On Sustainable Use of Renewable Resources in Protected Areas as an Instrument of Biodiversity Conservation: A Bioeconomic Analysis

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Hualien, Taiwan, im Mai 2002

Abstract

The objective of this dissertation is to provide a theoretical framework for answering the question, whether and under which biological and socio-economic conditions the sustainable use of wild species in or around protected areas is an adequate strategy for biodiversity conservation. To do this, the dynamic interaction between the use of wild species, management of protected areas, population levels of the utilized species and poaching is investigated. A nonlinear bioeconomic model with two state variables (resource stock, management capital stock) and two control variables (harvest rate, investment rate) is developed on the basis of the traditional bioeconomic model and optimal control theory. The model identifies eight fundamental factors that influence the equilibrium population levels of the utilized species. In sum, the lower the discount rate, the depreciation rate of management capital, the poaching coefficient and the cost coefficient of investment are, and the higher the intrinsic growth rate of species, the non-consumptive value coefficient of species, the efficiency coefficient of management capital and the gross profit coefficient of harvest are, the higher the equilibrium population levels of harvested species will be.

At the theoretical level, our model suggests that, apart from the usually discussed intrinsic growth rate, discount rate and price/harvest cost ratio, more factors should be taken into account when considering the impact of harvest on resource stock. Moreover, contrary to the conclusion of the Clark model and to the popular belief, our model demonstrates that, other things being equal, the higher the gross profit coefficient of harvest (termed as price/harvest cost ratio in the context of the Clark model) is, the greater the equilibrium resource stock will be. This conclusion is consistent with that drawn by the Swanson model, which first considered the important

role of management in resource harvest problem. However, compared to the Swanson model, our model provides a more deliberate and extensive modeling for investigating the resource harvest and management problem.

According to the model results, the eight parameters can be evaluated as indicators for assessing the feasibility of a sustainable use project as a conservation strategy before or when it is applied in specific sites. An assessment procedure is then developed and applied for the case study with reference to several conservation programs in the A-Li-Shan area of Taiwan. The results of the assessment procedure are in principle consistent with what happened in the reality. The case study shows that, some factors that are newly introduced in our model, namely the non-consumptive value coefficient of species, the efficiency coefficient of management capital and the cost coefficient of investment explain the differences of the performance of various conservation programs in the area concerned.

Contents

Chapter 1
Introduction

1.1 The problem concerned and the objective of dissertation

The concept ´biodiversity´ refers to the variety and variability within living organisms and the ecological complexes in which they occur. This term encompasses the diversity of life at all levels of organization, ranging from the gene, organism, and species levels to the community and ecosystem levels. Biologists usually define biodiversity in terms of gene, species and ecosystem diversity, and believe generally that it is an extremely critical factor of ecosystem health and ecological stabilization of the earth (Wilson, 1992).

Due to habitat destruction, overexploitation and poaching, the loss of biodiversity may constitute currently one of the most serious environmental problems for human beings, and has attracted widespread public concern. Based on extrapolations of measured and predicted rates of habitat destruction, and estimates of species richness in various habitats, some evaluations about the loss of biodiversity suggest that a possible loss of between 15 and 50 percent of the worlds total species will occur over the next century, if currently measured trends of habitat loss persist (Wilson, 1992). All these estimates about current and future extinction rates should be interpreted with very considerable caution, because they involve high degree of uncertainty. Nonetheless, it is hard to doubt that human beings is inducing mass loss of biodiversity. What such kind of mass extinction means precisely for the welfare of human beings is, to great extent, still uncertain. But it is certain that human beings will take much more risk of losing the life-support system of the earth than before, and human society is suffering considerable loss in economic value, including use and non-use value, of biodiversity from mass extinction. Conservation initiatives are therefore required to be precautionary enough to prevent further mass loss of biodiversity.

To protect biodiversity, one of the most critical approaches is establishing legally or privately designated protected areas. As a form of environmental regulation, the maintenance of protected areas is even, from the point of view of many conservation biologists, the only effective instrument for conserving biodiversity. Up to the year 1996, 30350 protected areas are known to have been designated worldwide, covering 8.83 per cent of total land area of the earth (IUCN, 1998). Many assessments indicate that in those parts of the world that have established protected area networks, some degree of success in preserving certain proportion, if not majority of the biodiversity in a country has been achieved. However, it has become increasingly evident that the identification, selection, establishment and management of protected areas are worldwide involved in many problems that need to be solved.

Many problems are threatening the survival of the existing protected area networks throughout the world. First, many protected areas are too small or too fragmented to effectively maintain the minimum viable population of some species in the long run. This problem will become increasingly evident as habitats outside protected areas become more and more degraded. Secondly, most protected areas have been acquired and created on a haphazard, but not scientific basis, depending on the availability of fund and land, because that socio-economic and political factors, but not ecological factors, are often the most important considerations in the establishing and siting of protected areas. This leads to the unbalanced representative of various ecosystems in protected areas at all levels, and raises a number of concerns about the ability of existing protected area networks alone to protect biodiversity adequately (Primack, 1998).

Moreover, besides adequate identification and selection, effective biodiversity conservation requires also adequate management of protected areas, since many factors with reference to management issues are threatening the biodiversity and ecological health of protected areas. A list of major threats faced protected areas include logging, mining, cattle grazing, poaching, cultivation, introduction of exotic species, excessive tourism, pollution, corruption of park staff and insufficient funding for management. To great extent, most of these threats have to do with the interest conflicts between protected areas and local residents living in or near protected areas. Usually, after protected areas are established, local communities are precluded from exploiting natural resources they need, as they traditionally have practiced. In many cases, this has resulted in confrontation between local communities and park authorities, illegal exploitation of resources in protected areas, and sometimes leads to refusal of local residents to establish new protected areas or to expand existing protected areas. In the long run, protected areas can survive only when they are supported, or at least tolerated by local communities. And unless local communities can benefit from protected areas, there will be no long-term incentive to support the existence of protected areas. This may be the most serious problem which existing protected area networks are faced. In addition, as a result of the prevailing insufficient funding for protected areas and corruption of park staff, some conservationists question also the ability and the willingness of central governments to conduct effectively the traditional top-down preservation approach (or the so called U.S. national park model) followed by most of the protected areas around the world. This query holds especially for the developing countries.

In sum, the present protected area networks need to be adjusted and expanded on a scientific basis to include a more complete pallet of various ecosystems and thus to safeguard most of biodiversity in the long run (MacKinnon, 1997). The present inadequate management practice of many existing protected areas needs to be improved (Brandon, 1997). All these aims requires the support of the interest groups, whatever they are local communities, private organizations or national governments, which bear the cost derived from the existence of protected areas. It follows that the traditional ´fence and police´ policy of protected areas, which emphasizes the strict protection of habitats but easily results in the hostility of affected interest groups toward protected areas, may be insufficient to reach the previous aims and should be reconsidered. An alternative approach, which enables people to benefit from the maintenance of protected areas in a sustainable manner without substantially harming biodiversity, must be found to supplement the strict preservation approach.

In recent years, many conservationists and scholars have promoted an incentive-oriented approach, namely, that people are allowed to use wild renewable resources in protected areas or in buffer zones around protected areas. In some cases, local communities are also authorized to management natural resources and human activities in protected areas. This alternative strategy is often called the sustainable use approach. Due to self-interest, it is expected that more protected areas, whether existing or new, will be accepted or even designated actively by people under such an approach (IUCN/UNEP/WWF, 1980, 1991). Numerous initiatives have been implemented around the world, and many relevant researches have been conducted to investigate the results of the sustainable use approach and their implications for both general conservation policy and specific protected area policy. However, most of the researches about this topic are based on case studies and relative few works have been conducted in a theoretical and rigorous way, especially in the way of economic rationale.

The objective and major task of this dissertation is, based on rigorous modeling, to investigate the dynamic relation between use of wild renewable resources, management of protected areas and biodiversity conservation, and thereby to afford a general framework for answering the question, whether and under which biological and socio-economic conditions the sustainable use strategy of wild renewable resources in and around protected areas is an appropriate instrument for biodiversity conservation. We intend that the general analysis in this dissertation may hold for all cases in both developing and developed countries. This may help to build a solid scientific basis for rethinking and modification of the current conservation policy.

1.2 Methods

In this dissertation the dynamic relation between use of wild renewable resources, management of protected areas and biodiversity conservation and the relevant policy implications will be investigated by the development of bioeconomic models and by the application of the optimal control theory. To do this, a simple bioeconomic model with one state and two control variables will be firstly constructed on the basis of the traditional bioeconomic model, namely, the Clark model (Clark, 1973, 1976). Afterward, the simple bioeconomic model will be extended and thereby a more complex model with two state and two control variables can be developed. Finally, a general model, which represents a generalized version of the previous extended model, will be completed. The necessary conditions for optimum will be derived. The uniqueness and stability properties of the steady state solution of the models, the relevant phase diagram analysis, and the comparative static analysis will also be presented.

In the general model, in which the interaction between control and state variables is so complex that phase diagrams can not be obtained through the analytic method, a numerical method with computer simulation will be applied to draw the relevant phase diagrams. With the assistance of numerical method, the final outcomes of the variables of the model under different scenarios will also be demonstrated.

In addition to the theoretical models, two empirical case studies based on the experiences from the conservation practice in Taiwan are investigated. The first one refers to the national park system of Taiwan which represents a typical top-down preservation approach following the U.S. national park model. The other one studies the Shan-Mei Community Conservation Project which emphasizes the sustainable use of renewable resources and represents an example of the bottom-up approach, or the so called community-based conservation (CBC) (Western and Wright, 1994). A comparison of these two case studies affords critical, though incomplete, evidences for assessing the relative performance of the two fundamental conservation approaches under specific socio-economic conditions in Taiwan. The ability of the theoretical models to predict possible outcomes of a sustainable use conservation project will be examined simultaneously through the deliberate investigation into the Shan-Mei Community Conservation Project and the other community-based conservation projects in the A-Li-Shan area in Taiwan. The findings of the case studies could be applied to the countries or regions which have similar conditions like Taiwan.

1.3 Contents of the dissertation

Following this introductory chapter, chapter 2 explains first some of the key concepts of biodiversity. It then provides background material on the issues about the measurement methods and some conceptual indicators for biodiversity. Finally, it chronicles the present state of biodiversity and its loss rate according to the results of some scientific assessments.

In chapter 3, the definition and classification of protected areas will be first introduced. Thereafter, the nature of protected areas as an instrument of biodiversity conservation will be explored from both biological and economic aspects. Thus we will review a few economic theories regarding nature and biodiversity conservation and discuss the economic rationale justifying protected areas as an instrument of biodiversity conservation.

Chapter 4 describes the limited success and present problems of existing protected area networks. While extensive efforts have been successful at preserving some types of habitat, two major problems of present networks can be identified, namely, insufficient representative and inadequate management. It follows that the traditional preservation approach of the protected areas alone may be insufficient to safeguard biodiversity, and conservation communities may need to consider the sustainable use approach which is assumed to possess the potential to solve these problems simultaneously. The definition of sustainability regarding the use of wild renewable resources will be addressed. The debate between the ´preservation approach´ and ´sustainable use approach´ will also be discussed. As an example, the case study with reference to the national park system in Taiwan will be investigated to evaluate its performance and to address its problems.

As an introduction into the theoretical modeling, chapter 5 reviews briefly three important economic models regarding biodiversity loss, including the Gordon model (Gordon, 1954), the Clark model (1973, 1976) and the Swanson model (1994). This is followed by a discussion about the policy implications of these models. This review affords a direction for the modeling in the later chapters.

In chapter 6, a bioeconomic model with on state variable (resource stock) and two control variables (harvest rate, management effort) will be first constructed on the basis of Clark´s bioeconomic model and optimal control theory under the assumption that people are allowed to use renewable resources in or around protected areas. Thereafter, the simple model of chapter 6 will be extended and thereby a more complex model with two state variables (resource stock, management capital) and two control variables (harvest rate, investment rate) can be developed in chapter 7 to investigate deliberately the dynamic development process of resource stock, management capital, harvest and poaching activity. The necessary conditions for optimum are derived. The uniqueness and stability properties of the steady state solution of the models and the comparative static analysis will be demonstrated. The policy implications of the models will also be addressed.

Then, a general model which represents the generalized version of the previous extended model will be developed in chapter 8. A computer simulation of the model is conducted through the use of numerical method. The results of the computer simulation of the general model may help offer more arguments for judgement of the current conservation policies.

Chapter 9 gives an example of how the community-based conservation, an important variant of the sustainable use approach, can work well under specific biological and socio-economic conditions. With the application of the analysis framework afforded by the theoretical models, we investigate deliberately the Danayiku Nature Park at Shan-Mei, Taiwan. The analysis shows us, which conditions contribute to the success of the Danayiku Nature Park at Shan-Mei, and which result in the failure of some other similar community-based conservation projects in the A-Li-Shan area of Taiwan. It follows a discussion about the policy implications of the findings from the case study.

The concluding chapter 10 synthesizes the findings and shortcomings of our theoretical models, and tries to offer some suggestions for modification of current conservation policy and some possibilities for future research.

Chapter 2
Biodiversity: Concept and its loss rate

2.1 Definition of biodiversity

The term biodiversity, a special terminology representing biological diversity, is used to describe the variety and variability within living organisms and the ecological complexes in which they occur. It encompasses all species of plants, animals, microorganisms and the ecosystems and ecological processes of which they parts. The emerging concern about biodiversity reflects the empirically based recognition of the fundamental interconnections within and among these various levels of ecological organizations, and the general belief that biodiversity is an extremely critical factor of ecosystem health and ecological stabilization of the earth (Wilson, 1992). To quantify the measurement of biodiversity and thereby to facilitate the management of biodiversity, it is necessary to disentangle some of the separate elements of which biodiversity is composed. Biologists usually define biodiversity in terms of genes, species and ecosystems (WCMC, 1992).

Genetic diversity refers to the range of variation within and between populations of organisms, or more precisely, it is the sum of genetic information contained in the genes of individuals of plants, animals and microorganisms. Ultimately, this again resides in variations in the sequence of the four base-pairs that, as components of nucleic acids, constitute the genetic code (WCMC, 1992). Wilson (1992) estimated that there are about 10^{17} different genes distributed across the world´s biota, refraining from entering into differences within organisms of any given species. Considering the fact that each species is made up of many organisms, the total number of different genes will be then far more. For example, the worldwide about 10,000 ant species have been estimated to comprise 10^{15} living individuals at each moment of time (Wilson, 1988). However, each of the estimated different genes does not make an identical contribution to overall genetic diversity because of different functions of various genes (WCMC, 1992). Moreover, we do not know even the number of existing species, and respectively the number of existing individuals within a given species. No practical tools up to now are available to evaluate these factors. Given these problems, it seems that genetic diversity is not yet applicable to the evaluation of both biodiversity loss and conservation programs.

Species diversity refers to the number and variety of species. A species is generally defined as populations within which gene flow occurs under natural conditions, although this definition may not work well for some species (Brown, 1993). Because the definition about what a species is differs considerably between various groups of organisms, species cannot be recognized and enumerated by biologists with perfect precision. Moreover, a straightforward count of the number of species affords only a partial indication of biological diversity, since species make different contributions to overall diversity, depending on the extent to which they differ from each other. Generally, the more different a species is from any other species, the greater its contribution to overall biological diversity. Furthermore, different species play different ecological roles, and thereby have different effects on community structure and overall biodiversity. For example, a keystone species whose activities govern the well-being of many other species apparently makes a greater contribution to the maintenance of biodiversity than a species on which no or only few species wholly depend (WCMC, 1992).

