.g. by the Federal Research Centre for Cultivated Plants (Schrader 2003; Dernovici 2006).
4. It is expected that people will avoid places that are infested by the plant or adapt outdoor activities to avoid high pollen counts. This impact would be similar to problems with the Giant Hogweed (*Heracleum mantegazzianum*). This plant also causes health problems and people avoid sites infested by the plants in order not to be directly exposed to the plants juice and its severe injuries when getting in making skin contact with the plant. Due to the similarity of experience in other European countries with the health effect of the Giant Hogweed, one can derive the assumption that people will react accordingly in the case of *Ambrosia* and will keep people out of infested sites (Nielsen *et al.* 2005: pp. 24).

Situations of uncertainty i.n.s. exist in the following context:

1. For several *provisioning services* impacts of *Ambrosia* have to be expected, because *Ambrosia* has been recorded to cause a loss in yield in cultures of soy beans, maize and sunflowers, one can expect the plant to also affect other cultures used e.g. for fuel production, i.e. rape or wheat. To determine answers whether or not there will be an impact has to be task of scientific investigation. An example will illustrate such a scientific question: *Ambrosia* needs idle land and no competition for light. Such conditions are provided by natural ecosystems, such as rockfields and gravel field

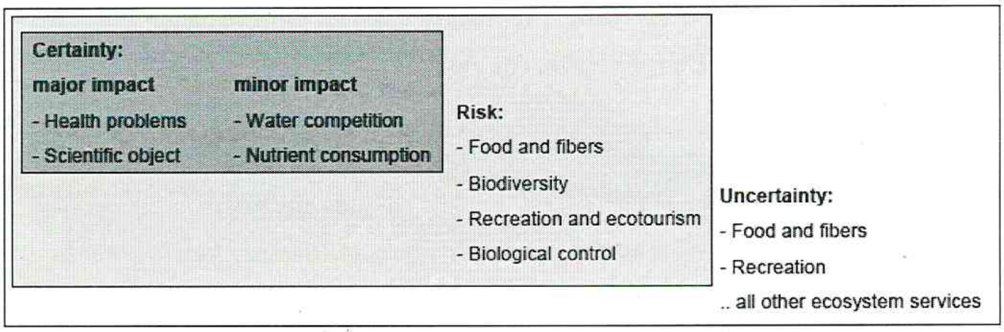
grasses. Thus these ecosystems are suspected to be prone to invasion by *Ambrosia.* Changes in their ability to provide habitats for a certain species might be a result of such an impact, although not recognised yet. Such an effect would be another impact on biodiversity on the species level.

2. For a group of *cultural services* further impacts might happen (those ecosystem services marked with a question mark). Although a hint whether or not a change of these ecosystem services will happen, does not exist right now, effects of, for instance, changes in recreational outdoor activities due to the high pollen counts in the air are imaginable. Changes in outdoor activities might lead to changes in aesthetical or cultural values of landscapes, if formerly touristic areas are now avoided.
3. For a number of *regulating services*, such as *erosion control* or *soil formation* and retention, at the moment there are no hints of change due to *Ambrosia.* A change of the respective ecosystem services even seems unlikely due to the fact that the plants ecological properties do not overlap with the ones of the ecosystem services. Impacts on ecosystem services related to hydrological phenomena, e.g. water regulation or water purification and waste treatment, seem unlikely as the plant favours dry conditions to grow.

   Nevertheless, to determine whether or not there is an impact on the respective ecosystem service, will be the task of scientific research. At the moment, a lack of results may stem either from the fact that changes in a number of ecosystem services that can be attributed to *Ambrosia* have not been investigated yet, or from the fact that there has no impact of *Ambrosia* on a number of ecosystem services been detected at this point.

Figure 20 summarises the different situations of knowledge. In the inner box the impacts that already occur in Germany are listed. Of these impacts we are certain. Changes in biodiversity refer to all situations of knowledge. A situation of risk exists due to the similarity of ecological conditions in German ecosystems, which are very likely to become invaded anytime in the future. Impacts in a situation of uncertainty i.n.s. refer to cascading effects on biodiversity induced by *Ambrosia* that are not foreseen right now, e.g. if *Ambrosia* will affect certain red list species that are of special concern to conservationists.

Figure 20: Impacts of *Ambrosia* according to their degree of available knowledge (own source)



Next to uncertainty about impacts of *Ambrosia*, the success of all management options is limited due to incomplete knowledge. Reasons for that have been outlined above. Now, they will be explained.

7.2.4.2 Uncertainty regarding the management of *Ambrosia*

Referring to section 3.4, uncertainty regarding the management of a species occurs in the context of the measures ecological effectiveness and its possible side effects.

For present management options of *Ambrosia* it has become clear that none of the measures are successful in terms of having an ecological effectiveness of 100%. There are two major reasons for that. The limited success of the measures can either (i) result from available data, or (ii) happen due to the ecological character of the invasive plant.

(i) Examinations about the success of measures have not been quantified, i.e. no data about the ecological effectives of a strategy is available. Next to the difficult quantification of success, control lacks the determination of a goal, especially relevant for control options. Setting a goal, for instance reducing impacts for about 50% according to the baseline, would help to identify the success of a measure. Such a precise goal is often missing. Instead, any kind of reduction of the population of the invasive plant is often considered to be a success, no matter how small it is.

(ii) The ecological characteristics of *Ambrosia* limit the success of any strategy. The plants opportunistic character of waiting for the suitable moment to grow is typical for segetal weeds and poses a problem for the management of all such plants, invasive as well as natives. Finding the appropriate point in time to interrupt the growth process is challenging.

(iii) Furthermore, the plants unintentional dispersion through the transport of soil substrates and grain polluted with *Ambrosia*-seeds explains why no measure will have an ecological effectiveness of 100%. The predominant reasons are either re-invasion or introduction into a new range (see above).

Side effects of *Ambrosia*-management have to be expected for the application of chemical and biological control measures. Side effects in this context are negative. Regarding the application of herbicides, typical effects of secondary damage from the use of chemicals, such as pollution of ground water, effects on non-target species, can be expected. Such a negative side effect has been already recognised for the spraying of sunflower fields (Dernovici 2006). Concerning the application of biological control agents, effects on non-target species, are known, evident in the case of *Puccinia xanthii*; *Ophraella communa* (see Table 18).

To summarise, the majority of management options to combat the short ragweed has yet not been successful, i.e. they do not reach a target of the respective management option. However, with the help of the management at least some reduction of *Ambrosia*-problems can be achieved, although often not the target. To what extent science can generate results that help to reduce uncertainty regarding impacts of *Ambrosia* and the success of its management is now discussed.
In the following section the task of research and science in exploring phenomena of *Ambrosia* will be discussed as it states a crucial approach for a successful management of the plant.

7.2.4.3 Scientific research: one way to reduce uncertainty regarding *Ambrosia*

One approach to reduce uncertainty, suggested by Faber *et al.* (1992), is further scientific effort. The less knowledge about impacts of *Ambrosia* we have, the more important is a scientific endeavour in questions determining to what extent the plant may impact currently non-affected ecosystem services at a future point in time. Addressing also questions regarding the improvement of management strategies will further limit uncertainty linked with the invasive plant. In situations of uncertainty, detailed descriptions of an expected impact on an ecosystem service or the success of a management option should be supplied. Results would provide approximate estimations of probabilities for the outcome to occur. Scientific efforts can result in a transfer from a situation of uncertainty i.n.s. to a situation of risk. A good example, where science could already reduce uncertainty, is the case of expected impacts of biological invasions on biodiversity, specifically the sustaining service *provisioning of* *habitats*. Earlier studies could not quantify any effect of *Ambrosia* on species composition and it has been doubted that there is one (Reinhardt *et al.* 2003: pp. 18). Meanwhile, research has shown that the invasive plant does have an impact on species composition, namely in the segetal flora in agricultural ecosystems (Pál 2004). So, further efforts might make it possible to answer the question of the range of ecosystems that might become infested by *Ambrosia*, and whether the plant will only occur in anthropogenic disturbed sites, or will also invade naturally disturbed sites.

Scientific research will also help to determine whether there are impacts on ecosystem services that are of minor relevance for the well-being of humans. Examples of such minor impacts are changes of the sustaining services of *nutrient cycling* and *water cycling.* To determine whether these impacts will be of minor or major relevance has a consequence on the management of the respective ecosystem services. Impacts of major relevance will require decisions about their management, whereas impacts of minor relevance will instead be neglected.

To show which decision makers are relevant for the management of *Ambrosia,* the following section depicts different stakeholders involved in problems regarding the invasive plant. Not only the people that are able to influence an ongoing invasion of the plant are relevant, but also those that are affected by the harmful effects of *Ambrosia.* To what extent the concept of ecosystem services is suitable for identifying stakeholders will be the object of the next section.

7.2.5 Identification of stakeholders involved in the *Ambrosia*-problem

As explained in section 6.3.2, the concept of ecosystem services can be used to assign impacts of invasive species, in this case of *Ambrosia*, to different stakeholders. The complete stakeholder analysis of the previous chapter will be now performed. Although the analysis in this thesis does not intend to be fully comprehensive[57], it is still of importance. Identifying addressees in terms of "who is responsible for, or affected by which impact?" illustrates that different stakeholders are involved in the invasion problem of *Ambrosia*, either by receiving impacts, or by driving the invasion. Those who are affected are usually not the ones that are drivers of the problem. To design effective strategies and to prevent further dispersal of the

[57] A comprehensive stakeholder analysis needs to take into account all institutional levels on which management decision regarding the invasive plant are made. As mentioned before, however, this was not intention of the thesis.

plant in Germany, this fact has to be taken into account.

In the following stakeholder analysis, all steps of (i) listing relevant stakeholders, (ii) characterising the relevant stakeholders, and (iii) determining their relevance for the decision process, are taken into account. The determination of the relevance of a stakeholder is guided by the idea of how a stakeholder possesses the ability to influence decisions regarding the management of *Ambrosia*.

Step one: Listing relevant stakeholders

Table 21 gives an overview of all stakeholders involved in the *Ambrosia*-problem. It shows that stakeholders can be damage recipients, i.e. these stakeholders are usually negatively influenced by the invasive plant. But also drivers of the invasions problem are listed, which means these stakeholders foster the *Ambrosia*-problem, for instance by dispersing propagules of the invasive plant. As mentioned in section 6.3.3.2, there are also negative drivers, i.e. stakeholders that promote restrictive management of the plant. These are stakeholders that typically implement management options. To illustrate the role of stakeholders in the context of *Ambrosia*, not only *primary* stakeholders that are *directly* involved, but also *secondary* stakeholders that *indirectly* influence the invasion process, have to be regarded. Table 21 lists all stakeholders, distinguishing primary and secondary stakeholders, but also recipients and drivers.

Table 21: Attribution of stakeholders for changing ecosystem services due to *Ambrosia* (own source)

Ecosystem service	Impact of *Ambrosia*	Stakeholder				
		Type	prim*	second**	Reci-pient	Dri-ver
Sustaining services						
Soil formation and retention	Change of nutrients in the soil					
Nutrient cycling						
Primary production and photosynthesis						
Provisioning of habitats	Shift in species: dominance in segetal flora of arable land	• Conservation		x	x	
Water cycling						
Provisioning Services						
Food and fibers	Loss of yield: production loss in peas, soya bean, maize, sunflowers	• Agriculture	x		x	x
Fuel	Loss of yield	• Agriculture	x		x	x

Ecosystem service	**Impact of *Ambrosia***	**Stakeholder**				
		Type	**prim***	**second****	**Reci-pient**	**Dri-ver**
Genetic resources	Shift in genetic resources through introgression or hybridisation	• Agriculture • Conservation	x 	 x	x x	
Biochemicals, natural medicine, pharmaceuticals						
Ornamentals	Loss of yield: production loss of harvestable sunflowers for bouquets	• Agriculture	x		x	x
Fresh water	Loss of water: Competition with native plants for water	• Agriculture • Municipality • Private households	x x x		x x x	
Regulating services						
Climate regulation						
Erosion control						
Water regulation						
Water purification and waste treatment						
Pollination						
Biological control	Application of research results for biological control	• Agriculture	x		x	X
Regulation of human diseases	Health problems: increase in asthma, hey fever, rhinitis due to the allergic reactions on *Ambrosia*-pollen	• Health care • Private households	x x		x x	
Storm and flood protection						
Cultural services						
Cultural diversity						
Aesthetical values						
Inspiration						
Spiritual and religious values						
Scientific and educational values	research on *Ambrosia*-traits, ecological, economic effects; informing about impacts	• Research • Media	x 	 x		x[1] x[1]
Recreation and ecotourism	Avoidance of invested areas; Informing about impacts	• Tourism • Educational institutions • Media		 x x	x 	 x[1] x[1]
Cultural heritage and historical values						

* primary stakeholder being directly involved; ** secondary stakeholder being indirectly involved;
x[1] negative driver promoting prevention and restriction of *Ambrosia*

Applying the typology of stakeholders from section 6.3.3.1 to the problem of *Ambrosia*, gives the following list of relevant stakeholders: agriculture, municipality, tourism, health care, conservation, private households, research and educational institutions. The list is complemented by adding stakeholders press and media. To clarify the specific role of the different stakeholders the second step of the analysis is conducted now.

Step two: description of stakeholders

The second step of the stakeholder analysis uses the distinction between recipients and drivers and whether a stakeholder has a dual role of being both. It explains how each stakeholder is interested in the phenomenon of *Ambrosia* by referring to the specific ecosystem service the stakeholder is concerned with.

Conservationists are stakeholders concerned with the ecosystems service *provisioning of habitats*. These stakeholders care about preventing negative changes to natural and protected species associations in the course of an invasion by Ambrosia. The only site, however, where *Ambrosia* has caused a problem yet, was the sand dune in lower Bavaria with its red list species (Alberternst *et al.* 2006). Next to the factual concern of conservationist about the shift in species composition, they also care about a change of natural genetic resources. However, at the moment such an impact is estimated to be unlikely, such as mentioned before. Next to their role as recipients, stakeholders from the field of conservation are also negative drivers because they enforce the CBD by actively protecting native species - and preventing establishment of *Ambrosia.* In Germany the problem of invasive species is mainly raised by these stakeholders. Not only are local working groups of conservationists starting action for eradicating the plant, also the Federal Agency of Nature Conservation is highly involved in increasingly circulating information to the public by providing large data bank on the most important invasive species in Germany (http://www.floraweb.de/neoflora/).

For stakeholders from the agricultural sector, impacts of *Ambrosia* on the ecosystem service *food and fibers* are important, because an invasion of the plant causes decreases in agricultural yields. If the plant starts to infest production sites of crop relevant for the production of fuel, such as rape or wheat, or of sunflower fields, an additional field of agricultural production would be affected. Agricultural stakeholders are very important, not only because they are in fact severely damaged by an invasion of *Ambrosia* on production sites. But also as negative drivers of invasions of *Ambrosia* that have the opportunity to interfere in the invasion, farmers are of special relevance. One important step would be if they attempt to produce non-polluted seeds. This result can be achieved by sieving seeds after harvest (Schrader 2003). A certification system would contribute to limiting dispersal of *Ambrosia*. Agriculture is essentially and directly involved in further dispersion of the plant, as many *Ambrosia*-populations have been found on arable land (Brandes and Nitzsche 2006). The application of control and eradication measures shows significant results, i.e. farmers act also as negative drivers by hindering further invasion of *Ambrosia*. Another important agricultural stakeholder in Germany is the Federal Research Centre for Cultivated Plants – Julius Kuehn Institute. A working group on invasive species organises an annual workshop that aims to establish an early warning system for *Ambrosia* in Germany (www.ambrosiainfo.de). Press releases of the institute foster public awareness of this plant. Furthermore the institute is one of the leading institutes that promote scientific research on phytosanitary management options (see below: scientific stakeholders). On the international level the institute is engaged in the EPPO (European Plant Protection Organisation) which is an important steering wheel for phytosanitary *Ambrosia*-management in Europe (EPPO 2002).

Private households are recipients as they face the problem of dealing with the various health problems induced by the plant. Aspects of the *Ambrosia*-problem relevant for this group of stakeholders are (i) suffering from diminished life quality due to allergic reactions, (ii) increase in spending money for medical treatment, and (iii) changing outdoor activities in order to avoid them. As a secondary effect (iv) an increase in a loss of working hours due to allergic health problems has been noticed (Burton *et al.* 2001). Thus, private households are confronted with impacts on ecosystem services, such as *recreation and ecotourism*, and *regulation of human diseases*. Private households are not only recipients but also essential drivers of the ongoing invasion of *Ambrosia*. The use of bird fodder contaminated with *Ambrosia*-seeds promotes the establishment of *Ambrosia* in private home gardens and has been identified to be one of the key dispersion ways of *Ambrosia* (Alberternst *et al.* 2006). About 75% of infested sites are estimated to be in private gardens (ibid). Households are one of the most important forces for daily prevention, eradication and control of the plant. By avoiding *Ambrosia*-contaminated products they contribute to prevention approaches of invasions, but also by detecting, reporting and cleaning *Ambrosia*-infested sites they support control of the species. The reporting of infested sites of private households was a campaign in Switzerland which showed considerable increase of *Ambrosia*-detections in private gardens (Bohren 2005).