It is evident that the number of species in different taxonomic groups at a site, or the so called ´species richness´, is not a perfect indicator for biodiversity and even for species diversity. However, probably because of the lack of knowledge and the difficulty with quantifying biodiversity at genes and ecosystems levels, and the fact that species are the primary focus of evolutionary mechanisms, biodiversity has in practice been presented primarily in terms of species richness, although we do not know the true number of species existing on earth as well, even to the nearest order of magnitude. Roughly 1.4 million species of all kinds of organisms have been formally described. Approximately 57,000 are vertebrates, 250,000 are vascular plants and bryophytes, and 750,000 are insects. The remainder include a complex array of invertebrates, fungi, algae, and microorganisms (Wilson, 1992). Wilson (1988) estimated that there are totally 5 to 30 million species on earth. Some biologists, such as Terry Erwin (1988), have put forward even higher estimates, up to 50 million. Of the different taxonomic groups, plants and vertebrates, as well as a few other groups such as butterflies, are relatively well known. For poorly studied fungi and microorganisms, the estimates of overall species numbers are probably inadequate (Wilson, 1992). For example, it has been estimated that as many as 1.5 million species of fungi may actually exist, with 69,000 known species (Hawksworth, 1991). One survey of the marine ecosystems estimated that the total unexplored new species could well reach upwards to 10 million (WRI, 1994).

Ecosystem diversity is defined as the variety of habitats, biotic communities and ecological processes in the ecosystems. While it is possible to define what is in principle meant by genetic and species diversity, there is no unique definition and classification of ecosystems at the global level, because ecosystems differ from genes and species in that they explicitly include abiotic components, being determined by the physical environment, such as climatic, edaphic and topographic condition. Even though various weightings can be ascribed to these different factors when estimating the diversity of particular areas, there is no one definitive index for measuring ecosystem diversity. The quantitative measurement of diversity at the ecosystem, habitat or community levels remains therefore problematic (WCMC, 1992).

2.2 Measurement of biodiversity and some indicators

To make the concept biodiversity operational and reduced to measurable quantities, some measures of biodiversity have been suggested by scientists. While biodiversity is very commonly used as a synonym of species richness, the number of species alone can be highly misleading as a measure of biodiversity, since it fails to consider the different facets of biodiversity, as discussed in the previous section. During recent attempts to discuss biodiversity, scientists have developed some concepts and methods to measure biodiversity more precisely, in the sense that, instead of a straightforward counting of numbers of species, these measurements consider explicitly the extent to which species differ from other species.

Based on the ´genetic distance´ data originating from DNA-DNA hybridization method, scientists try to develop measures that reflect precisely those characteristics that define the difference between various biological units, whether they are genetic material, sub-species, communities or ecosystems (Weitzman, 1992; Eiswerth and Haney, 1992; Solow et al., 1993). The genetic distance data represent differences between the DNA of various species, and hence provide information about differences at the genetic level. Furthermore, these data afford an indication for higher-taxon diversity as well, because the genetic distance between species that belong to different, higher taxa tends to be greater. Weitzman (1992) applied the criterion of genetic distance between species to develop a measure of biodiversity, which simultaneously considered the probability of the extinction of species. Solow et al. (1993) constructed a similar measure. However, in addition to the genetic distances between species, the factor of species richness is also considered by them. The measure introduced by Eiswerth and Haney (1992) is also based on genetic distance and species richness, but, unlike Weitzman and Solow et al. do, it does not take the factor of the probability of the extinction of species into account.

These methods discussed here are scientifically consistent measures of biodiversity, and they could in principle be used to assess conservation priorities at any level. In practice, however, these measures require substantial information from the biological sciences in large-scale problems. This limit their usefulness to a great extent. At most, they may be useful for extremely small-scale problems under given data base. In addition, while these measures emphasize the genetic properties and the endangered status of species, a critical facet of species, namely their ecological role in supporting the functioning and resilience of ecosystems is neglected.

At the species level, two measures, namely species richness and species diversity, are usually applied in practice. Species richness, an important dimension of biodiversity, refers to the number of species existing in an area. Species diversity indices are derived by weighting species by some measure of their importance, such their abundance, productivity, or size (Orians, 1994).

At ecosystem level, as discussed in the previous section, ecosystem diversity is very difficult to define and measure. Given the complexities of the numerous components of ecosystem diversity, some measures of ecosystem level diversity are introduced. Based on significant differences in flora, fauna, vegetation structure, and physical attributes such as climate, Udvardy (1975) developed a system of biogeographic analysis for terrestrial ecosystems. He divided the world into eight terrestrial biogeographical regions, and these eight regions are further subdivided into 193 provinces which may be very useful for assessing the effectiveness of the protected area network in protecting various ecosystems. Similarly, to establish priorities to conserve the most important areas, two policy oriented methods, the ´ecological hotspots´ (Myers, 1988) and the ´mega-diversity countries´ (Mittermeier & Werner, 1990) are developed, by the use of lists of plant species or other taxa to identify biologically rich biogeographic areas or countries.

Because measuring and monitoring all facets of biodiversity are very difficult, conservation biologists have proposed some indicators at species level as a shortcut whereby attention is focused on one or a few species to monitor and solve biodiversity conservation problems. These indicators which can be easily monitored includes the so called umbrella species, flagship species, keystone species and biodiversity indicator (Simberloff, 1998; Caro & O´Doherty, 1999). The concept of umbrella species, defined as a species that requires a large range of habitat so that protecting it will automatically protect many other species, have been applied to depict the type of habitat or size of the area for protection, though significant ignorance about how many other species could be saved under the protection umbrella of the target species still exists. Tiger is a well-known example for umbrella species which played a critical role in the designation of protected areas. The flagship species, usually a charismatic large vertebrate such as the giant panda, have been used to attract public concern and thereby promote conservation campaign. The keystone species, such as elephant, plays a pivotal role in the ecosystem and its activities have great impacts on the well-being of many other species, so that protecting keystone species contributes also greatly to the conservation of many other species and the maintenance of the health of the ecosystem. It is notable that an umbrella species is not necessarily an adequate flagship or keystone species, and vice versa. Therefore, management regimes of two indicator species can conflict. Furthermore, intensive management of an indicator species does not necessarily imply successful conservation of the rest of the communities to be indicated or protected, since they do not receive similar treatment (Simberloff, 1998). In particular, as Caro and O´Doherty (1999) suggested, these indicators are not necessarily adequate biodiversity indicator species. In addition to the concept of umbrella species, flagship species and keystone species, biologists usually apply the biodiversity indicator, namely the number of species in a well-known taxonomic group as an indicator for the number of the species in poorly-known taxonomic groups, for assessing the overall status of biodiversity in a given area. Once the biodiversity indicator species have been identified, their absence can be used as a sensitive indicator for the absence of other species in the same area. By the application of this concept, areas with high biodiversity could be identified more easily and then designated for protection.

The previous discussion apparently suggests that no single measure or indicator can capture all facets of the complex concept biodiversity. It follows that, rather than attempting to develop an universal indicator for biodiversity, a system of multiple indicators which assesses different facets of biodiversity may be a more practical solution to the measurement problem of biodiversity under given ignorance about relevant scientific knowledge. Reid et al. (1993) developed a set of 22 indicators for biodiversity which are used to assess the diversity of wild species and genetic diversity, the diversity at the community/habitat level, and the diversity of the domesticated species. Such an indicator system may help, though not perfect precisely, capture the full view of biodiversity status.

2.3 The loss rate of biodiversity

In this section, we briefly review the current assessments with reference to the loss of global biodiversity which has resulted in urgent concern about biodiversity conservation. Probably because of the lack of knowledge and the difficulty with quantifying biodiversity at genes and ecosystem levels, the problem of biodiversity loss has in practice been presented primarily in terms of species loss, although we do not know the true number of species existing on earth as well, even to the nearest order of magnitude. And it is evident that loss of other dimensions of biodiversity, though difficult to quantify in the manner of a universal indicator, may be greater still (Ehrlich and Daily, 1993).

Current estimates with reference to the loss of biodiversity taking forms of extinction of species are mainly based on the ecological relationship between area and number of species. The fundamental relationship between the size of an area and the number of species it supports, is an empirical generalization of the theory of island biogeography first developed by MacArthur and Wilson (MacArthur and Wilson, 1967). Originated from the observations using island data, the theory states that the size of an area and of its species number tend to have a predictable relationship, depending on various types of ecosystems. It implies that fewer species are able to exist in a number of small habitat fragments, like the islands in a sea of human-dominated landscapes, than in the original unfragmented habitat, and this can result in the extinction of species. This relationship is commonly presented in the following functional form:

$$S = cA^z$$

where S denotes the equilibrium number of species that should persist in a given habitat area, A represents the size of the area, and c and z are constants whose values depend on habitat type. Both c and z are positive constants, and most of the estimates with reference to the theory of island biogeography gave a value of z between 0.20 and 0.35 in many groups of organisms (Meffe and Carroll, 1994). This suggests that we can predict the reduction in numbers of species as the area of habitat decreases, if some estimates about c and z have been made.

The deforestation of tropical forests is commonly considered as a major cause of global biodiversity loss, since tropical forests support the majority of terrestrial species. As a result, based on extrapolations of measured and predicted rates of habitat destruction in tropical forests, and estimates of species richness in various habitats, some estimates of current rates of global species extinction have been undertaken by applying the theory of island biogeography. These estimates suggest that a possible loss of between 15 and 50 percent of the world´s total species will occur over the 21st century, if currently measured trends of habitat loss persist (see Table 2.1). Wilson (1988) argues that, due to the current destruction of tropical rain forests and setting aside from the moment extinction due to the destruction of other habitats, both the per-species rate and absolute loss in number of species would be about 1, 000 to 10,000 times the historic rate of extinction. Current extinction rates thus appear to be far higher than the so called ´natural´ or ´background´ rates.

Table 2.1. Estimates of the current rates of species extinction

Estimate of species loss	Basis	Source
15-20% by year 2000	Forest area loss	Lovejoy (1980)
50% by year 2000	Forest area loss	Ehrlich & Ehrlich (1981)
33% in 21st century	Forest area loss	Simberloff (1986)
25% in 21st century	Forest area loss	Raven (1988)
5-15% by year 2020	Forest area loss	Reid & Miller (1989)
0.2-0.3% per year	Forest area loss	Ehrlich & Wilson (1991)
2-8% by year 2015	Forest area loss	Reid (1992)

Source: WCMC (1992) and references.

Even on the best available present knowledge, these estimates involve high degree of uncertainty, because of the ignorance of the total number of species and their distributions, the patterns of habitat loss, and the effects of deforestation on species (Myers, 1994). Moreover, a straightforward count of the number of extinct species only provides a partial indication of biodiversity loss, since species that differ widely from each other in some respect by definition contribute more to overall diversity than those which are very similar, and the different ecological importance of various species could have a direct effect on community structure, and thus on overall biodiversity (WCMC, 1992). Besides, some scientists assert that estimations of species loss based on extrapolations of deforestation and on the theory of island biogeography are misleading, because these estimations fail to take the possible significant amount of biodiversity after deforestation into account (Lugo et al., 1993).

Assumptions and estimations with reference to the rate and extent of habitat loss have also raised the uncertainty when estimating the loss rate of species. As Table 1 shows, relatively new estimates suggested somewhat more conservative calculations of the rate of species loss. This may partly reflect the fact that, after a period of rapid deforestation in tropical forests, the pace of forest clearance have slowed down in the early 1990s. However, the pace of forest destruction in amazonian forests has again accelerated significantly in recent years (Laurance, et al., 2000). Therefore, these conservative estimations might be modified in the future. In any case, all estimates about current and future extinction rates should be interpreted with very considerable caution under given high degree of uncertainty. Nonetheless, these estimates appear to provide a useful approximation of the degree of threat to the global biodiversity during this period. It is hard to doubt that human being is inducing mass loss of biodiversity.

Chapter 3
The nature of protected areas as an instrument of biodiversity conservation

Protected areas are without doubt the focus of the current conservation activities. As one of the most important instruments of biodiversity conservation, it deserves our research into the question why the existence of protected areas can be justified in our crowded planet. In this chapter, we will explore the nature of protected areas from both biological and economic aspects. Some important economic theories regarding nature and biodiversity preservation will be briefly reviewed to explore the economic rationale justifying protected areas as an instrument of biodiversity conservation.

3.1 Definition and classification of protected areas: The IUCN system

One of the most critical approaches in protecting biological communities is establishing legally or privately designated protected areas. Based on the agreement at the Fourth World Congress on National Parks and Protected Areas, The World Conservation Union (IUCN) (IUCN, 1994) defined a protected area as ´*An area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means*´. Once land comes under protection, decisions must be made regarding how much human disturbance will be allowed. In order to be able to categorize protected areas, IUCN (1994) has developed a system of classification for protected areas that ranges from minimal to intensive allowed use of the habitat by humans. The following categories are arranged in ascending order of human use permitted in the area:

- Ia. Strict nature reserve: protected areas managed primarily for preserving representative examples of biological diversity for scientific study, education, environmental monitoring, and maintenance of genetic variation.
- Ib. Wilderness area: large areas of unmodified or slightly modified wilderness managed primarily for recreation, for subsistence economic activities, and for protection of natural process.

II. National park: large protected areas with outstanding scenic beauty and ecological importance, designed primarily to preserve unique natural beauty and resources, and maintained for scientific, educational and recreational purpose. Extractive use of resources in national parks is in principle not allowed.

III. Natural monument: smaller protected areas designed primarily for preservation of unique natural areas with specific natural or cultural significance.

IV. Habitat/species management area: strict nature reserves which require active management intervention, managed primarily for maintenance of the characteristics of the community. Controlled harvesting is allowed in some cases.

V. Protected landscape/seascape: protected areas managed mainly for the maintenance of areas in which people and the environment interact in a harmonious way. They include natural areas that have undergone considerable human transformation. Nondestructive use of resources is permitted.

VI. Managed resource protected areas: protected areas managed primarily for sustainable use of natural resources, in a manner that ensures the long-term protection and maintenance of biodiversity.

The categorization is based on the primary management objective of protected areas. Of these categories, the first six can be considered as true protected areas, with the objective managed mainly for protection of biodiversity. The main management objective of areas in category VI is not protection of biodiversity, but they can play an important role in conserving biodiversity, since they are usually much larger than strict protected areas, since they still contain many or even most of their original biological diversity, and since strictly protected areas are often surrounded by managed resource protected areas (primack, 1998). In addition, protected areas are usually managed for multiple objectives. According to the IUCN classification system and the priority assigned to relevant management objectives, a categorization with more detailed management objectives is made by Phillips and Harrison (1999), as Table 3.1 demonstrates.

Table 3.1. Potential primary management objectives of various type of protected areas

Objectives	Ia	Ib	II	III	IV	V	VI
Scientific research	1	3	2	2	2	2	3
Wilderness protection	2	1	2	3	3	NA	2
Preserve species and genetic diversity	1	2	1	1	1	2	1
Maintain environmental services	2	1	1	NA	1	2	1
Protection of natural/ cultural features	NA	NA	2	1	3	1	3
Tourism and recreation	NA	2	1	1	3	1	3
Education	NA	NA	2	2	2	2	3
Sustainable use of natural ecosystems	NA	3	3	NA	2	2	1
Maintain cultural/ traditional attributes	NA	NA	NA	NA	NA	1	2

1 = Primary objective
2 = Secondary objective
3 = Acceptable objective
NA = Objective not applicable
Source: Phillips and Harrison (1999), p. 15.

In the field of conservation biology, two approaches have been widely used to conserve biodiversity. The maintenance of protected areas is a part of so called *in situ* or *on-site* preservation, which leaves biological communities and populations in the wild, whereas the *ex situ* or *off-site* approach involves permanent collections of species in zoos, botanical gardens and the preservation of seeds and other genetic material in a controlled environment such as germplasm banks (Primack, 1998). It is generally agreed that in situ approach is the most effective way, even the single way in the long run to preserve biodiversity (Primack, 1998), because we do not have enough resources or knowledge to maintain the majority of the world´s species in captivity. As Woodruff (1989) pointed out, in situ preservation of biodiversity is far more cost-effective than ex situ preservation, although the latter has an important role to play when in situ approach fail. Moreover, only in natural communities are populations large enough to conserve relatively complete heritable base, and only within natural communities are species able to function adequately as a part of the complex ecosystems and continue the process of evolutionary adaptation to the changing environment.

Thus, for the sake of conserving biodiversity, the in situ approach, namely leaving biological communities in the wild, is extremely necessary from the biological point of view. However, of the de facto existing undisturbed or relatively undisturbed wilderness, why should we give some certain areas special protected status, i.e., declaring them as strictly protected areas and leaving them out of almost all development considerations? What is the economic rationale justifying strictly protected areas as an instrument of biodiversity conservation? To answer these questions, we may turn to several economic theories dealing with nature and biodiversity preservation.

3.2 Economic theories justifying the existence of protected areas: theories with reference to nonrivalry and nonexcludability

From the point of view of economics there are primarily four special characteristics associated with wilderness and biodiversity, i.e., nonrivalry, nonexcludability, uncertainty and irreversibility. With reference to nonrivalry and nonexcludability, Sherman (1989) showed that the main reason for the degradation of natural areas is that there is an underlying disparity between the private and social costs and benefits of wilderness use and conservation. Much of the benefits associated with wilderness exhibit nonrivalry and nonexcludability, such as the existence value derived from simply knowing that a certain wilderness area or a certain species exists, even though people will never truly see or use it (Krutilla, 1967). The problem with nonrival goods is that the market cannot set an efficient price for them. When goods are nonexcludable, there are problems of externalities. This mix of public goods and externalities problems results in significant market failures, and these market failures make it much more difficult for people to appropriate the benefits of protecting wilderness. As a result of market failures, there is a bias toward conversion and development of wilderness. The effect of this bias is that a smaller amount of areas is protected or left in natural state than would be the case if there was a full accounting of all the social benefits and costs associated with each alternative land use.

According to the theory of island biogeography, this result implies generally that a smaller amount of biodiversity is left in natural state than would be the case if market failures would not exist. To bridge this gap between suboptimal and optimal provision of biodiversity, additional provision of protected areas will be necessary. This requires usually government intervention, although private non-government organizations play as well an important role in this task.[1] In any case, the establishment of strictly protected areas is in this context used as an instrument to counterbalance the effects of market failures and helps increase the amount of remaining biodiversity to an amount much closer to the socially optimal allocation.

3.3 Economic theories with reference to uncertainty and irreversibility: option value, quasi-option value and the Safe Minimum Standard

The effect of uncertain and irreversible decisions compared to certain and reversible ones in the environmental field have long been addressed in the economic literature. Dealing with these two properties, several approaches have been suggested. One possible approach of handling uncertainty and irreversibility is to explicitly introduce the concept of ´option value´, which is defined as the value, in addition to expected consumer´s surplus from actually using a good, that arises from retaining an option to a good for which demand and/or supply is uncertain (Weisbrod, 1964; Bishop, 1982). For example, when addressing the issue about whether converting a piece of rain forests into a farm, the option value of this piece of rain forests will be that consumers are willing to pay more than the expected consumer´s surplus derived from converting rain forests into a farm, so that they can ensure that they can make use of this piece of rain forests later on, given that their tastes in the future are uncertain. In this context, option value could be interpreted as a risk aversion premium (Bishop, 1982).

Dealing with the demand side uncertainty, i.e., consumers are uncertain about their future demand, Schmalensee (1972) and Henry (1974) argued that the net option value may be positive, negative or zero for a risk averse individual, because of the fact that preservation, as well as development, can bring risks. While the demand-side option value is indeterminate, Bishop (1982) maintained that supply-side option value, i.e., option value associated with uncertainty about whether a good will be available when consumers want to use it, would unambiguously be positive for risk-averse individuals, if both the utility function and income are certain. However, Freeman (1985) and Johansson (1988) asserted that, under certain assumptions, the sign of the supply-side option value is indeterminate. Finally, in any case, option value depends on the attitude of people toward risk. In sum, it seems that the existence of option value is not a definitive argument for the preservation

[1] For example, through privately funded efforts to conserve biodiversity in the USA and elsewhere, The Nature Conservancy has established many protected areas and made a substantial contribution to biodiversity conservation (Grove, 1988; Primack, 1998).

of biodiversity and wilderness, especially in many developing countries, in which the future income level is highly uncertain.

In the context of an irreversible development decision where information about the future consequences of development would be available with time, Arrow and Fisher (1974) developed the concept of the ´quasi-option value´. When an irreversible development is undertaken, future alternatives become limited. As a result, there is a value in delaying a decision that involves irreversible effects whose values are not known. This value can be defined as the gain from being able to learn about future benefits that would be precluded by development by delaying an irreversible decision (Fisher and Krutilla, 1985), or as a conditional value of information, conditional on a particular choice of first-period development, i.e., that the development is postponed initially (Fisher and Hanemann, 1987). As Fisher and Hanemann showed, so long as there is a non-zero probability of such information becoming available, a positive quasi-option value always exists for the alternative of avoiding an irreversible development.

In the context of nature and biodiversity preservation, quasi-option value refers very specifically to the value that biological resources and wilderness have as resources of information that is not yet discovered. Just like usual preservation benefits, quasi-option value is, at least partly, nonrival and nonexcludable. The owner of a piece of land will hardly take quasi-option value into consideration when developing the land, and this fact leads to market failures and underprovision of wilderness and biodiversity. In this connection, the maintenance of strictly protected areas could be again interpreted as an instrument for correcting market failure bias.

An alternative approach suggested to dealing with uncertainty and irreversibility is known as the Safe Minimum Standard (SMS) (Ciriacy-Wantrup, 1952; Bishop, 1978). In essence, the SMS approach applies a modified version of the ´minimax´ criterion in game theory. In its strict sense, applying the minimax criterion involves choosing the alternative that minimizes the maximum possible losses that will arise when making the wrong decision. The SMS approach which uses the modified version of the minimax criterion also suggests minimizing the maximum possible loss, but only when the costs of doing so are not unacceptably high (Bishop, 1978).

The SMS approach is especially applicable for the cases in which resources can be irreversibly depleted. Many biological resources, such as plant and animal species, have a ´minimal viable population´. These resources are renewable, if their population levels are greater than those of the minimal viable population. However, they are subject to irreversible depletion, if their populations go below the minimal viable populations. The species in question may have little known value today but its future value may be significant, e.g., as a gene and food source, or functioning as an important keystone species which has a great influence on many other species within the same ecosystem. The irreversible depletion of these resources may cause enormous future social or economic losses. Using the SMS approach implies that such alternative involving irreversible potential loss should be avoided unless the costs of doing so are unacceptably large. Therefore, in the context of biodiversity conservation, the maintenance of strictly protected areas can be viewed as setting physical safe minimum standards to safeguard biodiversity, at least partly, and thereby to prevent from potential enormous loss in the future.

3.4 The Perrings and Pearce Model

In a paper addressing conservation of biodiversity and the relevant policy instrument issues, Perrings and Pearce (1994) provided a similar, but more detailed biological and economic rationale for the use of physical standards. They asserted that the problem of biodiversity loss is especially associated with the ecological threshold effects. The erosion of biodiversity is a process with the special characteristic of irreversibility. If the loss of biodiversity goes on, certain ecological threshold will sooner or later be reached, at which ecosystems are on the edge of losing their ability of self-organization and their ability as the life support system of the earth. In this context the ecological threshold can be defined as the critical values for populations of organisms or biogeochemical cycles. Once, for example, some populations of organisms are already on the edge of threshold, a marginal depletion of these populations will eventually result in the collapse of ecosystems and enormous costs for human beings. As Perrings and Pearce showed, the existence of the ecological threshold effects has important implication for the choice of appropriate instruments dealing with biodiversity conservation.

Let us demonstrate this at hand of a simple model. For convenience of comparison, we consider first the normal case with no threshold effects. Defining w as a strictly positive vector of market input costs, r a non-negative vector of biological resources, and q output. The total economic cost of exploiting biological resources includes usually the components of private cost and external cost:

$$TEC = C + E$$

where $C = C(w, q)$ denotes private cost function which is assumed to be continuous, differentiable and increasing in both w and q, and $E = E(r(q))$

represents external cost function which is assumed to be increasing in q. The external cost function can be neither differentiable nor continuous, as we will address later. Again, defining $R = R(p, q)$ as the private revenue function under given price p of the output q and Π_p as the marginal net private profit (private revenue minus private cost), then it is evident that the necessary condition for privately optimal output level is

$$\frac{\partial R}{\partial q} = \frac{\partial C}{\partial q}$$

and the necessary condition for socially optimal output level is

$$\frac{\partial \Pi_p}{\partial q} = \frac{\partial E}{\partial r}\frac{\partial r}{\partial q}.$$

These two conditions are satisfied at the output levels q_p^* and q_s^* respectively, as figure 3.1 shows. In this case, the privately optimal output level will diverge from the socially optimal output level, suppose that there is no environmental regulation.

Now let us consider the case with ecological threshold effects to show how the introduction of threshold effects change the nature of the problem. As defined previously, the existence of ecological threshold effects means that, once ecosystems or populations are depleted beyond these thresholds, the life support system will eventually collapse and result in enormous costs. This accordingly implies, as Perrings and Pearce argued, the discontinuity or at least the non-smoothness of the external cost function. Such a case is demonstrated in figure 3.2 in which the external cost function E is discontinuous at certain output level, and is thought to increase dramatically at that point to certain much higher level, and then go on to increase with the output level. It is obvious that, at the discontinuity point of the external cost function where $\frac{\partial E}{\partial q} < \frac{\partial \Pi_p}{\partial q}$, the first order condition for social optimum will not hold. In this case, the socially optimal output level will be q_s^* in figure 3.2, at which the threshold and the corresponding discontinuity of the external cost function happen.

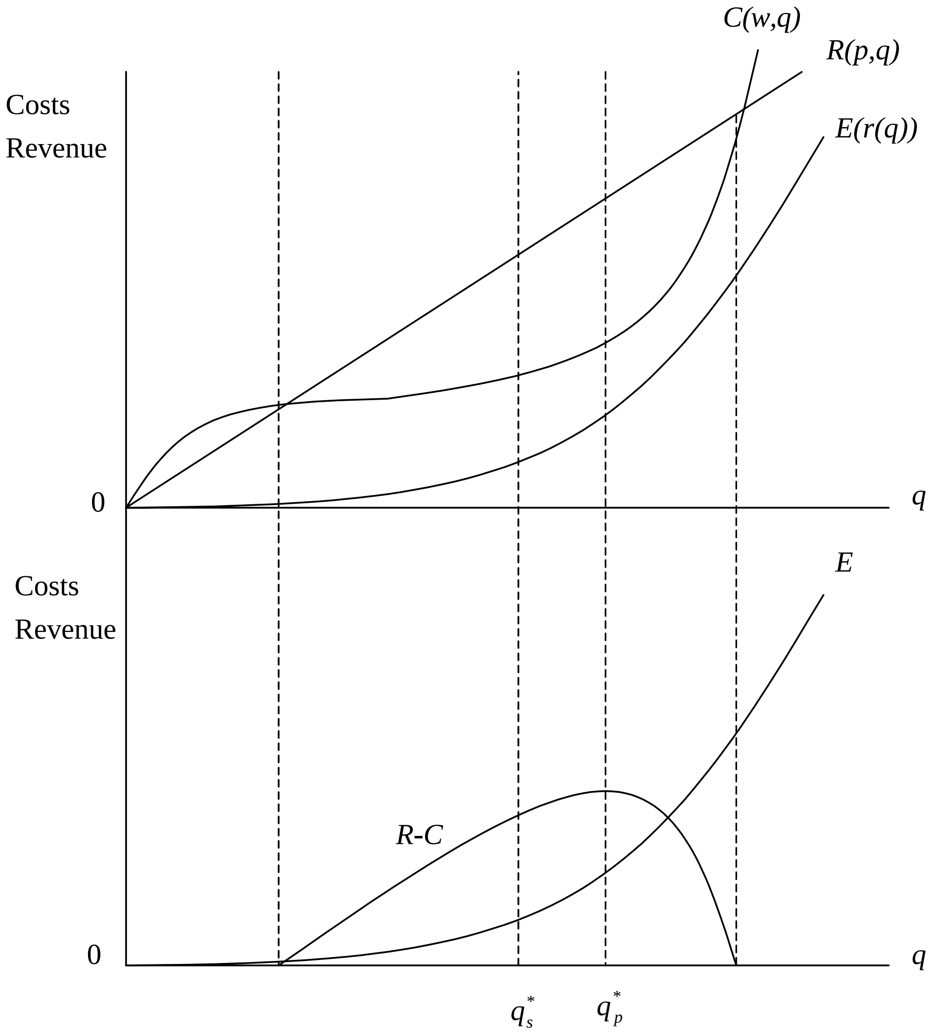


Figure 3.1. Private and social optima: continuous external cost function.
Source: Perrings and Pearce (1994), p.16.

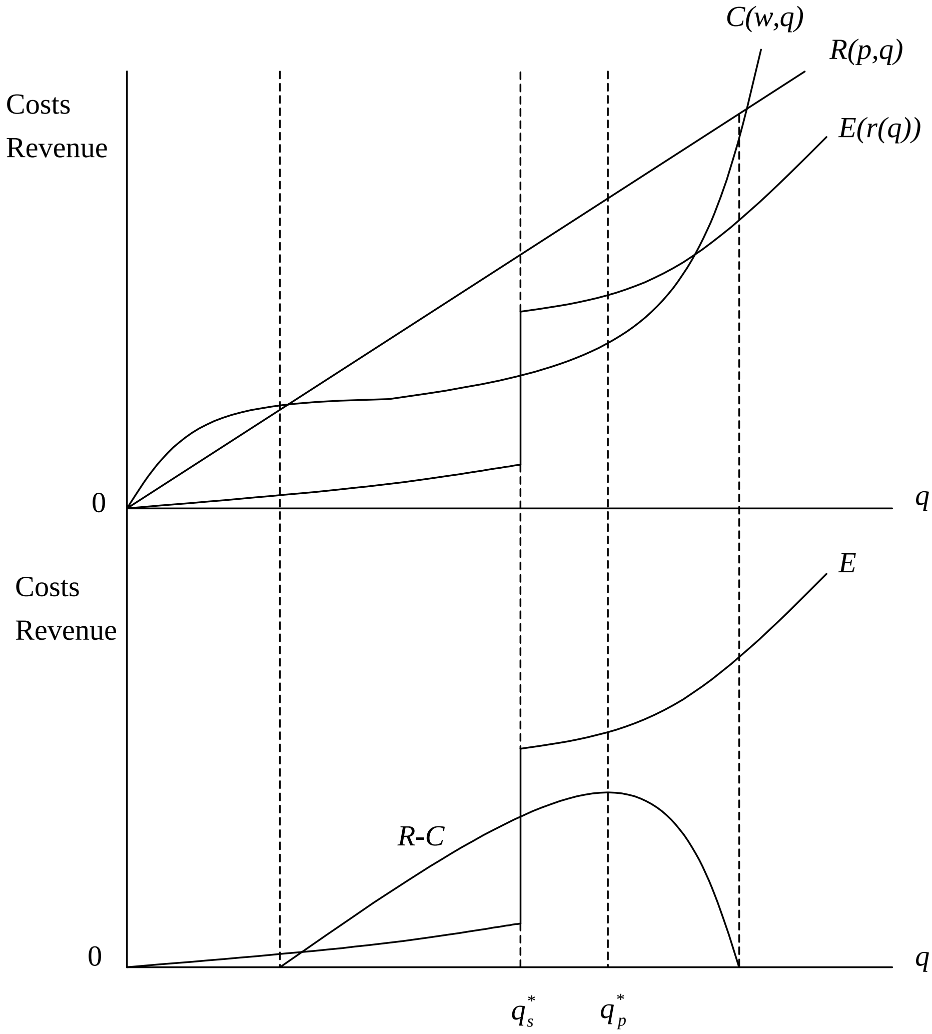


Figure 3.2. Private and social optima: discontinuous external cost function. Source: Perrings and Pearce (1994), p.17.