Municipalities are involved in the *Ambrosia*-problem as they are usually responsible for the local management of the problem. They apply appropriate management options to restrict *Ambrosia*-stands. Next to monitoring and the responsibility keeping public places and facilities clean of *Ambrosia,* the management of municipalities also means to raise public awareness in terms of information and education campaigns. Municipalities are stakeholders with a dual role, i.e. being recipients and drivers at the same time. They are drivers because they often use and transport polluted soil and soil substrates, grass-seed mixtures contaminated with *Ambrosia*-seeds for public places. Also plant transport for planting for public areas pose additional pathways for the dispersion of *Ambrosia*-seeds. Local communities are also main negative driver in the management of invasive species. For instance, if *Ambrosia* enters new ranges along highways and local roads a continuing control and eradication by the local highway board might restrict further spread[58].

Health care stakeholders have to deal with the short ragweed problem due to the various health problems. *Ambrosia* imposes costs to the health sector, by increasing the demand for medical treatment, labour costs of medical doctors, and days spent in hospitals for reducing health problems of asthma, hey fever and rhinitis. These stakeholders gain great importance as they have to deal with the most serious effects of an *Ambrosia*-invasion in Germany, next to private households. To act as a negative driver and to limit *Ambrosia*-induced health problems implies that doctors, pharmacies and other health care facilities should provide as much information to the public as possible. An example is the announcement of the presence of ragweed pollen in the air (Taramarcaz *et al.* 2005).

There are various stakeholders from *research.* They act from a special position as they perceive the *Ambrosia*-problem as an object of scientific examination and thus recognise a positive impact on the ecosystem service of scientific and educational values. This may seem unusual, but scientists and researchers are an important group of stakeholders. They may act as a negative driver of an *Ambrosia*-invasion. An example is the DWD (*Deutscher Wetterdienst*). For several years now it has collected data on the pollen count in the air. This information is provided to private households and the health care sector, which helps to avoid allergic reactions by adapting outdoor activities. Furthermore, the Federal Research Centre for Cultivated Plants is highly active in scientific research, especially on biological control of the plant. A number of universities and German research facilities put a lot of effort into the

[58] The maintenance of road systems is an example of different responsibilities that should be analysed in detail for a comprehensive stakeholder analysis. Some roads, e.g. in down town areas, belong to local municipalities, whereas other roads, such as highways are under federal control.

exploration of ecological, economic and social impacts of the plant. Their role is of great relevance because scientific findings contribute and improve the base for finding an appropriate management of the invasive plant.

Educational institutions, such as schools, as well as *media*, meaning press and broadcasting, are the main also crucial negative drivers, as they can disperse information about the plant. If schools include information about invasive species in their curriculum, adaptive behaviour might start earlier than now. The interest of press and media in problems caused by *Ambrosia* has increased during the last years, if one considers the high number of newspapers and broadcasts on the plant within the last years (e.g. SPIEGEL ONLINE – 12. Juni 2006[59]).

Stakeholders from *tourism* are affected if people avoid formerly touristic areas, due to a high amount of *Ambrosia*. As yet, this phenomenon has only been determined in some parts in France, but might one day become relevant for Germany (Rhône-Alpes Service Sante Environnement 2000).

Step three: determination of the relevance of stakeholder for the decision process

When the stakeholder analysis was introduced in chapter 6.3.3 the third step, namely the determination of the different stakeholders′ relevance, was said to work only in a concrete example. Now, in the case of *Ambrosia* the relevance of the respective stakeholders will be described. Relevance for the decision process in this context is tightly linked with the ability of stakeholder to influence decisions about the appropriate management of the species. The criterion of the goal will be used to determine the relevance of a stakeholder (Brade 2005: pp. 53). This criterion explains to what extent a stakeholder pursues the overall defined goal of the CBD-approach of first prevention, second eradication and third control. According to the criterion this means, if one stakeholder aims at prevention and another at control, the first would be of greater relevance than the second. A comprehensive analysis of the relevance of each stakeholder cannot be given here, as the topic of the work has been the *scope*- and the *uncertainty*-problem. A comprehensive analysis of all stakeholders linked with decision processes would imply that all decision structures within the federal system of Germany, i.e. all administrative and institutional structures, and all levels of decision-making for an appropriate management need to be analysed.

[59] http://www.spiegel.de/wissenschaft/natur/0,1518,420856,00.html

In the following, the different management options for the *Ambrosia*-problem of each stakeholder are described. Next to the type of strategies that are at hand, the federal level of implementation is important. Of course, in the first stance, ideally a national management strategy will be aimed at, but the implementation usually happens at the local level. As one can see in the following, most of the stakeholders act on the local level. Thus, for an overview about the relevance of the different stakeholders for a successful management two questions are regarded for each stakeholder:

1. What management strategies are at hand?
2. At what federal level does a stakeholder act?

Information concerning these two questions is provided in Table 22. Regarding the first question, information is derived from the description of each stakeholder above and re-arranged. The different management options comprise measures that are factually implemented, but also ones that are potentially useful to the successful management of *Ambrosia.*

Conservationist stakeholders usually act on the local level by starting control or eradication of the plant. The Federal Agency of Nature Conservation, however, acts on the national level by providing information about impacts of invasive species on an internet platform and via a databank.

Agricultural stakeholders act on both the national as well as the local level. If farmers would use and provide exclusively *Ambrosia*-free seeds, they help to prevent further spread of the plant. On the national level, the National Farmers Union, for instance, could advise farmers to use cleaned seeds and it could release a directive.

Private households are very important to any type of management on the local level. The great number of private households offers the opportunity for full-coverage monitoring of *Ambrosia*-occurrence in Germany. Households reporting every newly infested site would contribute to successful monitoring nationwide.

Municipalities are also of great relevance as they are responsible for eradicating *Ambrosia* from reported sites in public places. Since railroads and roads are main pathways for invasions, controlling such public places is also of great relevance[60].

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[60] For a successful control and eradication management along railroads, roads and highways a closer look at the administrative responsibilities is necessary as these responsibilities are not only located at the municipal level.

Table 22: Available strategies of stakeholders involved in the *Ambrosia*-problem (own source)

Stakeholder	Type of strategy	Description of strategy	Scale of action
Conservation	Eradiation	▪ Pulling and plugging of stands by local activists	▪ Local ▪ National
	Control	▪ Mowing and cutting of stands by local activists	▪ Local
	Adaptation	▪ Information platform and databank on traits of *Ambrosia*	▪ National
Agriculture	Prevention	▪ Cleaning of seeds ▪ Utilisation of cleaned seeds	▪ Local/National ▪ Local/National
	Eradiation	▪ Pulling and plugging of stands in fallows and fields	▪ Local
	Control	▪ Phytosanitary management in fallows and fields	▪ Local
Private households	Prevention	▪ Consumer choice of non-contaminated bird fodder	▪ Local
	Eradiation	▪ Pulling and plugging in home garden	▪ Local
	Control	▪ Mowing and cutting in home gardens ▪ Reporting infested spots	▪ Local
	Adaptation	▪ Adaptation of outdoor activities ▪ Avoidance of infested areas	▪ Local
Municipality	Eradiation	▪ Pulling and plugging of *Ambrosia* in public places, roads	▪ Regional
	Control	▪ Mowing and cutting in public places, roads ▪ Careful transport of contaminated substrates	▪ Regional
	Adaptation	▪ Informing citizens how to deal with *Ambrosia*	▪ Regional
Health	Adaptation	▪ Information campaigns about health impacts	▪ National
Science	Prevention	▪ Detection of invasion mechanisms, traits that can be used for preventive management	▪ National
	Control	▪ Development of control strategies (herbicides) ▪ Detection of biological control agent	▪ National ▪ International
Media & educational institutions	Prevention	▪ Information campaigns to raise awareness	▪ Local ▪ National

Science is of indirect relevance for the management of *Ambrosia*. Scientific results can be used for the appropriate management of the species, e.g. if phytosanitary control strategies have been tested and proven to be successful they can be applied in practice. The provision of new scientific results usually happens at the national or international level, as this information is published in international and national scientific journals.

Media, educational institutions and stakeholders from the health care sector can act at all levels if they provide detailed information about the health problems the plant can cause and how people can react accordingly to reduce these health problems. If health insurance companies would foster to provide information on how to avoid allergies induced by the plant and also to encourage lay people to monitor and eradicate the plant, they would become the role of a great multiplier.

So, which stakeholders are most relevant for the appropriate management than others? Are those that promote prevention of greater importance for the management of *Ambrosia* than those that control the plant? And, is *Ambrosia*-management on the national level more important than on the local level?