From the policy perspective, it is necessary to consider the possible policy instruments which can protect society against the costs of exceeding the threshold. One approach widely suggested is setting physical restrictions around thresholds, just like the rationale of the safe minimum standards. Nonetheless, instead of the quantitative restriction the safe minimum standards approach implies, Perrings and Pearce suggested an alternative instrument based on price incentive - special environmental levies or fines which are extremely severe. Their idea is as follows. If the social costs derived from threshold effects are higher than the maximum private net benefit derived from exceeding the thresholds, an arbitrary penalty between the two can be introduced and enforced to guarantee that the output level will not exceed q_s^*, the socially optimal output level, as figure 3.3 shows. The line 0abc in figure 3.3, which denotes the private cost function derived from charges and penalties, includes two components: environmental charges for the expected social costs derived from the use of biological resources below the threshold level of economic activity (0a), and a severe penalty (ab) when exceeding the threshold level of economic activity. The penalty must be high enough, at least equal to the maximum private net benefit of using biological resources, so that the private output level will not exceed the threshold value as a result of the self-interest motive. In addition, considering the factor of uncertainty in measuring private benefit, a penalty much higher than the maximum private net benefit can offer an additional margin for safety.

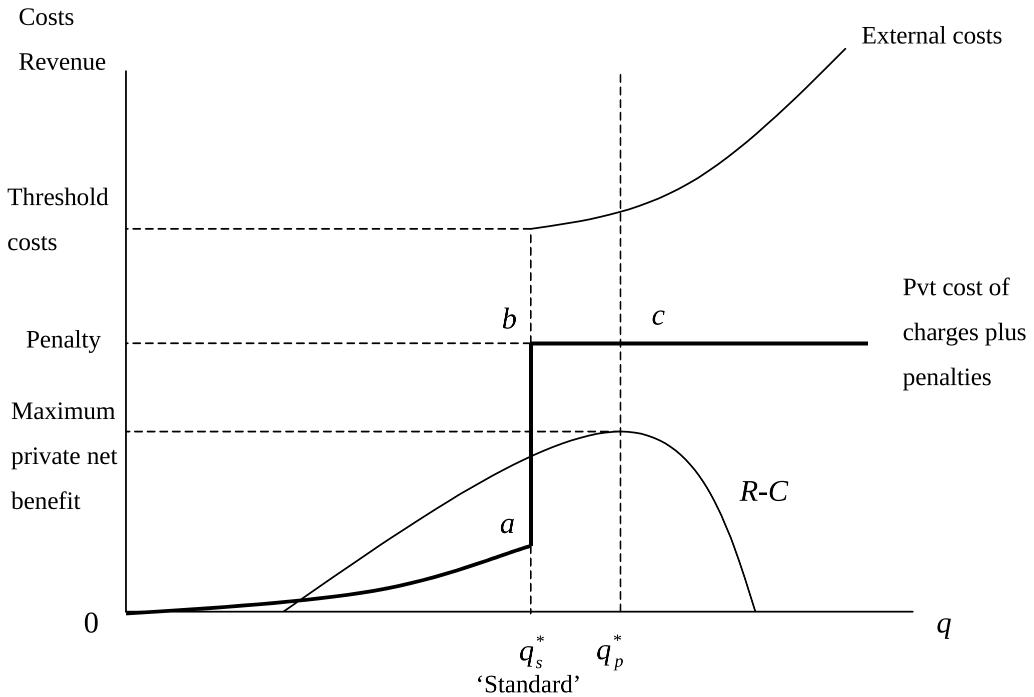


Figure 3.3. Penalty function: thresholds known with certainty. Source: Perrings and Pearce (1994), p.24.

The case considered above implicitly assumes that the ecological threshold level of output is known with certainty. Under the premise that the penalty can be effectively enforced, it can be guaranteed that the output level will not exceed that of the social optimum. However, the uncertainty as a result of the ignorance about ecological threshold values of biological resources usually still exists. In this case, even a severe environmental penalty can not guarantee that output level will not exceed the threshold level. What we can do is setting standards, on which the penalty function is based, conservatively relative to the thresholds which are assumed to be or are designed to protect, as figure 3.4 demonstrates in which the penalty function 0abc lies on the left side of the discontinuity at q_s^*. In this way the risk of exceeding ecological thresholds can be reduced.

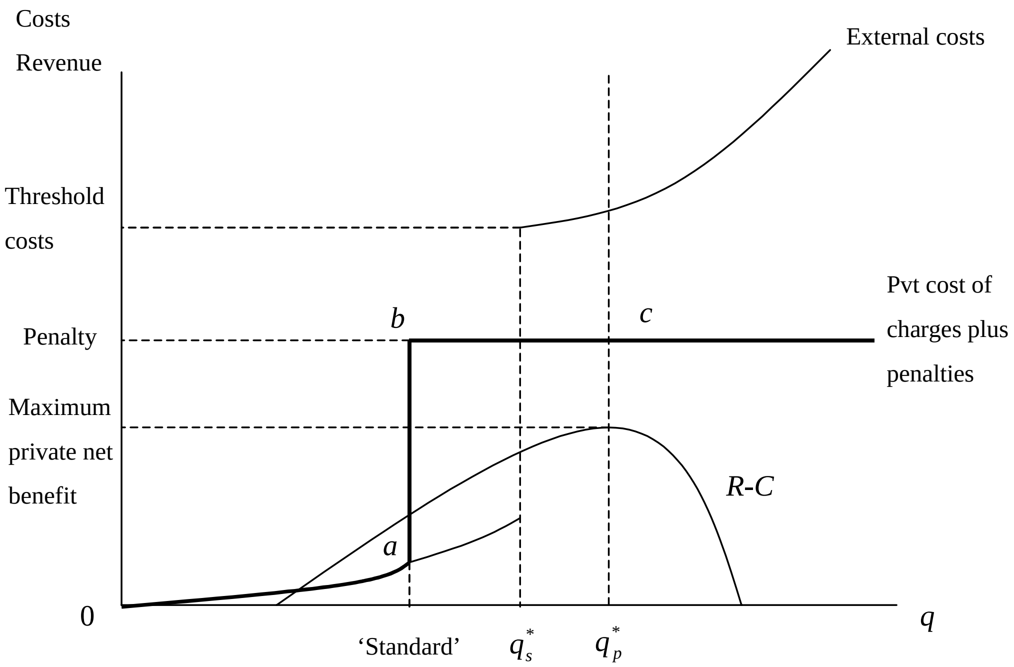


Figure 3.4. Penalty function: thresholds not known with certainty. Source: Perrings and Pearce (1994), p.25.

In any case, as Perrings and Pearce concluded, a judgement with reference to the socially acceptable margin of safety in the exploitation of biological resources is required, whether the policy instrument is based on the physical restrictions (safe minimum standards) or on the economic incentive (environmental charges and penalties) associated with the enforcement of those physical restrictions. In the context of biodiversity conservation, their argument justified the appeal for the preservation of at least part of biodiversity, and accordingly justified the existence of strictly protected areas as physical standard for safeguarding biodiversity.

3.5 Concluding remarks

As a critical instrument of biodiversity conservation, the existence of protected areas can be justified from both the biological and economic perspectives. From the point of view of biology, the designation of protected areas is the most effective way, even the single way in the long run to preserve large area of wilderness and biodiversity. It is also more cost-effective than the ex situ preservation approach. From the point of view of economics, the establishment and maintenance of protected areas can be viewed as an instrument for counterbalancing the effects of market failures and helping increase the amount of protected biodiversity to an amount much closer to the socially optimal biodiversity. In addition, it can be regarded as setting physical standards to safeguard biodiversity and thereby to avoid potential enormous costs in the future resulted from biodiversity loss.

Chapter 4
State of protected areas and the debate on sustainable use of renewable resources in and around protected areas as an instrument of biodiversity conservation

In this chapter we will first briefly review the effectiveness and problems of the present protected area network from a global perspective. The severe problems faced by protected area network aroused the widespread debate within conservation communities on the questions, whether the traditional preservation policy of the protected areas alone is sufficient to safeguard biodiversity, and whether the sustainable use approach could protect biodiversity more effectively than the preservation approach. To explore this crucial topic which has to certain extent dominated the direction of conservation policy at regional, national and international levels, the debate between the preservation approach and the sustainable use approach will be investigated in details. The definition of sustainability regarding the use of wild renewable resources will accordingly be addressed. Finally, as an example of the previous discussion, a case study with reference to the national park system in Taiwan will be investigated to evaluate its performance and to address its problems.

4.1 Effectiveness of protected areas: a global perspective

The extent of the global protected area network has increased steadily in the last several decades. Up to the year 1996, 30,350 protected areas are known to have been designated worldwide, covering a total land area of 13.23 millions square kilometers. This amount represents 8.83% of the total land surface of the world (IUCN, 1998).[2] Of these protected areas, 90% are strictly protected (IUCN categories Ia-V), covering 73% of the protected surface, and 10% are partly strictly protected (IUCN category VI), representing 27% of the protected surface (Green and Paine, 1999). However, substantial variations of the protected percentage can be observed between countries, biomes, biogeographical realms and biogeographical provinces (WCMC, 1992). Moreover, landscape usually encompasses large area of a fairly uniform habitat

[2] It is worth noting that this percentage includes many marine protected areas or protected areas which encompass a marine component.

with only a few small areas of rare habitat types. It seems that, for the sake of conserving biodiversity, including representatives of all the habitats in a system of protected areas will be more important than only preserving large areas of the common habitat type (Primack, 1998). These facts complicate substantially the assessment about the effectiveness of protected area networks to conserving biodiversity. The conclusions that can be drawn from the high level of aggregation at such as the biogeographical realms or biome level is limited. A more accurate picture of biodiversity conservation can only be obtained from an analysis at lower level or even from case studies.

Some investigations have been carried out to assess how effective the limited protected area networks are in maintaining biodiversity. At national level, for example, a survey of 25 mostly temperate countries indicated great variation in the extent to which listed threatened plant species discovered in protected areas, i.e., from 35-40% in Spain to 100% in Czechoslovakia (WCMC, 1992). Siegfried (1989) found that, with only 6% of the land area of the region, the protected areas in southern Africa protect respectively more than 90% of amphibian, reptilian, avian and mammalian species native to the region. Sayer and Stuart (1988) found that, in 11 of the 12 large tropical African countries, the majority (at least 75%, and generally well over 80%) of the native bird species have populations inside protected areas. In Taiwan, 55% of all vascular plant species (2200 of 4000) and 50% of the bird species (200 of 400) have been found to exist within the boundaries of Kenting National Park, which covers only 0.5% of the total land area of Taiwan (COA and DNP, 1992). These figures indicate that in those parts of the world that have established protected area networks, some degree of success in preserving many, if not most of the species in a country has been achieved.[3]

On the other hand, while the number of species existing in protected areas is important as an indicator of the ability of protected areas for biodiversity conservation, the occurrence of species in protected areas is no guarantee of their long-term security. Many protected areas are too small or too fragmented to effectively maintain the minimum viable population of some species in the long run (Primack, 1998). These problems will become increasingly evident as habitats outside protected areas become more and more degraded. As the theory of island biogeography predicts, under plausible assumptions, about half of all species will in the long run inevitably go extinct, if 90% of all habitat area are destroyed, leaving protected areas as

[3]For more examples see the summary of the relevant papers made by Primack (1998), pp. 404-406.

isolated ´biological islands´of natural or semi-natural habitat (Wilson, 1992). Moreover, as Daily and Ehrlich (1995) argued, only considering the aspect of species extinction may greatly underestimate the rate of loss of organic diversity as a whole. They asserted that, to maintain the ecosystem services on which human economic system depends, the conservation goal should be the preservation of a minimum ratio of natural to human-dominated habitat in all habitat types. It supports the point of view that protected area networks should try to ensure the representatives of as many types of ecosystems as possible.

From the perspective of representative, at the biome level, there are substantial differences between the extent to which they are protected. Of the 14 major biomes, the subtropical/temperate rainforest/woodlands are relatively better protected, with 10.29% under protection. In opposition to this, only 0.98% of the temperate grasslands and 1.12% of the lake systems are under protection. Compared with the 10% target planned for the protection of biomes at the Fourth World Parks Congress, almost all biomes remain underpresented in the present protected area networks (Green and Paine, 1999).[4] Dinerstein and Wikramanayake (1996) found that, lowland moist tropical forests are greatly underpresented in terms of coverage by large protected areas in Indo-Pacific region, while most of the large protected forests are lower-mountain or mountain forests. Globally, the marine and coastal regions are poorly represented in the protected area networks and emerge as an obvious gap (McNeely et al., 1990).

In another way, Udvardy (1975) divided the world into eight terrestrial biogeographic regions. These eight regions are further subdivided into 193 provinces. A survey suggested that 50 of the regions have a protected coverage more than 10%, 34 have a protected coverage between 5% and 10%, and the other possess less than 5% of the land under protection or have no protected areas (WCMC, 1992). Studies of protected areas coverage at regional and national levels provide a similar result. The case studies of Sattler (1992) about Queensland, Australia, of Turner et al. (1992) about Canada, of Barnard et al. (1998) about Namibia, and of Powell et al. (2000) about Costa Rica showed that all these countries have not yet achieved an ecologically representative network of protected areas. To reach this goal, for example, Queensland require an addition of 3.6 million hectares of national park. Klubnikin (cited in Scott et al., 1987) concluded that in California, 95% of the alpine habitats are in reserves, but less than 1% of biologically

[4] A detailed list of these biome and their protected percentages see Green and Paine (1999).

rich riverbank is protected. It is here worth noting that, the countries addressed above have established well-managed protected area networks, and have generally good reputation within conservation communities. For most of the countries, it is plausible that the situation could not be better, though rigorous assessments are still required. At least from the point of pure biological view, all above studies indicated that, while extensive efforts have been successful at preserving some types of habitat, many regions are still virtually ignored.

4.2 Current problems of protected areas

Protected areas are the cornerstone of the modern conservation movement. While some degree of success in preserving certain proportion, if not majority of the biodiversity has been achieved in those parts of the world that have established protected area networks, it has become increasingly evident that the identification, selection, establishment and management of protected areas are worldwide involved in many problems that need to be solved.

From the perspective of an ideal model of protected area network, strictly protected areas should be large enough to maintain the minimum viable population of some critical species, selected and created on a rigorously scientific foundation to maintain the representative of various habitat types, connected via a network of corridors to avoid isolation of populations, and surrounded by buffer zones to mitigate the impacts of human activities from the world outside protected areas. Compared with these ideal standards, some problems of the present protected area system emerge. First, as discussed previously, many protected areas are too small or too fragmented to effectively maintain the minimum viable population of some species in the long run. Green and Paine (1999) indicated that 59% of the total protected areas throughout the world are smaller than 1000 hectares, and only 6% are larger than 1000 km^2 in size.

Next, the present protected area system suffers from the problem of unbalanced representative of various ecosystems in protected areas. From this point of view, the protected area networks in almost all parts of the world is apparently inadequate for the protection of a representative proportion of the ecosystem. This bias can trace back to several factors. Historically, the initial purpose of many protected areas was to protect spectacular scenery and provide recreational resources (Dixon and Sherman, 1991). Up to now, many national parks tend to be selected for their aesthetic value and recreational opportunity rather than for their biological richness. Ecological considerations are dominated by scenic and recreational values, probably because individuals can benefit more directly from these values (Krautkraemer, 1995). Moreover, protected areas are established typically in those regions which are unfavorable for farming, settlement and other human activities. Socioeconomic and political factors, not ecological factors, are often the most important considerations in the selection of the sites of protected areas (WCMC, 1992; Sattler, 1992). Thus, most protected areas have been acquired and created on a haphazard, but not scientific basis, depending on the availability of fund and land. In addition, as Shafer (1999) indicated in the example of the US national parks, the absence of biological considerations in the past may partly reflect the fact of scientific ignorance. Only until 1960s, the concept of representative on a systematic basis began to evolve (Eidsvik, 1992). This leads to the unbalanced representative of various ecosystems in protected areas at all levels, and raises a number of concerns about the ability of existing protected area networks alone to protect biodiversity adequately (Primack, 1998).