When referring to the current state of the species occurrence in Germany two management approaches are necessary. First, as the plant has already been detected for several places, especially near corridors such as highways, control is a crucial part of every management option, in order to prevent further spread of the plant. Second information campaigns are necessary as preventive and adaptive strategy. For the management of *Ambrosia* that is successful overall, collaboration of all stakeholders is needed. Early detection on not yet infested sites on the local level is in the same way important as the aim of a national strategy for a successful management. This fact has also been underlined for other countries (Taramarcaz *et al.* 2005). Thus, concerted action addresses the collaboration at the national as well as on the local level of all stakeholders involved in the *Ambrosia*-problem. An initiative to establish a network for concerted action has already been started to become implemented by various stakeholders, among them the Federal Research Centre for Cultivated Plants, the *Deutsche Wetterdienst*, and several universities in Germany.

7.3 Conclusions from the case study: a physical assessment of Ambrosia with the concept of ecosystem services

The case study of *Ambrosia* was used to illustrated advantages of the concept of ecosystem services to enhance economic assessments of invasive species. So, to summarise, what have been the advantages and how did they work in practice?

To answer these questions the three major points where the concept offers enhancements for economic assessments are summarised in the following.

1. Contributions to the *scope*-problem:

The challenge of the scope-problem in economic assessments of invasive species is to consider all impacts on ecosystem services that invasive species may and do have. A determined analysis structure has been claimed to be necessary for the physical assessment in order to structure available information about impacts and to enable comparability between the results (see section 4.5.4.1). The list of ecosystem services has been proven to serve as an analysis structure for the impacts of the invasive plant. This also applies in the case of *Ambrosia*. The application of the 26 ecosystem services discloses how manifold impacts of *Ambrosia* can be. By using the different services as an analysis structure, available knowledge and information about current impacts could be structured and analysed. By doing so it becomes clear that *Ambrosia* affects not only just one group of services. The impacts the species has vary from changes in the locally supplied sustaining service of *provisioning of habitats*, to several regulating and provisioning services. The use of the concept of ecosystem services also reveals that compared to the comprehensive list of ecosystem services just few impacts have in fact been recorded in Germany. Among them is the great interest from science in the plant that corresponds to a positive impact in the sense of increasing *scientific values*. Other impacts, especially the one of *regulation of human diseases* has great importance due to its negative effect on human well-being. The great number of health problems induced by the plant does and will have great influence on the well-being of many people in Germany.

2. Contributions to the *uncertainty*-problem:

The challenge of the *uncertainty*-problem in economic assessments of invasive species is to display already at that stage of the physical assessment where we have to deal with uncertainty. This means incomplete knowledge about impacts on certain ecosystem services should be included and made explicit in economic assessment, not just in the economic valuation step.

As mentioned in section 7.2.2, compared to the entire list, only a few impacts of *Ambrosia* are known for Germany. Several other impacts are very likely to occur, i.e. we are in a situation of risk. For them it seems to be only a matter of time to recognise them in Germany because they already occur in adjacent countries or under similar conditions in other ecosystems. Other impacts are rather unlikely to occur when considering the species ecological properties and those of certain ecosystem services. As a species growing on dry and disturbed sites *Ambrosia* is rather unlikely to damage for instance riparian ecosystems. But, for all these 26 ecosystem services we are not certain whether, when and how an impact might occur. Thus, we are in a situation of uncertainty.

Also in this matter, the concept of ecosystem services enhances economic assessments. It helps to disclose the different situations of knowledge regarding the different potential impacts of *Ambrosia*, and whether they are likely or not. It helps to structure and analyse the different situations of uncertainty.

Furthermore the list of ecosystem services used as an analysis structure reveals where scientific research is needed in order to reduce uncertainty. By investigating questions about possible impacts uncertainty and even communal ignorance, as situation in which impacts happen that are outside the set of expected changes of one of the 26 ecosystem services, can be reduced. Some questions regarding impacts will be easy to answer, due to the probability of impacts to occur. Others will be very challenging, for instance, if effects of the plant on other ecosystem services have to be determined.

3. Stakeholder analysis:

The concept of ecosystem services provides another advantage. It is an attached stakeholder analysis. By determining which ecosystem service affects specific stakeholders, impacts of *Ambrosia* could be assigned to the recipient stakeholders. The analysis of relevant stakeholders showed that various groups with different interests are involved in the problem. Some of them are drivers of the problem that promote the dispersion and introduction of *Ambrosia*-seeds, whereas others are recipients as they receive the damages of impacts of the plant. Certain stakeholders are both driver and recipients. It also became clear that even if a stakeholder is only indirectly involved in the problems that an invasion of *Ambrosia* causes, it can be of great relevance for the management of the species. In this context, the role of media was explained as an important supplier of information. Although these stakeholders are not affected directly by the plant, they are decisive promoters and disseminators of information on how to deal with the problem.

To depict the different interests in the problem linked to *Ambrosia*, furthermore discloses that the invasive plant is not only of concern for conservationist stakeholders. To address the problem of *Ambrosia* as a matter of biodiversity loss and thus rather of nature conservation, is much too narrow. As seen with the stakeholder analysis, there are many different stakeholders that would not be addressed if *Ambrosia* would be only considered as a problem of biodiversity loss[61]. Severe health problems indicate that also private households and the health sector care about the development of the species in Germany.

With the help of the stakeholder analysis, also the different management options regarding the problem can be illustrated. Each stakeholder has a certain set of possible management strategies which can be applied in order to restrict further problems caused by the plant. By illustrating the different management approaches for each stakeholder a policy recommendation can be launched, namely the need for concerted action.

[61] Unfortunately biodiversity loss is often understood as species diversity and due to biological invasions causing the loss of species, conservation is mostly the political field that is assigned to the problem.

4. Conclusion

To conclude, using the concept of ecosystem services can decisively enhance economic assessments of *Ambrosia.* The great advantage of using the concept is the support of decision-making already at the stage of the physical assessment, and not, what is quite common, just after the economic valuation. This is an argument to talk about ecological-economic assessments as it takes into account that an economic valuation of impacts of invasive species is very difficult due to the high degree of uncertainty about the impacts. Hence, every economic valuation runs the risk of valuing what is only known for sure. Compared to the entire list of ecosystem services, only those impacts would be economically evaluated for which we are in a situation of certainty. A valuation that considers only impacts that are certain would be much to narrow as it neglects all those impacts that are likely and also expected, and hence underestimates the total impacts of biological invasions. The case study also supports the argument that problems of economic assessments of invasive species lie in the first step, in the physical assessment. The physical assessment determines the quality of the subsequent economic valuation as it determines the circumstances under which an economic valuation takes place. Only a small number of aspects of the problem with *Ambrosia* can be addressed with an economic valuation and that the majority of impacts cannot be evaluated at the moment. Of course, an economic valuation of certain impacts of A*mbrosia* can support raising awareness if results indicate enormous costs just for the few impacts. Decision aid which needs to account for net benefits of a management measure, however, is difficult because the success of a measure will remain uncertain. Furthermore, the quantification of the benefits is challenging due to the high number of vaguely known impacts that are aim of reduction by a chosen management option. Thus, it is questionable whether a physical assessment with the help of the concept of ecosystem services is not more decision aid than, for instance, cost-benefit analysis. Such a physical assessment can depict which impacts are already problematic, it shows which impacts may become problematic and who is involved in the problem and has the ability to start successful action, i.e. political recommendations can target collaboration of several stakeholders which usually would only pursue their specific interest.

7 Biological invasions in Germany and the case study of *Ambrosia artemisiifolia*

8 Final conclusions and recommendations for further research

Goal of this chapter is to critically discuss the findings of the thesis for economic analyses of biological invasions and the suitability of economic analyses for decision support. The chapter is structured as follows: in section 8.1, the main findings of the thesis are summarised; in section 8.2, the implications of these findings for the role of economic analyses of invasive species in decision-making are discussed; section 8.3 discusses the shortcomings of the findings and outlines further research needed in the research field of the economics of biological invasions.

8.1 Summary of main findings

This thesis addresses the growing demand for economic assessments of global environmental problems such as climate change (see for instance the Stern-Review (2007)), biodiversity loss (see for instance the Potsdam Initiative (CBD 2007)), and in particular biological invasions. Economic assessments are needed in order to support decision-making and the selection of appropriate management options. Many case studies in this field already exist and this work does not intend to simply add to this literature. Instead, the thesis aims to develop a consistent framework that ensures a sound assessment. Sound assessment results are of great importance, which has been shown by a review of current literature. The survey of current economic studies on invasive species, performed in chapter 5, has disclosed that such case studies are often not suitable for supporting decision-making. Doubts therefore exist about the reliability of the high cost figures that these assessments typically show. Since such case studies concentrate on the gross costs but not on the net benefits of management, they help to raise awareness but do not offer advice on the appropriateness of management options. Apart from their lack of suitability for decision aid, methodological problems with economic assessments of biological invasions also exist.