Moreover, besides adequate identification and selection, effective biodiversity conservation requires as well adequate management of protected areas, since many factors with reference to management issues are threatening the biodiversity and ecological health of protected areas. Almost all these factors are, directly or indirectly, after all human factors, even though in some cases in which seemingly natural factors have caused the problems. For example, the overpopulation of deer and thereby caused overgrazing in many protected areas is caused in fact by human beings, because major predators in those areas were eliminated (Primack, 1998). A list of major threats faced protected areas include logging, mining, cattle grazing, poaching of wildlife, cultivation, introduction of exotic species, excessive tourism, pollution, corruption of park staff and insufficient funding for management (Dixon and Sherman, 1991; Shah, 1995; Primack, 1998). Protected areas in different countries are faced different threats. In developing countries, the major threats to protected areas include logging, cattle grazing, poaching, fire, cultivation, insufficient funding, corruption and excessive tourism (Machlis and Tichnell, 1987). The major threats in developed countries are often mining, exotic species, tourism and pollution (Dixon and Sherman, 1991; Mitchell, 1994).

To great extent, most of these threats have to do with the interest conflicts between protected areas and local residents living in or near protected areas who bear mostly the opportunity cost resulted from the existence of protected areas. In the last several decades, most of the newly established protected areas, especially in developing countries where large area of wilderness still exists, have followed the strict preservation model (or the so called U.S. national park model) which emphasizes keeping ecosystems functions in their natural state and minimizing any possible human interference. To do this, after protected areas are established, local communities are usually precluded from exploiting natural resources they need, as they traditionally have practiced, or they are even forcibly resettled without compensation. In many cases, this has resulted in confrontation between local communities and park authorities, illegal exploitation of resources in protected areas, and sometimes leads to refusal of local residents to establish new protected areas or to expand existing protected areas. This has become increasingly evident especially because the rapidly increasing populations in developing countries need more land and resources to survive. Some authors even accused the preservation model that it is an implicit form of second wave of colonialism (Adams and McShane, 1996). Leaving aside the debate about whether such an accusation is fair or not, the fact is that political pressure increased dramatically in developing countries to change the concept of conservation (Kramer and van Schaik, 1997). Recently, this has partly led to the marked decrease in numbers of the newly established protected areas in the tropics as a whole, and to near zero in some countries of tropical Asia and Africa (Terborgh and van Schaik, 1997). It must be recognized that, in the long run, protected areas can survive only when they are supported, or at least tolerated by local communities. And unless local communities can benefit from protected areas, there will be no long-term incentive to support the existence of protected areas. This may be the most serious problem which existing protected area networks are faced. In addition, as a result of the prevailing insufficient funding for protected areas and corruption of park staff, some conservationists question also the ability and the willingness of central governments to conduct effectively the traditional top-down preservation approach followed by most of the protected areas throughout the world. This query holds especially for the developing countries in which the commitment for conservation has been strongly weakened because of the severe financial reality. Many protected areas in developing countries are in fact the so called ´paper park´ where protected areas are designated but never implemented.

In sum, as a result of the problems of insufficient size and inadequate representative, the present protected area networks need to be adjusted and expanded on a scientific basis to include a more complete pallet of various ecosystems and thereby to maximize the protected biodiversity in the long run (MacKinnon, 1997). To effectively protect biodiversity, the present inadequate management practice of many existing protected areas should be improved (Brandon, 1997). All these require the support of the relevant interest groups which bear the cost derived from the existence of protected areas, whatever they are local communities, private organizations or national governments. It follows that the traditional preservation model of protected areas, which emphasizes the strict protection of habitats but easily results in the hostility of affected interest groups toward protected areas, alone may be insufficient to achieve the previous aims, and the general conservation policy should therefore be reconsidered. An alternative approach, which enables people to benefit from the maintenance of protected areas in a sustainable manner without substantially harming biodiversity, must be found to supplement, or to substitute for the strict preservation approach under some circumstances.

In the last two decades, many conservationists and scholars supported the concept that protected areas should be part of a larger portfolio of sustainable resource use, namely protected area network should include the component of strictly protected core reserves, and the component of areas in which limited resource use is permitted. They promoted accordingly an incentive-oriented approach, namely, that people are allowed to use wild renewable resources in protected areas or in buffer zones around protected areas. In some cases, local communities are also authorized to management natural resources and human activities in protected areas. This alternative strategy is often called the sustainable use approach. Due to self-interest, it is expected that more protected areas, whether existing or new, will be accepted or even designated actively by people under such an approach, if people really benefit from the maintenance of protected areas (IUCN/UNEP/WWF, 1980; 1991). Numerous initiatives have been implemented around the world, and many relevant researches have been conducted to investigate the results of the sustainable use approach and their implications for both general conservation policy and specific protected area policy.

On the other hand, the introduction and application of the sustainable use approach in conservation practice, and the fact that this approach has greatly influenced the conservation policy of the so called mainstream conservation organizations such as Worldwide Fund of Nature (WWF, formerly known as World Wildlife Fund) and World Conservation Union (IUCN) (Kramer and van Schaik, 1997), have induced widespread debate on the question, whether and under which conditions the sustainable use approach is appropriate as an instrument for biodiversity conservation. This is just what we concerned about in this dissertation. Before the details of the debate are discussed, we should turn our attention to the definition of the concept sustainable use of renewable resources.

4.3 Defining ´sustainable use of renewable resources´

The use of renewable resources for material or immaterial purposes is an old tradition of human beings and the basis of human civilization. However, the sustainable use of renewable resources as an instrument of biodiversity conservation is a new concept. To avoid misunderstanding in the discussions throughout the remaining part of this dissertation, it is critical to define precisely the relevant terminologies.

First, the term ´renewable resources´, both in the customary usage of the conservation community and in this dissertation, refers to the wild biological resources. Therefore, some non-biological renewable resources such as water, and biological resources which are domesticated, are not within the scope of our discussion.

Next, the term ´use´ has reference to the different ways in which wild biological resources are utilized by human beings. Usually, it encompasses the consumptive and the non-consumptive use of wild biological resources. The consumptive use, or the extractive use, refers to the direct utilization of parts or products of organisms of biological resources such as meat, skin or wood. It includes often hunting of wildlife, fishing, logging and gathering of plants and of other nontimber products. In the way of the non-consumptive use, no parts or no individual will be taken, directly and intentionally, from their population. Some losses in the population may happen as a result of the indirect impact of human activities, for example in the case of ecotourism, but generally to a minor extent. Both consumptive and non-consumptive use are important issues and are within the scope of this dissertation. However, in the context of biodiversity conservation, we are mainly concerned with the consumptive use, as most members of the conservation community do, because of the easily understood reason that, compared with the non-consumptive use, consumptive use of wild biological resources is generally less compatible (at least seemingly) with conservation of biological resources.

Now let us turn to the definition of sustainability in the context of the biological resources utilization, or more precisely the following question: when is the use of a certain species or a certain population sustainable? The World Conservation Strategy (IUCN/UNEP/WWF, 1980), the well known document which first promoted the sustainable use of renewable resources as an instrument for conservation, defined that the use of wild biological resources is sustainable if their wild populations are not significantly affected.

Under this definition, natural resource stock is viewed as a certain kind of capital, and resource user takes only away the interest derived from the natural capital. The deficiency of this definition is, that any consumptive use will inevitably lower the population of utilized species, but even though the population is greatly affected, it does not necessarily imply that the use is unsustainable.

To correct such a deficiency, we may define alternatively that the use of biological resources is sustainable if the production of resources can balance with their harvest . But this definition possess as well deficiency in that, while harvest can equal production at many different population levels, the population can be reduced to such a low level that local or global extinction could happen, once a natural disaster destroys the remaining population. And even though the population level remains stable, it could lose its ecological role to maintain the essential ecological processes and services, or lose its significance as an useful resources to meet human needs under such a low population level (Bennett and Robinson, 2000a). Therefore, as Bennett and Robinson asserted, the definition of sustainability should, in addition to the resource itself, take wider management goals with reference to resource users and ecosystems into account.

We apply in this dissertation the definition of sustainability suggested by Bennett and Robinson (Bennett and Robinson, 2000b), because their proposal includes more complete criteria which assess the different aspects of sustainability from biological, ecological and economic perspectives. The use of renewable resources is sustainable, if the following criteria can be satisfied:

1. Utilized species cannot show a consistent decline in their populations.
2. The populations of utilized species cannot be reduced to such levels that they are vulnerable to local extinction.
3. The populations of utilized species cannot be reduced to such levels that their ecological roles in the ecosystem is impaired.
4. The populations of utilized species cannot be reduced to such levels that they lose their significance as useful resources to human users.

While criteria 1 and 2 can be applied on an in principle unambiguously scientific basis, it is evident that criteria 3 and 4 are to certain extent ambiguous and therefore can only be applied in a general way. These criteria cannot provide a clear-cut answer to the question about sustainability. However, they are undoubtedly critical foundation for the assessment of sustainability.

4.4 The debate on sustainable use of renewable resources in and around protected areas as an instrument of biodiversity conservation

4.4.1 Background of the debate

During the last two decades, the sustainable use strategy of renewable resources as an instrument of biodiversity conservation has increasingly become an important component of the policy of the mainstream conservation organizations. The IUCN and WWF gave more particular emphasis to the need to increase the number of protected areas in the IUCN´s Categories V, Protected Landscapes/Seascapes, and VI, Managed Resource Protected Areas, in which the sustainable use strategy of renewable resources plays a critical role in conserving biodiversity. They also emphasized the important role of non-governmental organizations, private companies, individuals, local communities and indigenous peoples in managing protected areas. These changes within the conservation organizations´ policy reflect a growing sophistication in their understanding of the relationship between protected areas and the human societies in which they exist. On the other hand, these changes have simultaneously induced one of the most serious controversy regarding conservation issue within conservation community. It involves the following questions: is sustainable use of renewable resources possible and appropriate in and around protected areas? Could such a strategy really help to protect biodiversity more effectively, or just on the contrary, it would degrade biodiversity more rapidly? To answer these questions, two often conflicting approaches dominated the debate.

The traditionally prevailing model of protected areas, which originated from the U.S. National Parks System in the nineteenth century, is often called the preservation approach. This approach asserts that more progress will be made toward maintaining biodiversity on the remaining wilderness of the world by establishment of strictly protected parks and reserves under the management of national government or international organizations (Noss, 1991). By drawing boundaries around specific areas, absolute banning of consumptive use of natural resources and effective legal enforcement, human interference would be minimized and protected areas would be preserved in their natural, at best non-inhabited state. This view is mainly founded on the historical experiences that harvest of natural resources has generally led to serious degradation of biodiversity. Furthermore, it doubts the possibility of success of the sustainable use model under the current insufficient management capacity in many places, and especially under the socio-economic conditions that prevail in developing countries, in which a large proportion of the world´s remaining intact biological community exists (Brandon et al., 1998; Oates, 1999; Madhusudan and Karanth, 2000).

While the preservation approach has, to great extent, successfully safeguarded a small proportion of the remaining wilderness, it potentially antagonized a section of the human community denied access to natural resources, especially the local community. Consequently, poaching, degradation of resources, and local hostility against protected areas and park authorities increased. In many cases, the interest conflicts between local communities and management authorities have prevented existing protected-area networks from enlargement. Under the threat of habitat fragmentation, ecological isolation, poaching and other factors which will greatly degrade the existing protected area networks, some scientists, conservationists and conservation organizations began to question the long-term viability of these isolated biological islands. They argue that the protected-area system should be expanded and, owing to the fact that the preservation model may have reached the limit of its ability, there must be an economic incentive for conservation. In contrast to the protectionist philosophy, they advocate that more wilderness and biodiversity will be conserved by developing sustainable use strategy of renewable resources in those wildlands that are not strictly protected, or in buffer zones around strictly protected areas. The underlying philosophy is that people will protect what they receive value from, especially when they receive tangible financial benefits (McNeely, 1988; Western and Wright, 1994; Freese, 1998).

To help realize the diverse dimensions of this controversy, we summarize the fundamental arguments for and against the two positions of the debate in the following subsections.

4.4.2 Perspectives of the sustainable use approach

The primary concern of the proponents of sustainable use approach is, that the existing protected areas are too small or too fragmented so that, as the theory of island biogeography predicts, they may prove incapable of maintaining most of the biodiversity in the long run. Even in the existing protected areas, scientists have shown that the maintenance of protected areas cannot guarantee the long-term survival of all, or most of the wild species initially present. Extinctions still occur in protected areas, also in strictly protected reserves, largely as a result of inadequate area (Lovejoy et al., 1986). To mitigate biodiversity loss derived from habitat degradation and fragmentation, there are two possible ways. The first one is to create much larger strictly protected areas. This is hardly realistic under current circumstances because of the associated high economic and social costs. Another option suggests that, through the creation of buffer zones around strictly protected areas in which sustainable use of renewable resources is allowed, the total protection effect would be similar to outright expansion of strictly protected areas but without, or with less economic and social costs. And what the most important is, that regulated, sustainable use strategy will provide important economic incentives for local people to accept and maintain the existence of protected areas (Shaw, 1991). If local communities which bear part of the costs derived from the maintenance of protected areas never benefit from protected areas, they will not support the expansion of existing protected area network, unless we can accept the undemocratic way usually used in the past when creating new protected areas, especially in developing countries.

A stronger position is asserted by the proponents of the so called ´use-it-or-lose-it´ strategy. As David Western, the former Director of Kenya Wildlife Service (KWS), and R. M. Wright indicated, that the most important conservation issue is how to deal with the vast majority of the earth´s land surface where are not protected by parks and the interests of local communities prevail (Western and Wright, 1994). If conservation agencies do not encourage local communities to conserve biodiversity by sustainable use of natural resources in natural and seminatural ecosystems, the remaining wilderness will sooner or later be converted into human-dominated land and therefore lose the biodiversity on it, since wilderness as a land use pattern financially cannot compete with other profitable land use options such as monocrop agriculture. Instead of focusing on strictly protected areas as the preservation approach traditionally does, this position tends to emphasize the importance of conservation actions outside of these areas. Once natural resources and their sustainable use has become integrated into people´s way of life, the areas with rich biodiversity will be well protected at minimal costs and biological resources will have a habitat larger than the existing protected areas. Janzen (1994) asserted that the use-it-or-lose-it approach can probably protect 80-90 percent of tropical terrestrial biodiversity on a land surface of 5-15 percent of the tropics, in sharp contrast with the result of the conserved 10-30 percent of biodiversity on 1-2 percent of the lands when the preservation approach still dominates the conservation policy. Furthermore, sustainable resource use may potentially support conservation in the way that harvesting on one site may relieve pressures to harvest on other sites with higher conservation priority (Freese, 1998). It is here worth noting that many conservationists support the sustainable use approach, not just because of the moral or public relations´ considerations, but because they believe that it is the best strategy under prevailing conservation constraints, especially in developing countries.

While any use of biological resources has inevitably direct impacts on their populations and involves a loss of biodiversity (Robinson, 1993), some proponents of the sustainable use approach argued that, through the creation of economic incentive and thereby enlarged habitats and better protection, there are many cases in which the sustainable use of wild species has in fact supported the conservation of the target species, such as the well-known examples of white-tailed deer and snow goose (Medellín, 1999). These two species have today much higher levels of total population than those of the period before sustainable use strategy was applied. Moreover, ecosystem conservation has in many cases benefited from specific sustainable use programs. For example, the waterfowl management and fostering programs in North America has greatly contributed to the conservation of wetland and biological resources living there (Medellín, 1999). Similar examples can be found in many southern African countries where sustainable use of wildlife resources comprises an even better form of land use than other options. And this has greatly promoted wildlife conservation and habitat protection simultaneously in those countries (Prins et al., 2000).

In addition to the pragmatic considerations discussed above, some people support the idea of sustainable use as a result of the moral considerations of justice and human rights. As Adams and McShane (1996, P.xviii) claimed *´conservation has long operated on the comfortable belief that Africa is a paradise to be defended, even against the people who have lived there for thousands of years´*. Strictly protected areas have come under criticism for the fact that, after protected areas are created, local people are usually precluded from utilizing natural resources on which their livelihood depends. They ignored the potential benefit of sustainable resource use for local people. Sometimes traditional preservation approach is accused of being a style of imperialistic (Gadgil, 1992) or colonial operation (Adams and McShane, 1996). Pimbert and Pretty (1997), and Ghimire and Pimbert (1997) claimed that the main reason of the failure of the protected areas is that western norms and practice of conservation science ignores the fact that local communities should be in a better position to assess what is good for them than western scientists. Some people stake out an extreme position that all protected areas should be open to some kind of use (Wood, 1995; Ghimire and

Pimbert, 1997). In short, this view tries to achieve the goal of ´win-win´, namely, both people and nature are winners. These considerations about justice and human rights have especially reference to the issue of indigenous peoples´ self-determination and territorial control.

Indigenous peoples currently inhabit and claim a land area of between 20% and 30% of the earth´s surface, in which most of the wilderness, and most of the existing and potentially new protected areas of the world are embedded. This is as large as four to six times more territory than is included in the strictly protected areas of the earth. And as recent judicial decisions in many countries showed, indigenous peoples may reassert ownership of significantly larger land than they have at present (Stevens, 1997). The land use patterns on this large territory are of great importance from either the point of view of human rights or of biodiversity conservation. Some conservationists asserted that, from the perspective of justice and human rights, sustainable resource use by indigenous peoples should be allowed, even in strictly protected areas. Moreover, through sustainable use of natural resources and co-management of protected areas in which indigenous peoples share resource ownership and responsibility, conservation will be more likely effective, because, apart from the effect that protected areas will be accepted more easily, indigenous peoples have possessed necessary ecological knowledge, and developed social and cultural mechanisms that regulate resource use to sustainable levels, or at least mitigate its impact on the environment to the acceptable extent. Conservation actions might be conducted more cost-effective by indigenous people in a decentralized way than the traditionally state-centric approach. This optimistic attitude toward indigenous peoples´ role in conservation is often called the modern version of ´the ecologically noble savage´ (Redford, 1991).

4.4.3 The Community-Based Conservation (CBC)

Conservation thoughts and policies have changed to great extent in the 1980s in response to the call that local communities should be involved in conservation rather than being ignored, and that more attention should be paid to the positive roles of sustainable resource use and economic incentives in conservation. This has led to the popularity of a special form of the sustainable use strategy, the Community-Based Conservation (CBC). The idea is so popular and so widely accepted that it sometimes becomes the synonym of sustainable use approach in developing countries, though, not only at community level, sustainable use approach includes in fact a wider range of conservation initiatives at other levels. As a result of its importance in conservation practice in developing countries, it is here worthy of briefly investigating the idea of community-based conservation.

In the broadest sense, Western and Wright (1994, p.7) maintained that *´community-based conservation includes natural resources or biodiversity protection by, for and with the local community´*. Adams and Hulme (2001, p.13) defined the community-based conservation as *´those principles and practices that argue that conservation goals should be pursued by strategies that emphasize the role of local residents in decision-making about natural resources´*. In practice, conservation goals can be pursued by community-based conservation in three ways: (1) permitting local residents living in or near protected areas to participate in resource use and management policy; (2) transferring ownership or user rights over natural resources to local residents; and (3) benefiting local community through conservation (Hackel, 1999). This includes a wide range of various conservation initiatives with different titles, such as community wildlife management, collaborative management, community-based natural resource management, and integrated conservation and development programs[5] (Adams and Hulme, 2001).

According to the criteria of objectives, ownership/tenure status and management characteristics, Barrow and Murphree (2001) identified three primary types of community conservation approach:

1. Protected Area Outreach: The primary objective of this approach is conserving ecosystems and biodiversity. The land and natural resources of protected areas are owned legally by state, and all decision-making with reference to the management of protected areas are made by state alone. However, it seeks to share the benefits derived from protected areas, such as entrance fee, with local community, and thereby resolve the conflicts between protected areas and local communities in a mutually agreeable manner. Extractive use of renewable resources is in principle not allowed. Local communities play a passive participatory role in management practice of protected areas.

2. Collaborative Management (or Co-management): In addition to the primary objective of conserving ecosystems and biodiversity, this approach pursues also some local livelihood benefits through sustainable

[5] Integrated conservation and development program (ICDP) refers to the social and economic development program that offers local community alternatives to natural resource use (Brandon and Wells, 1992). It usually focuses on the improvement of infrastructure and seldom involves the utilization of renewable resources.

use of natural resources. The land and natural resources of protected areas are owned and managed legally by state, but some user rights and management responsibilities of certain resources are devolved upon local communities to achieve conservation as well as livelihood improvement objectives. Controlled extractive use of renewable resources which is agreed by conservation authorities and local communities is allowed. Local communities share certain management responsibilities of protected areas and of natural resources with conservation authorities and thereby play a more active role than in protected area outreach, but conservation authorities still play a predominant role in deciding management policy.

3. Community-Based Conservation: The primary objective of this approach is sustainable management and use of natural resources by local communities. Through devolution of the user or property rights of natural resources to local communities, the land and natural resources are owned and managed either de jure or de facto by local communities, although state generally retains some control of last resort over land and natural resources. Sustainable extractive use of renewable resources is allowed and encouraged. Local communities play an active and predominant role in management practice of protected areas.

All these ideas have great influences on conservation policies and practice throughout the world. Of the three primary types of community conservation approach, we are particularly interested in community-based conservation and collaborative management which emphasize the role of sustainable use of natural resources as an instrument for conservation. Whether these models are as effective as they are said to be by their proponents has become the focus of the debate.

4.4.4 Perspectives of the preservation approach

The point of view that utilization of natural resources is in principle compatible with conservation objective has been criticized by proponents of the preservation approach. They asserted that, given current ignorance about biological and ecological knowledge, and given the prevailing socio-economic circumstances in most countries, the sustainable use approach will result in substantial losses of biodiversity. Maybe the use strategy is politically correct and intellectually appealing, but it is in most cases less effective than it is said to be. There is growing scientific literature that question the probability of successfully achieving sustainability in practice. For example, after reviewing some historical experiences, Ludwig et al. (1993) asserted that, once natural resources are opened to utilization, they are usually overexploited to extinction or to the population level of collapse. In the following we briefly review the primary factors that contribute to overexploitation.

Some species are particularly vulnerable to harvest as a result of their special biological attributes. For example, it has long been recognized that long-lived and slow-reproducing species, such as primates, elephants, whales, sharks and tropical hard woods have low intrinsic growth rates and are particularly vulnerable to harvest (Mangel et al., 1996; Gullison, 1998; Bennett and Robinson, 2000b). The mammals in tropical forests are also easily overexploited because of their low standing biomass (Robinson and Bennett, 2000). Species whose behavior allow easy harvest, that do not have the ability to recolonize hunted area, or that are intrinsically rare are highly vulnerable to harvest (Bennett and Robinson, 2000b). In addition, uncertainty which emerges from the inherent stochasticity of ecosystems and from human ignorance about biological and ecological knowledge makes the task of sustainablly utilizing resources more difficult than it is expected to be. Many cases show that the fluctuations in resource stock is so great and unpredictable that it is usually too late until significant decrease in resource stock is detected, and this significantly increases the risk of extinction (Ludwig et al., 1993; Lavigne et al., 1996).

At ecosystem level, the use approach is often criticized that it rarely consider the impact of exploiting target species on all the other species living within the same ecosystem and on the ecosystem itself as a whole. There would be enormous impact especially when the target species in question is a keystone species. For example, the exploitation of certain tree species in old-growth forests are usually not compatible with the conservation of the other non-harvested species whose survival depend on the integrity of forests (Struhsaker, 1998). Furthermore, the economic incentive behind the use strategy usually favors the increase of those more highly valued species and lead to the simplicity of ecosystems (Freese, 1997). Simplicity means that through modifying biological or abiotic factors, such as eliminating those species that have competitive or predatory relationship with the highly valued species, the environment will become more feasible for the production of target species at the expense of other components of biodiversity.

In addition to biological and ecological factors, some fundamental socioeconomic factors play an important role in influencing sustainability of natural resource use. Clark (1973) demonstrated in his pioneering paper that low growth rate of renewable resources, high discount rate and high price/cost ratio of harvest are the primary factors contributing to overexploitation. The usually prevailing economic, social and institutional conditions in many countries, such as high interest rate, poverty and uncertainty about tenure and resource market, create high discount rates that are disadvantageous to sustainable use of renewable resources (Freese, 1997). With reference to the problem of price/cost ratio, certain new economic model such as the Swanson model[6] (1994) and empirical cases such as trophy hunting in southern African countries (Child, 2000) show that high price/cost ratio, in contrast to the result of the Clark model, can help conserve utilized species by offsetting the opportunity costs of competing land use options under appropriate property rights and policy environment. However, Freese (1997) maintained that an unusually high price/cost ratio may create a destabilizing environment against sustainable use of renewable resources, such as the numerous examples of the collapse of coastal fisheries throughout the world.

Based primarily on the arguments discussed above, many question the feasibility of the community-based conservation projects in achieving conservation objectives. Songorwa et al. (2000) examined the four premises underlying the community-based conservation: (1) that national governments are willing to devolve user rights, property rights and management responsibilities for natural resources to local communities; (2) that local communities are willing to participate in resource management; (3) that local communities are able to management resources; and (4) that conservation is compatible with rural economic development. After reviewing existing literature regarding the practice of community-based conservation in Africa, they take the skeptical view that, in the African context, the four premises are problematic and the approach is less effective than it is expected to be by its advocates. Community-based conservation is expected to work only in areas with large piece of wilderness where big populations of wildlife still exist and human population density is low. Similar views are shared by Noss (1997), Spinage (1998), Hackel (1998) and Oates (1999). In addition to the arguments discussed above, Noss emphasized the problems of ethnic diversity as a result of immigration and of the lack of conservation ethic that challenge the practicability of grass-root conservation initiatives. Spinage blamed the fallacy of the ecologically noble savage hypothesis for its influence on misleading conservation policy. Hackel addressed the problems of poverty, long-standing economic stagnation and rapid population growth in Africa, and concluded that the idea of community-based conservation is oversold, because it is unlikely that such idea can be applied generally in rural Africa. Oates denounced

[6] The Swanson model will be deliberately investigated in chapter 5.

the claim of some proponents that central governments of many developing countries are not willing to and/or not capable of safeguarding their protected area network. He recognized that many protected areas in developing countries did become increasingly ineffective as a result of the serious economic, social and political problems. However, that is the flaw of rather use approach than of preservation approach. He questioned whether, with much lower management capacity and without public authority, local communities could perform conservation actions more effectively than central governments under the same circumstances. Based on his empirical studies in developing countries, he concluded finally that, in contrast to numerous failures of the community-based conservation initiatives, the traditional top-down preservation model can work relatively well in developing countries, even in the face of the serious economic, social and political problems.

A synthesis of the above arguments leads to the skeptical attitude of many people toward sustainable use approach as an instrument of conservation. They assert therefore that the preservation approach should still be the cornerstone of all conservation strategies (Kramer et al., 1997; Brandon et al., 1998; Oates, 1999). This does not mean that the use approach should not be applied in any case, but that more emphasis and resources should be devoted to the protection, instead of use, of protected areas.

4.5 A case study: the national park system of Taiwan

4.5.1 Introduction to the national park system of Taiwan

Taiwan, with a land area of 35,570 square kilometers, is a subtropical/tropical island located between cool-temperate Japan to the north, subtropical south China to the west, and tropical Philippines and Indo-Malayan islands to the south, a location which is just on the mixed edge of several different biogeographical regions. The island is dominated by rugged, forested mountains with more than two hundred peaks over 3,000 meters. All these climatic, biogeographical and topographic characteristics support a highly diverse flora and fauna communities (COA and DNP, 1992). More than 4,000 species of vascular plant, of those a quarter being endemic to Taiwan, and a spectrum of 6 forest types have been found existing in Taiwan. In addition, Taiwan has a fauna world with 62 species of mammals, 500 species of birds, 95 species of reptiles, 32 species of amphibians, 150 species of freshwater fish, and an estimated 50,000 insect species (of those more than 400 species of butterflies) (COA, 1997).

On the other hand, the rapid economic growth, high population density (over 600 people per square kilometer) and illegal harvesting of natural resources have significantly degraded almost all coast areas, flatland and slopeland below 500 meters in the past fifty years, and are still threatening the remaining intact wilderness. To effectively protect the natural environment of Taiwan, 58 strictly protected areas have been designated in last two decades by the central government of the Taiwan, Republic of China. Of these there are 6 national parks (NP), 18 natural reserves (NR), 11 wildlife refuges (WR), and 23 national forestry natural protected areas (NFNPA) (Lu, 1999). This system covers an area of about 451,951 hectares, or about 12.6% of the land surface of Taiwan (Table 4.1). As Table 4.1 shows, the national park system, comprising 70% of the total surface of the strictly protected areas, is the cornerstone of the nature conservation in Taiwan. Moreover, in contrast to the de facto ´paper park´ status of NR, WR and NFNPA, the national park system generally possesses sufficient staff, budget, and relatively complete management institutions. We therefore, as an example for illustrating the arguments stated in Sections 4.1 and 4.2, concentrate our discussion on the performance and problems of national park system in Taiwan.

Table 4.1 Protected areas in Taiwan

Types of protected area	NP	NR	WR	NFNPA	Total
Numbers	6	18	11	23	58
Coverage/ha	322,207	63,279	11,714	82,654	451,951
% of land surface of Taiwan	9.0	1.80	0.30	2.30	12.6
% of all protected areas	70.12	13.77	2.55	17.99	100

Source: Lu (1999), p.66.

The establishment of the national park system in Taiwan can be traced back to early 1970´s. As a result of the international conservation movement and the growing environmental consciousness of citizens, the National Park Law was promulgated in 1972. However, it was not until 1981 that the authority which is responsible for establishing and supervising national park system, the National Park Department, was established in the Ministry of Interior (DNP, 1999). Thereafter, the first national park in Taiwan, the Kenting National Park, was designated in 1984. The number of national Parks increased rapidly in 1980´s and early 1990´s. Up to now, the system includes 6 national parks and accounts for nearly 9% of the total land surface of Taiwan (see Table 4.2 and Figure 4.1).

Table 4.2 List of National Parks in Taiwan

Name of National Park	Major Protected Features	Area (ha.)	Date of Designation
Kenting National Park	Marine Ecosystem, tropical coastal rainforest, waterfowl, migratory birds, butterflies, limestone caves, cliffs.	17,731 (land) 14,900 (marine)	1.1.1984
Yushan National Park	Virgin forest, wildlife, rare species of flora and fauna, high peaks and mountainous terrain.	105,490	4.10.1985
Yangmingshan National Park	Butterflies, birds, amphibians, volcanic topography, meadows, hot springs, waterfalls.	11,456	9.16.1985
Taroko National Park	Virgin forest, wildlife, marble gorge, cliffs, high mountains.	92,000	11.28.1986
Shei-Pa National Park	Virgin forest, wildlife, rare species of flora and fauna, high mountains.	76,850	7.1.1992
Kinmen National Park	historical battlefields, traditional villages, natural scenic areas, flora and fauna of island.	3,780	10.18.1995

Source: COA and DNP (1992), p.17; DNP (1999), p.26.