This thesis identifies an urgent need to improve the basis of economic assessments to ensure results with higher quality and reliability. This basis is constituted by the physical assessment, in which all impacts of a biological invasion can be comprehensively recorded, analysed and described. Economic valuation as a second part of economic assessments is based on the results of the physical assessment. The physical assessment phase, however, is systematically ignored in many economic assessments. As a result, impacts that occur indirectly or are not easy to quantify are neglected, although they have an influence on human well-being. As a consequence of such results, on the one hand the limited number of impacts that are valuated in monetary terms might indicate wrong costs and benefits because too many impacts have overseen. On the other hand, results are not comparable as each study considers a different set of impacts. Comparability is a prerequisite if the economic assessment should be used for management decisions that face budget constraints. To set priorities is necessary in this context. This means decision makers ought to emphasise the management of those species for which the net benefits of their management are highest.

One way to approach the mentioned problems is a standardised framework for the whole assessment procedure. This thesis has therefore devoted special attention to this step, in contrast to the majority of economic studies of biological invasions, which primarily focus on the monetarisation of certain given effects. The focus on the physical assessment as an innovation in the field of economic assessments of biological invasions is justified by the fact that the major problems with these assessments arise because the initial stage of the assessment is commonly ignored. Thus, to improve the basis of economic assessments of biological invasions, taking account of full range of the physical effects of invasions has been found to be a crucial step towards improving decision-making. This means there is a need to really enlarge the scope of economic assessments by the ecological side, i.e. assessments have to become real ecological-economic assessment. To achieve this enhancement has been the goal of the thesis by systematically analysing biological invasions with respect to their economic and ecological properties. As a consequence of the examination of ecological and economic properties, two major challenges for ecological-economic assessments have been identified:

1. The *scope*-problem, which targets the range of impacts due to an invasion, their magnitude and the need to consider all impacts within ecological-economic assessments.
   The *scope*-problem occurs when considering the wide range of impacts, some of major and direct influence on human well-being, others of minor and indirect relevance, but still meaningful. Thus, how can it be guaranteed that minor, possibly not quantifiable impacts are adequately taken into account? This thesis has argued that for this problem the concept of the Total Economic Value (TEV) is inadequate as it concentrates on categories for the impacts of biological invasions that are not specific enough and is too much rooted in economic theory to meet the problem of biological invasions. As a concept commonly used to guide an economic valuation of the environment, the TEV does not prove to be

suitable for physical assessments of the impacts of biological invasions. An analysis structure to record all impacts is needed.

2. The *uncertainty*-problem acknowledges that knowledge about impacts and the success of management options is incomplete and that this fact must be made explicit.

   The *uncertainty*-problem emerges because usually neither all impacts of biological invasions, nor the time and the location of occurrence and the severity of damages are known. Often not even the probabilities with which impacts are likely to occur are available. Reason for that is a lack of knowledge regarding the ecological mechanisms of biological invasions. As a consequence of these unknown mechanisms, the success of management strategies in preventing or controlling biological invasions is also uncertain. Both aspects greatly increase the difficulty of making "good" decisions on management policies.

The thesis shows how both challenges, the *scope*- and the *uncertainty*-problem, can be met. The essential improvement of economic assessments to become ecological-economic assessments of biological invasions suggested in this thesis is the application of the concept of ecosystem services, such as used by the Millennium Ecosystem Assessment (MEA). The concept offers a list of defined ecosystem services which can serve as a comprising analysis structure. Working with this list enhances the entire procedure of the physical assessment of biological invasions because it provides information that can support decision supporting even before the economic valuation of impacts takes place. The following paragraphs explain why this way of proceeding is innovative and offers some crucial advantages to decision-making.

The thesis has argued that using the concept of ecosystem services contributes to the *scope*-problem as it makes it possible to explicitly indicate the different kinds of impacts of invasive species on human well-being, including direct as well as indirect effects. The list of ecosystem services thus presents a comprehensive system for an adequate analysis of impacts. This list also includes impacts on sustaining ecosystem services that are of long-term relevance, and act on the global level but are locally supplied.

Such services are often hastily neglected in conventional economic assessments as the marginal valuation of sustaining ecosystem services is difficult or even impossible due to their fundamental role in maintaining all ecosystems. However, it is possible to describe changes in these services due to biological invasions in the physical assessment phase, as no quantification is needed yet at this stage of an ecological-economic assessment. With the help of the concept of ecosystem services, a structured and systematic way of proceeding is possible, because the impacts of biological invasions are recorded for each of the ecosystem services. The application of the concept of ecosystem services enlarges the scope of economic assessment to factual ecological-economic assessments.

Furthermore, the concept of ecosystem services addresses the *uncertainty*-problem by making a systematic detection of uncertainty possible. Structuring the available information about impacts on the different ecosystem services reveals that often only a few impacts are fully known. For the majority of ecosystem services, however, neither the impacts that can be expected nor their extent have been clearly identified. Thus we are in a situation of uncertainty. This means that the fixed set of ecosystem services used as analysis structure not only shows the factual impacts, but also for which ecosystem service an impact is uncertain or likely.

This thesis has applied the concept of ecosystems services in a case study of the invasion by the plant species *Ambrosia artemisiifolia* in Germany. The case study has underlined how scarce data on impacts usually are. For *Ambrosia artemisiifolia* at the moment we only know its impacts on human health and the production of food, and we know that there is scientific interest in the phenomenon. Impacts on species composition, which is an aspect of biodiversity, on water regulation or nutrient availability are hardly documented. For all other ecosystem services, almost nothing is known about whether and when the species will have an impact. The case study has demonstrated that with the inclusion of the full range of all ecosystem services the different degrees of knowledge concerning impacts can be made explicit. This is a great and innovative advantage of the concept of ecosystem services, because uncertainty is a major problem with biological invasions. By using the concept of ecosystem services, uncertainty can now be included already at the physical assessment stage. This fact stands in contrast to CBA where uncertainty is usually considered after the monetarisation of impacts via sensitivity analysis. CBA might even be unfeasible due to the high uncertainty. The suggested procedure, however, enables an assessment although knowledge is limited.

An additional element of the systematic investigation and recording of the relevant impacts of invasive species can be a stakeholder analysis based on the concept of ecosystem services. This can be achieved by linking relevant stakeholders to the different ecosystem services. By doing so, the relevance of stakeholders involved in problems caused by an invasive species can be disclosed. Hence, if used alongside the concept of ecosystem services, stakeholder analysis offers a method to identify (i) the stakeholders that drive the problem, (ii) the stakeholders that are recipients of damages. Moreover, such an approach to stakeholder analysis based makes it possible (iii) to illustrate available management options for each stakeholder, and (iiii) to outline management recommendations on the base of available management options for each stakeholder.

In contrast to CBA, with the presented stakeholder analysis more than just losers (those who bear the costs) and winners (those who gain the benefits) of biological invasions′ management can be analysed. Apart from the distributional aspects of (the costs and benefits of) a management option, recommendations for the general management itself can also be given, as drivers of the problem can be identified, which will not be captured by CBA. This is an essential advantage compared to CBA, as it indicates decisive driving forces of the problem of biological invasions and allows the development of management strategies that go beyond the optimisation of net benefits. As a result of the stakeholder analysis it is evident that combined efforts are necessary for implementing a successful strategy. Each stakeholder acts in a different way and on a different level, i.e. only the combination of local eradication measures of nature conservationists, the monitoring of private households and the implementation of nation-wide strategies for preventing further accidentally incoming propagules of alien species, such as the certification of non-polluted seeds in agriculture, can lead to a restriction of further invasions. Thus, cooperation is strongly recommended.

Concluding from the findings of the thesis, ecological-economic assessments which use an analysis structure constituted by the set of ecosystem services offer an improvement over current economic analyses used in decision-making. Further implications for economic analyses, such as CBA, will be explained in the following.

8.2 Implications of the findings for economic analyses

The findings of the thesis have implications for the general role of economic analysis as a decision support tool. At the moment, there seem to be two major fields that are concerned with biological invasions and their economic aspects. On the one hand, there is the field of ecological economics research that studies problems on the interface between ecology, economics and society; on the other hand, there is the field of politics that has to make urgent decisions about problems such as invasive species. Both fields have developed specific attitudes towards environmental problems such as biological invasions, and how decisions about them can be supported. Accordingly, both fields use tools they consider to support decision-making appropriately.

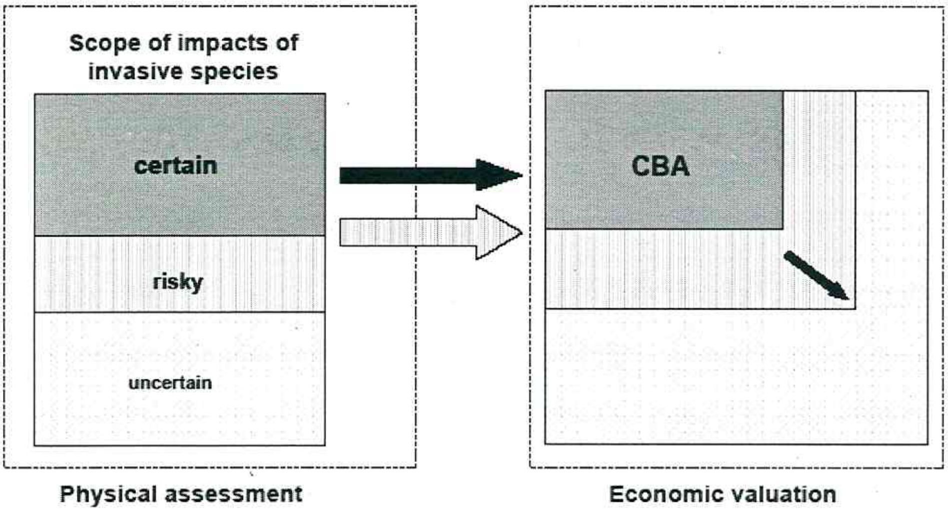
Among many *ecological economists* the demand for deliberative evaluation techniques in order to work issue-driven has been strongly addressed (e.g. Wilson and Howarth 2002; Cook and Proctor 2007; Spash 2007). These scientists acknowledge that "typically facts are uncertain, values in dispute, stakes high and decisions urgent" (Funtowicz *et al.* 2002: 53). Such a description of environmental phenomena certainly holds for the problem of biological invasions. One method suggested to appropriately support decision-making concerning such problems is SMCA - social multi-criteria analysis (De Marchi *et al.* 2000). According to this method, problems are considered in a local or regional context and thus solutions ought to be developed within this frame. It enables the inclusion of the different values and stakes, but also to deal with uncertainty (ibid). This means, only results produced on a deliberative basis and in the context of a defined area can provide decision supporting results that meet the demand of society.

The application of SMCA to developing policies on biological invasions implies some challenges: a successful management needs to be implemented, of course, on the local level, but also on the national or even European wide level, especially if the aim is to achieve prevention of biological invasions. Local solutions contribute to a spot-specific management and will mostly target control. The need to set up national and European early detection strategies is strong, see, for instance Klingenstein *et al.* (2003: 24}. The number of stakeholders involved in the development of international policies, however, is very large and getting their co-operation may be difficult. How uncertainty about the ecological mechanisms of biological invasions can be taken into account in such an international decision process, is as yet an unanswered question.

In contrast to suggestions of ecological economists, *politicians* (but also many neoclassical economists) often prefer another approach, namely CBA. It is still a favoured method for decision-making although criticised in many ways (see for instance Pearce, Atkinson *et al.* (2006)). It supplies results that are simple to understand (because expressed in the unit of money), comparable and decision advice in terms of costs and benefits is clear. A decision pro or contra a measure just follows the criterion of net benefits, i.e. if benefits outweigh costs of a measure it ought to be implemented. The various problems concerning CBA have been discussed in this thesis (see section 4.5). An issue such as biological invasions, for which many impacts are unknown, cannot be assessed adequately with CBA, because this method can only account for impacts that are well-known and properly quantified, i.e. for which we are in a situation of risk. Thus, only a small part of all impacts of a biological invasion will be considered in a decision-making process that looks only at costs and benefits. Politics therefore runs the risk of making wrong decisions when relying solely on a decision support instrument that neglects a great number of relevant but non-quantifiable impacts. As further consequence of neglecting non-quantifiable impacts of biological invasions, CBA misses relevant stakeholders that function as drivers of the invasion problem. In the case of invasive species, there are stakeholders that are important but outside the scope of CBA. Such stakeholders are often not directly involved in the problem but important for a successful management. If CBA, for instance, focuses on health costs of *Ambrosia*, agriculture as an important driver of the problem is unlikely to be incorporated in the assessment procedure. To use the entire list of ecosystem services, however, prevents such a narrow scope of ecological-economic assessment as it shows that various stakeholders are involved in the different impacts of one invasive species. To identify the agricultural sector as a driver of *Ambrosia*-invasions, for instance, is crucial when providing decision supporting assessment results for possible management options to control this invasive species.

The results of the thesis meet both the *scope*- and the *uncertainty*-problem of biological invasions: The offered procedure suggests using ecosystem services as analysis structure. Doing so offers an opportunity to take uncertainty into account, identify stakeholders that are relevant for the decision process about the management of invasive species, and incorporate monetarisation of impacts as well. This is considered to be the great contribution of the thesis. Figure 21 shows how, with the help of the concept of ecosystem services, the suggested physical assessment of the impacts of biological invasions enlarges the scope of ecological-economic assessments. The Figure shows that a physical assessment considers impacts that are certain, quantifiable and thus become monetarised in a CBA (black arrow), but also those impacts are taken into account that are risky (i.e. estimations about their likelihood to occur are at hand), or even uncertain (i.e. no probabilities about their occurrence is available). By using the different ecosystem services as analysis structure, the depiction of the different situations of knowledge regarding the specific impacts can be done in a very structured way as for each ecosystem service the respective degree of knowledge can be shown. The advantage of using the concept of ecosystem services is the guaranteed performance of risky impacts (light grey) in CBA, because the list clearly indicates them. Such risky impacts are in practice often neglected. Thus, the structure of ecosystem services enables taking account of minor and maybe risky impacts (light grey arrow).

Figure 21: Illustration of the enlarged scope of ecological-economic assessments with the physical assessment of biological invasions impacts



However, not only risky impacts can be taken into account, also impacts that are outside the scope of standard CBA because they are uncertain. Thus, CBA remains part of ecological-economic assessments of biological invasions. But with the help of the suggested analysis structure it is supplemented by an assessment of those impacts that are likely or uncertain but not quantifiable, but nevertheless important to decision-making. Working with the list of ecosystem services furthermore allows comparability of assessment results which is very difficult to do with CBA. The results of CBA vary greatly depending on the investigated species and the most obvious impacts of the species. In contrast, ecological-economic assessments along all ecosystem services would use the same basis, namely the complete list of ecosystem services. If CBA is part of ecological-economic assessments that are based on ecosystem services, it will still only be possible to monetarise certain impacts but not all. With the proposed approach, however, the capacity for decision support is not fully exhausted. Economically evaluated impacts via CBA are only one part of an economic assessment of invasive species. The overall frame of ecosystem services shows that what differs is the number of certain, risky and uncertain impacts. Therefore, it can be indicated that for one species more impacts are known than for another. Uncertain impacts belong to a field where research has to provide additional knowledge. For urgent political decisions, however, the extent of uncertainty on which the decision is based, becomes obvious with this procedure. It thus becomes integral part of the decision.

8.3 Remaining problems and recommendations for further research

Although the suggested procedure enhances challenging problems of ecological-economic assessments of biological invasions, a number of problems still remain. A few of them are discussed in the following.

1. Addressing research to provide answers to open questions of biological invasions, i.e. to reduce uncertainty, is a time consuming process, however. It will probably take many years for the many open questions to be answered by science. In the meantime, uncertainty endures and political decisions about the problem about invasive species will continue to have a significant degree of urgency. The results of the thesis do not provide a final solution to this conflict, except that the open questions are explicitly identified and that the framework offers possibilities to deal with this.
2. The concept of ecosystem services is prone to double counting during the economic valuation phase, which is a problem of this phase of ecological-economic assessments in general (see for instance Hein *et al.* (2006)). In the context of invasive species this means that if an invasive species changes a sustaining service, its effect automatically has consequences for all other services because supporting services maintain the overall functioning of all ecosystems. The basic reason for double counting is the dependency and interaction of the different ecosystem services, which are hardly known. To fully exploit the potential of the concept of ecosystem services, these interactions must be known, which implies there is a need for further research in this field.

When in this thesis the advantages of the concept of ecosystem services in improving ecological-economic assessments of biological invasions′ impacts were outlined, it was also mentioned that with the detection of uncertainty, also the need for further scientific research can be addressed. There are two major fields which seem to be of particular relevance for further scientific research in order to support decision-making. One is concerned with feedback mechanisms between the different ecosystem services, the other one with the fundamental role of stakeholders.

The first field of investigation targets further ecological research. Generalised explanations on mechanisms that drive the invasion process will help to reduce communal ignorance and uncertainty. Knowledge on ecosystem complexity, i.e. the functioning of feed backs between different ecosystem services will also help to understand how one impact on an ecosystem service of an invasive species can trigger the change of others. An improved knowledge about these processes is a decisive element.