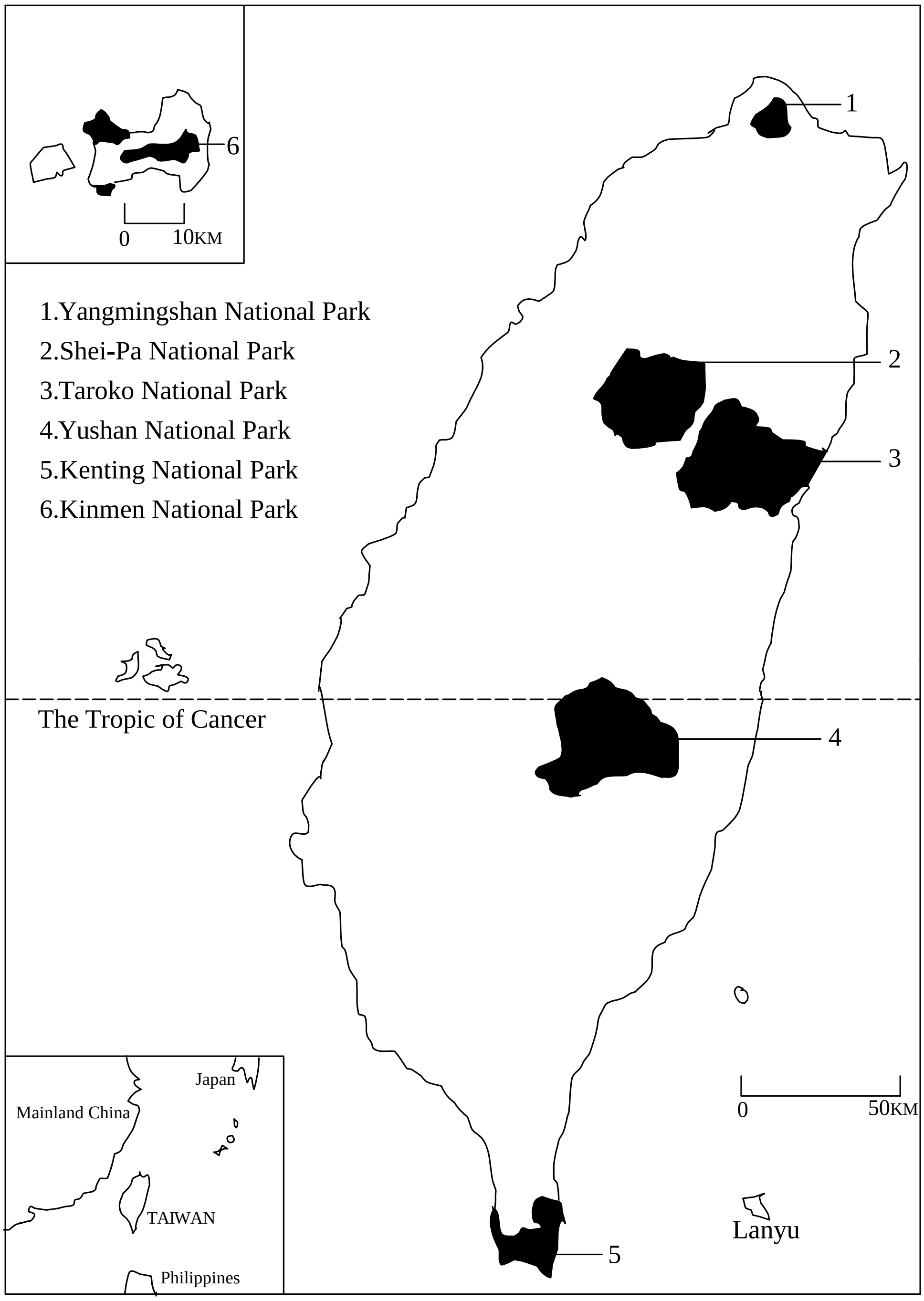


Figure 4.1. National Parks in Taiwan. Source: DNP (1999), P.2.

4.5.2 Management issues

Like most national park systems throughout the world, the national park system of Taiwan aims primarily at protecting natural and cultural resources. And under the premise that the above aim is achieved, national parks can help achieve some minor aims. More specifically, the goals of the national park system of Taiwan include (COA and DNP, 1992):

1. protecting ecologically significant areas
2. conserving gene pools
3. promoting scientific research and environmental education
4. providing nature-based recreational opportunities
5. promoting local economic development through developing tourism that is compatible with nature conservation.

The primary aim of almost all national parks is conserving biodiversity. Only the Kinmen National Park, the first cultural national park in Taiwan, was established primarily for protecting historically and culturally important sites. The park system follows principally the strict preservation model of the U.S. National Parks. Therefore, the extractive use of natural resources in national parks is generally not allowed, with the exception that some limited land use forms, such as grazing, farming, fishing (only in marine area) and use of hot springs may be allowed under the supervision of the park authority, if they have long existed before the park is designated. In any case, harvest of wildlife is absolutely forbidden. In order to achieve the multiple objectives of park system which are in certain cases conflicting with each other, the area within the boundary of every national park is divided into several different management zones. These includes (Lin, 2000):

(1) General Protection Areas: they are land or marine areas in which certain land use forms have long existed before the park is designated. Limited land use under the supervision of the park authority is allowed to continue to mitigate the potential conflicts between local residents and park authorities. They accounts for 30.36% of total area of the park system.

(2) Recreation Areas: these are areas in which recreational facilities are allowed to be established and large numbers of tourists are allowed to enter. They accounts for 0.69% of total area of the park system.

(3) Cultural/Historic Preservation Areas: these are areas with special cultural and/or historic significance in which tourist use is limited. They comprise only 0.12% of total area of the park system.

(4) Special Scenic Areas: those strictly protected areas with special or spectacular natural scenery. Any development activities are forbidden. Tourism is allowed, but its impact on natural environment must be kept to a minimum level. They accounts for 10.74% of total area of the park system.

(5) Ecological Protection Areas: those strictly protected areas with unique ecological value. Any development activities in and human access to these areas are forbidden, with the exception that scientists and hikers with special permits are allowed to enter these areas. Hikers should stay on existing trails. They comprise 58.10% of total area, and are the cornerstone of the in situ conservation in national park system.

According to the National Park Act, the department of National Parks in Construction and Planning Administration, Ministry of Interior, is the authority responsible for selecting and managing national parks. To effectively manage individual national park, each park is supervised by a park headquarter which consists one police corps and five administrative divisions, including the planning, the construction, the conservation, the tourism and the interpretation education division. In general, the central government of Taiwan, Republic of China, advocates the in situ conservation in national parks, and equip park headquarters with sufficient funds and young, highly qualified staff. This contributes to the result that the department of National Parks and park headquarters have become the most active public authorities in safeguarding biodiversity in Taiwan.

Table 4.3 tabulates the budget of the national park system during the time period 1982 to 2000. The budget devoted to the national park system by central government has steadily increased since the first park was planned. This reflects partly the rapid increase in the number of national parks during 1986 to 1996. However, even after the number of parks ceased to increase in 1996, the total budget still slightly increased until the fiscal crisis of the central government changed the trend in 2000. Of the many budget categories, staff salary and investment in equipments and infrastructure constitute the majority of the total budget. The steady increase of expenditures on staff salary reflects that both staff and average pay have increased in succession, or generally speaking, that the human resources of the park system have significantly improved in the last two decades. Similarly, the generally increasing expenditures on investment in equipments and infrastructure reflects that the park system is improving in physical capital. All these factors contribute to the relatively well performance of the park system in recent years.

Table 4.3 Budget of the national park system of Taiwan (Unit: NT$1,000)

	Staff salary	Investment in equipments and infrastructure	Others	Total
1982	2,508	0	2,710	5,218
1983	2,640	0	2,710	5,350
1984	13,801	101,000	101,057	215,858
1985	40,686	239,330	87,452	367,468
1986	92,816	415,710	152,386	660,912
1987	69,955	400,480	202,369	672,804
1988	93,702	628,428	234,410	956,540
1989	117,523	829,041	286,851	1233,415
1990	130,757	1074,679	362,110	1567,546
1991	194,519	1211,041	449,349	1854,909
1992	297,583	1054,861	410,730	1763,174
1993	348,733	1116,150	647,381	2112,264
1994	444,001	1211,916	405,813	2061,730
1995	481,028	1315,790	379,209	2176,027
1996	544,125	1086,095	449,835	2080,055
1997	601,425	1292,226	427,188	2320,839
1998	625,020	1664,187	332,698	2621,905
1999	677,816	1636,608	335,400	2649,824
2000	983,514*	1804,736*	661,068*	3449,318*

*Figures including the budget of one and a half year.
Source: EYROC (1981-1999) and personal calculation.

4.5.3 Effectiveness of the national park system

In this subsection we make use of some indicators to evaluate the effectiveness of the national park system in conserving biodiversity. For some rare plants and animal species, especially mammals with large body size, national parks have become their last refuges under the strict preservation policy. Table 4.4 shows the number of plants species which have been found to exist in national parks. As Table 4.4 indicates, each national park, except the Kinmen National Park, protect more than one fourth of the total about 4,000 species of plants existing in Taiwan. As a result of the overlapping of some species, it is unknown whether national parks protect most of the plants species existing in Taiwan. But it is hardly probable that national parks safeguard only a minor proportion of total plants species.

Table 4.5 tabulates the number of animal species which have been found to exist within the boundary of national parks. Leaving aside the animal species found in Kinmen National Park, the other five national parks protect most of the mammals, birds and amphibians species, and about half of the reptiles and fresh fish species. In addition, 1,105 sea fish, 616 Mollusk, 327 coral and 295 crustacen species have been found in marine area of the Kenting National Park (CPA, 2000).

In the past five decades, habitat degradation and illegal harvesting of wild plants and animals has severely reduced populations of any valuable species to an endangered level. Some species, such as clouded leopard (*Neofelis nebulosa brachyurus*) and Formosan flying fox (*Pteropus dasumallus formosus*) are suspected to have gone extinct for a long time. Until after the national parks have been designated in succession and the National Park Act effectively enforced by park headquarters, the depressing situation has, at least within the boundary of national parks, to great extent changed. Some observations and reports in the field showed that populations of wildlife, including almost all of the mammal species with large body size and rare bird species such as Formosan black bear (*Ursus thibetanus formosanus*), Mikado pheasant (*Syrmaticus mikado*) and Swinhoe´s pheasant (*Lophura swinhoii*), have significantly recovered in national parks (The Nature, 1991; 1995b). Based on the standing observations of scientists and park staff, it seems to be reasonable to assert that national parks have effectively safeguarded the flora and fauna communities within park boundaries, although more rigorously scientific works are still needed to evaluate the long-term effectiveness of national park system in conserving biodiversity.[7]

Equipped with sufficient funds and highly qualified staff, park headquarters have actively dampened the illegal activities within national parks (Sung, 1999). Table 4.6 tabulates the number of illegal activities detected by park headquarters in 1990s. No clear-cut conclusions with reference to the trend of illegal activities can be drawn from the statistics. However, even under the severe pressures resulted from economic development and increasing human

[7]Some exceptions to this assertion, however, will be discussed in the next subsection.

population, illegal activities are as ever kept under control by park headquarters to an acceptable level, except in a few cases discussed later.

One of the important tasks of the national park system is to provide interpretation education service for tourists which aims at raising public environmental consciousness. Table 4.7 shows the statistics of the relevant briefing and guided tour attendance in 1990s. Both the briefing and guided tour attendance have dramatically increased, almost year by year. It is difficult to measure the precise effects of interpretation education service on the formation of public environmental consciousness, but the more environmental friendly behavior of the tourists in national parks in recent years might be partly attributed to the active promotion of environmental education initiated by park authorities.

4.5.4 Current problems of the national park system

While certain degree of success in preserving majority of the floral and faunal species and in protecting some types of habitat have been achieved by the national park system of Taiwan, a critical investigation into the system shows that, like almost all of the protected area systems throughout the world, the selection, establishment and management of national parks in Taiwan are involved in some problems which need to be solved.

First, from the perspective of representative, the national park system should try to encompass the representatives of as many types of ecosystems as possible. However, under the circumstances that lowland, coastal and marine areas have long been intensively cultivated, settled or utilized, some ecosystem types in these areas, including lowland forests, wetlands, coastal regions and marine ecosystems, are virtually underpresented in national park system, while most parts of the national parks are located in lower-mountain or mountain forests. This is recognized by the Department of National Parks to be one of the current deficiencies of the national park system. To improve the representative, they plan to establish a few new national parks which include wetland and marine parks (The Nature, 2000).

Next, although the effective enforcement of the National Park Act has brought most illegal activities, such as poaching, logging, mining and cultivation, under control, the existing national parks are generally suffered from some management problems, in particular the excessive tourism and hostility of local people toward national parks. Some popular and spectacular sites in national parks attract several millions of tourists every year, and this has severely degraded some important habitats in the parks. A well-known example is the destruction of coral reefs in Kenting National Park caused by pollution and human disturbance brought by about four millions of tourists per year (The Nature, 1996; Dai et al., 1998, 1999). Similarly, about 12 millions of tourists per year visit the Yangmingshan National Park which is located near the metropolitan city Taipei, and seriously degrade some parts of the park (Lin, 2000). Table 4.8 demonstrates the rapidly increasing pressure of tourism putted upon the national park system in recent years. From 1992 to 1999, the number of tourists visiting national parks has dramatically increased by 78.13%.

The solutions to the problem of excessive tourism may be limiting tourist numbers and limiting access to fragile habitats, but such measures will probably, at least in the short term, reduce tourism revenues and arouse more severe hostility of local people toward national parks than before. Indeed, as a result of the strict protection policy of national parks, the conflicts between local residents and park authorities have long lasted. Although limited land use under the supervision of the park authority is allowed in general protection areas if these land use forms have long existed before parks are designated, local residents living in and around national parks still claim that their traditional land and natural resource user/property rights have suffered from the designation of national parks. All national parks have therefore, to various extent, experienced the protest of local residents (The Nature, 1993, 1994a, 1994b, 1995a, 1999; Sung, 1999; Huang, 1999). In particular, the indigenous people played a primary role in the protest initiatives, because the majority of national parks is located in the traditional territory where indigenous people live and lay claim to. They claimed the land tenure, respect for the indigenous culture, relaxation of land use limitations and legal hunting, fishing and gathering rights of renewable resources in national parks.[8] Meanwhile, national park authorities persist in their strict protection policy, and these conflicts led finally to the failure of the efforts to expand the existing national park system. Under severe protest of indigenous people, the Department of National Parks stopped designating two planned new national

[8]It is here worth noting the fact that not all indigenous people oppose the designation of national parks. Some people consider that national parks effectively prevent the modern market economy from invading indigenous communities in and around national parks, and thereby, maybe without intention, help protect their traditional culture and land tenure (Huang, 1999; Ming-I Gu, personal communication, 4.17.2001). Without national park or similar protected area and under the pressure of the predominant modern civilization, as the example of the planned Lanyu National Park shows, indigenous people might have lost their culture, natural resources and land tenure more rapidly than expected in the situation if national park is successfully designated (Huang, 1999).

parks, including the Lanyu National Park (The Nature, 1994b; Huang, 1999) and Nandan National Park (Huang, 1999). In addition, whether the planned Chilanshan National Park can be successfully established under protest, remains still to be seen. In short, except the successful designation of the Kinmen National Park in 1995, all plans for new parks proposed after middle 1990s were forced to been abandoned or deferred by protests initiated by indigenous people.

Table 4.4 Numbers of plant species in national parks of Taiwan

National Park	Pteridophyta	Angiospermae	Dicotyledons	Monocotyledons	Total
Kenting	194	3	898	143	1,238
Yushan	254	18	726	146	1,144
Yangmingshan	181	2	747	294	1,224
Taroko	223	18	747	294	1,183
Shei-Pa	223	19	706	155	1,103
Kinmen	36	2	351	153	542
Taiwan area	582	28	na	na	na

Source: Lin (2000), p.26; CPA (2000), p.143.

Table 4.5 Numbers of animal species in national parks of Taiwan

National Park	Mammals	Birds	Reptiles	Amphibians	Fresh fish	Sea fish
Kenting	15	184	35	14	30	1105
Yushan	34	154	17	12	3	0
Yangmingshan	14	77	35	20	12	0
Taroko	31	147	30	14	16	0
Shei-Pa	32	97	14	6	16	0
Kinmen	8	281	13	5	na	9
Total animal species in national parks*	43	260	57	25	42	na
% of animal species of Taiwan*	69%	65%	44%	86%	50%	na

*Figures excluding the animal species of Kinmen National Park.
Source: Lin (2000), p.26; CPA (2000), p.142.