The second research field addresses aspects of stakeholders involved in the invasion problem. As shown throughout the thesis, humans are main drivers of the phenomenon. A deeper understanding of the way how different stakeholders promote or restrict invasion, is necessary for the well-aimed management of invasive species. Two points should gain further attention here: the first is a focus on the institutional framework of decision-making for the management of biological invasions. To elicit the different decision responsibilities on the federal or the municipal level would improve the basis for the creating successful strategies. The second focus considers a stakeholder's motivation for the management of invasive species. The personal motivation is a matter of perception and reasons to start action against invasive species might stem just from the character of such species being "invasive and alien" or from factual damages a person faces. There are many examples where invasive species are feared for being "alien" even though they are actually not more harmful than native species. Invasive species seem to threaten cultural identity which is tightly linked with old grown middle European landscapes. Such "social costs" point to the need to leave the narrow understanding of assessing costs in terms of marginal monetary changes and opens the door to a wider understanding of what an (ecological) economic assessment of biological invasions comprises.

To sum up, the present thesis has shown that an analysis on the interface of ecology and economics is necessary to understand the phenomenon of biological invasions and to enlarge the scope of economic assessments to become real ecological-economic assessments. By focussing on the physical assessment of impacts of biological invasions, decision supporting results are provided already before the monetarisation of impacts. The great advantage of this focus is that although major problems of biological invasions linked with high uncertainty, scarce data and the range of impacts of biological invasions still exist, decision supporting results can be provided. The proposed integrative assessment procedure offers a more realistic picture of biological invasions than CBA can. A realistic view, however, is important for politicians in the design of management strategies and does not only mean sound numbers of impact damages, but also that not only quantifiable impacts are considered and that stakeholders are properly addressed. This is guaranteed by the proposed ecological-economic assessment of biological invasions, such as presented in this thesis. The advantage of this procedure is furthermore that politicians do not have to wait neither for scientists to reduce uncertainty by providing further knowledge on biological invasions, nor for comprehensive cost-benefit analyses. With the analysis structure of ecosystem services there is the opportunity for a qualitative and cost saving description of the problem. There are many cases where such a description is sufficient to start action, although impacts of invasive species have not been quantified.

Ecological-economical assessment is a term used in manifold ways and often demanded in the field of conservation nature conservation. This thesis takes the meaning of the words seriously and provides a really interdisciplinary approach. It constitutes one step towards a pluralistic understanding of what problems exist concerning biological invasions and how decision-making can be improved.

9 Summary (*English* and *German*)

This thesis deals with the economics of biological invasions. There is an increasing demand from decision makers for economic assessments. Their application, however, is often problematic. Main reasons for this are the complexity of the assessed good, uncertainty about invasion mechanisms that drive damaging impacts, and insufficient knowledge regarding effective management strategies. But also methodological problems have to be considered. Due to those reasons a holistic approach is needed that provides sound results suitable to support decision-making under conditions of limited data availability.

Main aim of this thesis is to realise a contribution to the methodology for an economic assessment of biological invasions. In order to achieve this aim, a comprehensive analysis of biological invasions has been performed that takes ecological as well as economic properties of the phenomenon into account. Ecological characteristics, especially the invasion process, are important to understand the phenomenon of biological invasion in general. Management strategies have to be adapted to the different phases of the process. Economic properties, such as the public good character of biological invasions, demand specific economic valuation techniques and tools.

Economic assessments contain of two parts:

1. The physical assessment and
2. The economic evaluation.

The thesis considers both steps while taking ecological and economic properties of the problem into account. By means of a literature survey it was shown that major challenges lie within the first step. Thus, the thesis focuses on the physical assessment. Recent economic studies on biological invasions extensively regard economic valuations of biological invasions with its costs and benefits. However, framework conditions as the very base for an economic valuation have been neglected so far, despite their importance for the quality of economic assessments. An urgent need for an enlargement of current economic assessments by an ecological dimension has been noticed, and thus to develop an ecological-economic assessment.

By focussing on the physical assessment two major problems prove to be of importance: (i) the range of impacts that have to be covered by an economic assessment, the *scope*-problem, and (ii) the inclusion of uncertainty (as a synonym for incomplete knowledge), the

uncertainty-problem. The *scope*-problem considers the challenge to encompass all impacts that occur due to biological invasions because all of them are relevant for human well-being. The *uncertainty*-problem arises in the context of impacts (when, how, and where do invasive species have effects on ecosystems and humans), but also regards the success of management strategies.

This thesis presents an approach to improve economic assessments with the concept of ecosystem services, such as the one used by the Millennium Ecosystem Assessment. Ecosystem services are suggested to be used as a framework for the physical assessment of biological invasions. By doing so it contributes to the *scope*- and the *uncertainty*-problem. Regarding the *scope*-problem, the encompassing list of ecosystem services serves as a defined analysis structure, which has been lacking in the majority of surveyed studies. Such a structure enables a profound analysis of impacts since each impact can be illustrated via the affected ecosystem service and this overview is encompassing due to the various services an ecosystem provides. Furthermore, with the help of ecosystem services a standardised base for every physical assessment is possible. The structured analysis of impacts of invasive species also helps to elicit available information and to identify knowledge gaps.

To address the uncertainty-problem in an adequate manner a nomenclature based on the concept by Faber *et al.* (1992) has been developed. Next to risk and uncertainty in the narrow sense the concept refers to ignorance as another type of uncertainty. Especially this last type is of relevance for the invasion problem, as often enough neither all effects of the invasion are known nor their probabilities. For each ecosystem service the degree of knowledge can be depicted accordingly, i.e. whether an impact is already occurring, likely or uncertain to occur. Uncertainty is thus made explicit already at the stage of the physical assessment, which is a great advantage compared cost-benefit analysis (CBA) which takes usually uncertainty, via sensitivity analysis, only into account *after* the economic evaluation. To make uncertainty already explicit at this early assessment stage, enables the decision maker to include it right from the start of the assessment and allows assessment under conditions where CBA cannot be applied due to lack of data on social costs and benefits.

The third great advantage of the concept of ecosystem services is the possibility to undertake a structured stakeholder analysis. For each ecosystem service certain stakeholders are relevant. Through this linkage, stakeholders who are concerned with impacts of invasive species can be identified. Such an analysis contributes greatly to the supporting role of assessment results because identifying relevant stakeholders together with their available set of management options means to detect drivers of the problem. Results might suggest drivers to collaborate with stakeholders who are affected by the invasion and who possess different management strategies than drivers do. As the case study of common ragweed *(ambrosia artemisiifolia)* shows collaboration makes successful management more likely than the implementation of strategies for each stakeholder. This is valuable information for decision makers.

The outcome of this thesis poses an enhancement of economic assessments of biological invasions due to three reasons:

- They provide decision supporting information at an early assessment stage and before the economic evaluation is performed,
- They enlarge the scope of an economic assessment decisively. CBA as standard tool focuses on those impacts that are quantifiable and on those for which we have probabilities. With the help of the suggested procedure also those impacts are taken into account that are uncertain but nevertheless relevant for the decision. This last information is of great importance for the decision maker as he will understand the base on which the decision is made, i.e. on how much uncertainty the assessment is based.
- The attached stakeholder analysis can identify drivers of invasion problems, which would not be possible with a CBA that merely concentrates on winners and losers.

Nevertheless, CBA ought to be an essential part of economic assessments of biological invasions. But they should not be exclusively applied, as this would underestimate the complexity of the problem and the broadness demanded to assess it appropriately.

This thesis emphasizes that with the help of ecosystem services the very base of economic assessments will be enhanced. By conducting real ecological-economic assessments results can be provided suitable for decision aid before the monetarisation of impacts, which is usually hardly possible, anyway. Thus with the proposed procedure of using ecosystem services as analysis structure, an important step towards an integrative assessment of ecological and economic properties of biological invasions has been made.

Zusammenfassung

Die Evaluierung ökonomischer Folgen invasiver Arten im Rahmen von Entscheidungsprozessen ist Inhalt der vorliegenden Arbeit, da in den letzten Jahren eine verstärkte Nachfrage von Seiten politischer Entscheidungsträger nach ökonomischen Bewertungsstudien zu beobachten gewesen ist. Allerdings sind ökonomische Bewertungen in diesem Bereich nicht unproblematisch. Dies liegt vor allem an der Komplexität des zu bewerteten Gutes. Probleme treten hier zum einen hinsichtlich der Ökologie auf, da meist große Unsicherheit in Bezug auf die dem Invasionsprozess zugrunde liegenden ökologischen Mechanismen vorliegt. Zusätzlich ist das Wissen über mögliche Bekämpfungsstrategien und deren Wirksamkeit begrenzt. Zum anderen treten die grundsätzlichen methodischen Probleme auf, die mit jeder ökonomischen Bewertung von Umweltgütern verbunden sind.

Aus diesen Gründen besteht dringender Bedarf nach einem holistischen Ansatz, der eine fundierte Bewertung invasiver Arten ermöglicht und politikrelevante Ergebnisse liefert.

Es ist das Hauptziel dieser Arbeit hierzu einen Beitrag zu leisten, indem sie konzeptionell aufzeigt, wo die methodischen Schwierigkeiten liegen und wie mit diesen umgegangen werden kann. Dazu erfolgte eine umfassende Analyse der Problematik biologischer Invasionen, die sowohl die ökologischen als auch ökonomischen Besonderheiten berücksichtigt. Im Gegensatz zu bisherigen Studien wurde Unsicherheit systematisch analysiert und ein Umgang mit ihr vorgeschlagen. Dieser interdisziplinäre Ansatz ist von zentraler Bedeutung. Zum einen bestimmt das Wissen über den Invasionsprozess die Auswahl der Managementstrategien, da deren Erfolg vom jeweiligen Invasionsstadium abhängig ist. Zum anderen determiniert der öffentliche Gut-Charakter invasiver Arten welche umweltökonomischen Bewertungstechniken für eine (volkswirtschaftliche) Betrachtung der Invasionsfolgen notwendig sind.

Jede ökonomische Analyse (economic assessment) gliedert sich in zwei Teile:

1. die Erarbeitung des Mengengerüstes (analysis structure) und

2. die nachfolgende ökonomische Bewertung (economic evaluation) der identifizierten Effekte (einer Invasion).

Wie ein in dieser Arbeit durchgeführter Survey ökonomischer Studien zur Thematik invasiver Arten zeigt, liegen die Hauptschwächen vorhandener Studien eher in der unvollständigen Erfassung aller Invasionsfolgen – und damit dem Mengengerüst – als im eigentlichen

Bewertungsschritt, auf den sich die methodischen Arbeiten bisher vornehmlich konzentrieren. Da das Mengengerüst die Basis jeder ökonomischen Analyse darstellt und entsprechend die Qualität der gesamten Analyse bestimmt, wurde der Fokus dieser Arbeit auf die Weiterentwicklung des Mengengerüsts gelegt. Das Ziel ist es, so eine systematische Erfassung aller relevanten Folgen invasiver Arten zu ermöglichen, d.h. auch eine Betrachtung indirekter bzw. schwer quantifizierbarer Effekte aufzuzeigen.

Bei der Erarbeitung eines konsistenten Mengengerüsts wurden zwei Hauptprobleme identifiziert: (i) die Schwierigkeit die große Vielzahl von Effekten invasiver Arten zu erfassen, im Folgenden als „scope"-Problem bezeichnet und (ii) die Einbeziehung von Unsicherheit, die in Bezug auf das begrenzte Wissen zu Invasionsfolgen und der Wirksamkeit von Managementstrategien herrscht, im Folgenden „uncertainty"-Problem genannt.

Als Lösungsansatz der oben genannten Probleme schlägt die Arbeit vor, das Konzept der Ökosystemdienstleistungen (ecosystem services), wie vom Ecosystem Millennium Assessment verwendet, als Mengengerüst zu benutzen. Dieses Mengengerüst garantiert zum einen eine systematische Analyse der Effekte invasiver Arten, da jeder Effekt durch eine Veränderung der jeweiligen Ökosystemdienstleistung dargestellt werden kann. Zum anderen bietet es eine standardisierte Basis für alle ökonomischen Analysen. Die festgelegte Anzahl von Ökosystemdienstleistungen, die alle innerhalb der Analyse betrachtet werden müssen, lassen die Ergebnisse später besser einordnen. Damit wird ein entscheidender Betrag zum „scope"-Problem geliefert.

Um das „uncertainty"-Problem adäquat zu adressieren, wird auf das Konzept von Faber et al. (1992) aufgebaut. Neben Risiko und Unsicherheit im engeren Sinne verweist es noch auf komplette Unwissenheit (ignorance). Gerade letzte Form von Unsicherheit ist bei Invasionen von großer Bedeutung, da in vielen Phasen des Invasionsprozesses weder die potentiellen Effekte noch die Wahrscheinlichkeiten für deren Eintreten bekannt sind. Nur wenn alle Typen von Unsicherheit erkannt sind, kann adäquat mit ihnen umgegangen werden. Eine Strukturierung von vorhandenem Wissen bezüglich der Veränderungen aller Ökosystemdienstleistungen, zeigt nicht nur welche Effekte sicher bezüglich ihres Eintretens bzw. des Ausmaßes sind, sondern auch welche wahrscheinlich (Risiko) oder unsicher (Unsicherheit im engeren Sinne) sind. Dieses Vorgehen bietet den Vorteil Unsicherheit schon bei der Erfassung der Invasionsfolgen explizit zu machen und unterscheidet sich damit von Kosten Nutzen Analysen (KNA), wo Unsicherheit lediglich nach dem Bewertungsschritt mittels Sensitivitätsanalysen einbezogen wird. Unsicherheit bereits zu diesem frühen Stadium der Analyse explizit zu machen, ermöglicht dem Entscheidungsträger sie von Anfang an in den Entscheidungsprozess einzubeziehen. Ein vergleichbarer Ansatz mit Unsicherheit umzugehen ist in der wissenschaftlichen Literatur bisher nicht diskutiert worden. Er schließt damit eine wesentliche Lücke in der Bewertungsproblematik invasiver Arten.

Neben der Möglichkeit einen Beitrag sowohl zum „scope"- als auch zum „uncertainty"-Problem zu leisten, zeigt die Arbeit, dass die Verwendung des Konzepts der Ökosystemdienstleistungen einen weiteren Vorteil bietet: die Stakeholder-Analyse. So lässt sich entlang der jeweiligen Ökosystemdienstleistungen eine differenzierte Stakeholderanalyse durchführen, bei der für jede Dienstleistung analysiert wird, welche Gruppen von einer Veränderung besonders betroffen sind und welche als sogenannte Drivers auftreten. Diese Art der ökosystemdienstleistungsbasierten Analyse liefert politikrelevante Ergebnisse, da Triebkräfte der Invasion identifiziert und Managementvorschläge erarbeitet werden können. Die durchgeführte Fallstudie zur Ambrosie (Ambrosia artemisiifolia) zeigt, wie wichtig eine Zusammenarbeit aller Stakeholder ist, da Drivers meist über andere Managementansätze verfügen als Betroffene. Erst das Zusammenspiel aller Möglichkeiten verspricht ein erfolgreiches Management invasiver Arten. Diese Information ist ebenfalls von Bedeutung für politischen Entscheidungsträger.

Die oben genannten Punkte stellen eine wichtige Weiterentwicklung ökonomischer Analysen biologischer Invasionen aus dreierlei Gründen dar:

- Es werden Ergebnisse geliefert, die bereits während der Erfassung der Effekte mittels des Mengengerüsts für den politischen Entscheidungsträger relevant sind. D.h. die ohnehin diffizile Monetarisierung ist nicht mehr notwendiger Schritt, um politikrelevante Ergebnisse zu liefern.
- Der Rahmen bisheriger ökonomischer Analysen wird entscheidend erweitert. Die KNA als Standartmethode konzentriert sich auf quantifizierbare Effekte für welche Wahrscheinlichkeiten vorliegen. Aufgrund der hohen Unsicherheit bezüglich der Effekte biologischer Invasionen und der damit verbundenen Vernachlässigung des Großteils von Folgen, greift diese Art der ökonomischen Analyse zu kurz. Mit Hilfe des vorgeschlagenen Prozedere können auch Effekte in der Analyse betrachtet werden, die unsicher, jedoch relevant für den Entscheidungsträger sind. Dieser weiß nun auf welcher Basis eine Entscheidung gefällt wird, d.h. er kann abschätzen, wie groß der Anteil unsicherer Effekte ist im Vergleich zur Anzahl vollständig erfasster Effekte.

- Durch die Stakeholder-Analyse können Triebkräfte des Problems identifiziert werden, die bei einer KNA nicht von Belang sind, da sich diese Methode auf Gewinner und Verlierer eines Projektes (hier Managementmaßnahme) konzentriert.

Diese Punkte zeigen, dass die KNA zwar Bestandteil ökonomischer Analysen invasiver Arten sein kann, von einer ausschließlichen Verwendung sollte jedoch abgesehen werden, da sie der Komplexität des Problems nicht gerecht werden kann.

In dieser Arbeit wird ein Vorgehen aufgezeigt, wie mit Hilfe des Konzeptes der Ökosystemdienstleistungen ein holistischer Ansatz für ökonomische Analysen von biologischen Invasionen erreicht werden kann. Die Arbeit leistet damit einen fundierten Beitrag zur Weiterentwicklung ökonomischer Analysen, um politikrelevante Ergebnisse zu produzieren, die eine adäquate Entscheidungsunterstützung beim Management invasiver Arten gewährleisten.

9 Summary

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