Table 4.6 Numbers of detected illegal activities in national parks of Taiwan

	Squatter	Cultivating	Hunting/ Fishing	Lumbering	Polluting Environment	Others*	Total
1992	215	80	26	246	81	1,467	2,115
1993	69	43	58	123	9	747	1,049
1994	45	79	85	121	65	851	1,246
1995	47	55	69	64	33	923	1,191
1996	92	114	26	71	17	977	1,297
1997	124	104	53	84	49	2,087	2,501
1998	116	78	18	54	7	1,534	1,807
1999	90	59	24	39	32	752	996

*Including vending, dumping garbage, arsoning, extracting coral, and trespassing and driving into protected areas.
Source: CPA (2000), pp.128-129.

Table 4.7 Briefing and guided tour attendance in national parks of Taiwan

	Briefing attendance	Guided tour attendance	Total
1992	404,722	139,553	544,275
1993	409,072	124,678	633,750
1994	414,567	158,900	573,467
1995	444,246	174,288	618,534
1996	452,111	176,312	628,423
1997	517,699	211,659	729,358
1998	578,257	387,643	965,900
1999	584,349	346,593	930,942

Source: CPA (2000), pp.114-115.

Table 4.8 Number of tourists and entrance fee revenue of national parks in Taiwan*

	Number of Tourists	Entrance fee revenue (Unit: NT$1,000)
1992	4845,661	26,033
1993	5805,332	26,036
1994	6811,596	21,343
1995	7437,589	28,730
1996	6429,216	27,083
1997	8525,707	27,878
1998	8290,840	28,443
1999	8631,763	36,448

*These figures do not include all tourists, because people are allowed to visit some parts of national parks without paying entrance fees.
Source: CPA (2000), pp.102-103.

4.5.5 Prospects of the national park system

The young national park system of Taiwan is, to great extent, successful in protecting biodiversity within park boundaries, at least at its beginning stage between the year 1984 and 2000. Its success can be attributed to the two pivotal factors: the support of the central government and the strict protection policy. While the support of the public authority is in any case necessary for the maintenance of such a large protected area system, however, the strict protection policy, as a critical factor contributing to safeguarding biodiversity, ironically hindered the expansion of the national park system, and thereby hindered the possibility of protecting biodiversity out of range of the existing park system. From the long-term perspective, the existing national park network need to be enlarged to include a more complete pallet of various ecosystems and thereby to maximize the protected biodiversity. To effectively protect biodiversity, the current management problems in some national parks should be solved. All these require the support, or at least the tolerance of the local residents, in particular of the indigenous people, who virtually bear the majority of the costs derived from the designation of national parks. The question is, instead of the strict protection policy, is there a better alternative under current circumstance in Taiwan?

To resolve the conflicts between local communities and park authorities, the central government proposed to modify the National Park Act, so that indigenous people have the legal rights to utilize the renewable resources in national parks (The Nature, 2000). The possible modification of current conservation policy aroused the controversy between proponents and opponents of the sustainable use strategy, like the prevailing debate within conservation communities throughout the world (Liu, 2000; Chang, 2001). Furthermore, some conservationists proposed that the idea of co-management could be applied in national parks. In any case, whether and under which conditions use of renewable resources is feasible will be the core of all policy issues in national park system of Taiwan. This is also what we are interested in and will focus on throughout the dissertation.

Chapter 5
Economic models of species extinction and biodiversity loss

In chapter 4 we have identified some primary problems of existing protected area systems. All these problems indicate that protected areas cannot exist in isolation from the social and economic system in which they have been created. While their major task is conserving biodiversity, they are confronted with the same problems which biological communities generally are faced. To remedy the problems of protected areas, it is therefore necessary to investigate and grasp the fundamental causes of species extinction and biodiversity loss.

In this chapter we will briefly investigate three important economic models which have been derived with regard to the optimal utilization of renewable resources and to the problems of species extinction and biodiversity loss, including the Gordon model, the Clark Model and the Swanson model. These models provide the general analytical framework for most of the analysis that handle the same issues, also for our models presented in the following chapters. Moreover, the investigation into these models will provide critical insight for the rethinking of the prevailing conservation policies.

5.1 The Gordon model

The Gordon model (Gordon, 1954) may be, though indirectly, the origins of the economic theory of species extinction. In his paper Gordon examined how property rights influence the extent to which fish population is exploited, especially under the so called ´sole owner´ regime and the ´open access´ regime. The latter means that no one, de facto and/or de jure, owns the natural resources, and access to resources is open to all. The well-known conclusion drawn by the Gordon model is, that the equilibrium population level under the sole owner regime will be higher than the one under the open access regime, since, under open access, the resource will still be exploited by any new entrants until that total revenues gained from harvesting equal total harvest costs when profits are totally dissipated, while the sole owner will choose to maximize his profits and thereby maintain a higher population level as a result of the self-interest motive. It follows that the phenomenon of overexploitation can be attributed to the inadequate property rights. What in the Gordon model is particularly noticeable is the fact that it is not possible that fish population is exploited to extinction, irrespective of under open access or under sole owner regime, as long as the harvest costs are positive. Even though at zero price of harvest effort, the sole owner will choose to harvest the ´maximum sustainable yield (MSY)´, and thereby preserve a positive population. This conclusion contradicts obviously the extinction phenomenon which is occurring worldwide.

Gould (1972) has made some supplements within the framework of the Gordon model. He pointed out that, under open access regime, extinction of a species is possible, if (1) the species has a minimum viable population size, or (2) the price of resource is so high that the price of the last remaining unit of resource before extinction is still higher than the marginal harvest costs. In sum, in the context of the Gordon model, the open access regime significantly raises the extinction risk of exploited species, while the sole owner regime and the equivalent profit maximization behavior never leads to species extinction.

5.2 The Clark model

The Gordon model is a purely static model, and has therefore neglected an important factor that significantly influences the use of renewable resources, namely, the time factor. In his pioneer work Clark (1973, 1976) demonstrated that a dynamic analysis will change the basic conclusion of the Gordon model, so that exploitation to extinction may, under certain biological and economic conditions, appear as the optimal policy, even to the profit-maximizing sole resource owner.

Now let´s demonstrate the Clark model by the use of a version of the model suggested by Pearce and Turner (1990). Clark suggested that it is plausible to assume that the sole owner of the resource prefers present to future revenues, and would try to discount future revenues at certain rate and maximize the present value of the profits derived from his harvest yields. The profit maximization behavior of the resource owner can be formulated as follows[9]:

$$Max \quad \int_0^\infty [p - c(x)]\, he^{-\delta t} dt$$

$$s.t.\ \dot{x} = F(x) - h,\ x(0)\ given \tag{5.1}$$

[9]For simplicity, the following variables are assumed to be subscripted for time throughout this chapter.

where x: resource stock level
p: constant unit price of resource
$c(x)$: unit harvest cost when resource stock level$=x$ with $c'(x) < 0$
h: harvest rate
δ: instantaneous rate of discount
$F(x)$: natural reproduction rate of the resource stock with $F(x) \geq 0$ and $F''(x) < 0$.

The following optimal condition for the optimization problem can be then derived:

$$F'(x) - \frac{c'(x)F(x)}{p - c(x)} = \delta. \tag{5.2}$$

If we define

$$R = [p - c(x)] \ F(x) = [p - c(x)]\, h \tag{5.3}$$

in a stationary state, R can then be interpreted as the sustainable rent at population level x. Equation (5.2) can be rewritten as

$$\frac{1}{\delta} \cdot \frac{dR}{dx} = p - c(x) \tag{5.4}$$

which states that the marginal profit derived from an increase in the current harvest must equal the present value of the marginal future loss when x is optimal.

According to equation (5.4), it can be shown that the optimal resource stock level is lower, the higher the discount rate and the resource price, and the lower is the cost per unit harvest. In addition, Clark (1976) proved that a zero equilibrium population level would be optimal, if an immediate profit can be made from harvesting the last remaining individual of the population, and the discount rate is more than two times as large as the growth rate of the endangered population. Moreover, the open access regime is only a special case of the Clark model, namely, in which the sole owner adopts an infinite discount rate and thereby sets a zero value on future revenues. Therefore, an extremely high discount rate implies also indefinite property rights.

It follows, within the framework of the Clark model, that the combination of a high price-cost ratio, high discount rate, low growth rate of species and overexploitation that results from the former three factors are the fundamental causes of species extinction. Its policy implication is straightforward.

While the growth rate of species is biologically given and not easy to be increased, we may manipulate the three variables resource price, harvest cost and discount rate to prevent from the extinction of species. Means such as building definite property rights (reducing the discount rate), prohibitions of wildlife trade (reducing the demand for resource and thereby reducing resource price) and criminalisation of hunting (raising the harvest cost) are therefore frequently recommended as the necessary and adequate policies for species conservation.

5.3 The Swanson model

While the Clark model can certainly be applied to some cases of species extinction resulted from overexploitation, it is evident that, instead of overexploitation, habitat degradation and conversion, such as conversion of rain forests to ranches, is the only cause in most cases that results in mass extinction, or the so called biodiversity loss. To remedy this deficiency, Swanson (1994) developed a generalized model that explains extinction as a human decision process of the naturally biological resources to retain in the assets portfolio. As a part of the human assets portfolio, wild biological resources must generate high enough flows of benefits to compete with the man-made assets. Otherwise, they will be converted to different forms of man-made assets until the returns between various assets are equilibrated. The different rates of return between assets, as the Swanson model indicates, are the fundamental forces which drive numerous species to extinction.

Compared to the Gordon and the Clark model, the Swanson model is characterized by two special features. First, it introduces an important factor in the model, namely, the so called base resources (habitats in natural state, water, etc.) on which the subsistence of all wild species depend,[10] and thereby endogenizes the allocation decision of base resources to wild species, while the Gordon and the Clark model are in principle based on the fishery economics, and thus assumed implicitly that base resources, namely the sea in the case of the fishery, are exogenously given and costless. Next, it asserts that many wild species require not only base resources, but also an another important form of ancillary resource, namely, the so called management services, which are mainly used for institution-building, such as the establishment of property rights. The Swanson model endogenizes thereby the investment decision process about the amount of management services which are allocated to

[10] For example, once a piece of rain forest is converted into cattle farms, almost all the wild species on this land will not survive. Whether retaining a piece of rain forest in its natural state, is a decision made by human society or land owner.

some given species. These features imply that the flow of benefits derived from a given species depends on three factors: harvest rate, level of base resources and level of management services.

Based on the Clark model, the bioeconomic model of species extinction can be modified to investigate the important role of the base resources for the survival of species. To do this, the Swanson model can be developed step by step as follows. First, a new variable R is introduced in the model which represents the level of base resources retained for the survival of biological communities. The traditional logistic growth function of species which depends only on resource stock level, $F(x)$, must be revised to adopt the new variable R, so that we have a new logistic growth function of species $F(x;R)$ which depends on both the resource stock level x and the amount of base resources R allocated to given species. This function implies that an increase in R would lead to an upward shift of the natural growth function of wild species, and vice versa, as shown in Figure 5.2. Such an upward shift will raise the overall growth potential of wild species at every population level. However, the allocation of base resources to wild species, for example returning a cattle farm to wilderness, causes the resource owner to incur opportunity cost which can be expressed as the potential flow of benefits available from the best alternative use of the base resources. For each period, the incurred opportunity costs will be those resulted from the market value of base resources, $\rho_R R$, multiplied by the interest rate, r. Together with the gross profits received from harvesting target species, the maximization behavior of the resource owner can be formulated as follows:

$$Max \quad \int_0^{\infty} [ph - c(x)h - r\rho_R R]\, e^{-rt} dt$$

$$s.t. \; \dot{x} = F(x;R) - h \tag{5.5}$$

where x: resource stock level

p: constant unit price of resource

$c(x)$: unit harvest cost when resource stock level$=x$ with $c'(x) < 0$

h: harvest rate

r: interest rate

R: amount of base resources

ρ_R: unit price of base resources

$F(x;R)$: natural reproduction rate of the resource stock with $F(x;R) \geq 0$, $F_{xx}(x;R) < 0$ and $F_R(x;R) > 0$.

This is a dynamic model with one state variable x and two control variable h and R. With the inclusion of the additional control variable R in the model, it yields accordingly an additional first-order condition regarding the optimal investment in base resources:

$$R^* : \frac{\mu \cdot F_R}{\rho_R} = r \tag{5.6}$$

where μ denotes the shadow price of the resource stock in the steady state. This condition states that the marginal value product derived from an allocation of base resources to production of species must equal the incurred marginal costs in the steady state, or in other words, the resource owner will retain base resources for growth of the target species only to the extent that the species can offer a competitive rate of return from retaining base resources.

By the application of the optimal condition, let´s examine what would happen if the price of the target species rises. On the one hand, this would raise the incentive to exploit species, but on the other hand this implies that the shadow price of species in the steady state μ would increase. An increase in μ would raise the relative rate of return for this species, as shown by the left hand side of (5.6), and thereby raise the optimal level of investment in base resources for the species, as shown in Figure 5.1. More base resources would be then made available to the species because of its increased investment-worthiness. This would lead to an upward shift of the growth function of species, as demonstrated in Figure 5.2, and thereby decrease the extinction risk of the species. At the same time, numerous other species would benefit from the increased allocation of base resources, a situation in which biodiversity is better conserved than ever. In the reverse case, if the price of species falls, a smaller amount of base resources would be allocated to the species as a result of its investment-unworthiness. The extinction risk of the harvested species and of all the other species would therefore rise, a situation in which biodiversity is worse conserved than ever.

Therefore, while overexploitation is one of the primary factors contributing to species extinction in some cases, most extinction cases of numerous unknown species are in fact the direct result of the habitat degradation and conversion which can be regarded as the active removal of base resources by resource owner as a result of the investment-unworthiness of these species. Instead of overexploitation, it is active conversion of base resources into man-made assets that threatens most species, or biodiversity of the world today.

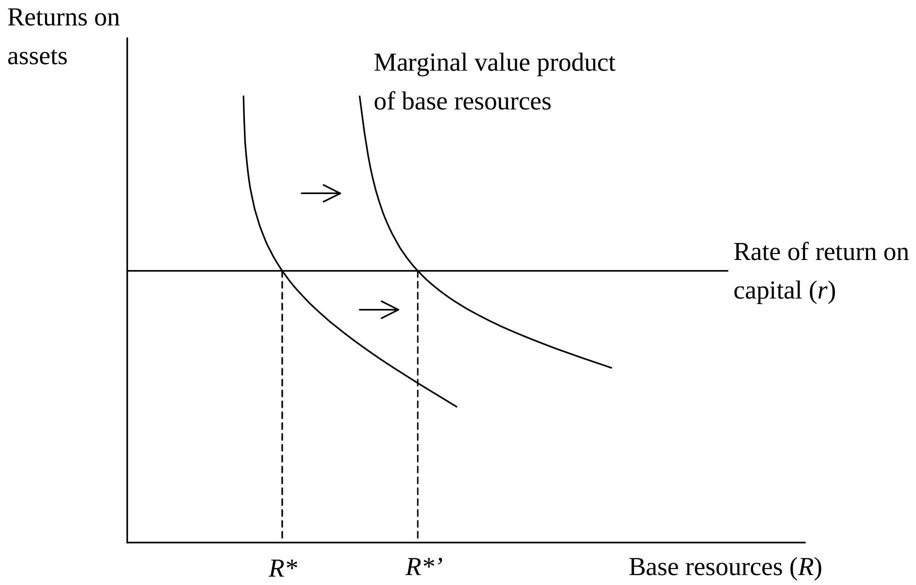


Figure 5.1 Optimal allocation of base resources. Source: Swanson (1994), p.63.

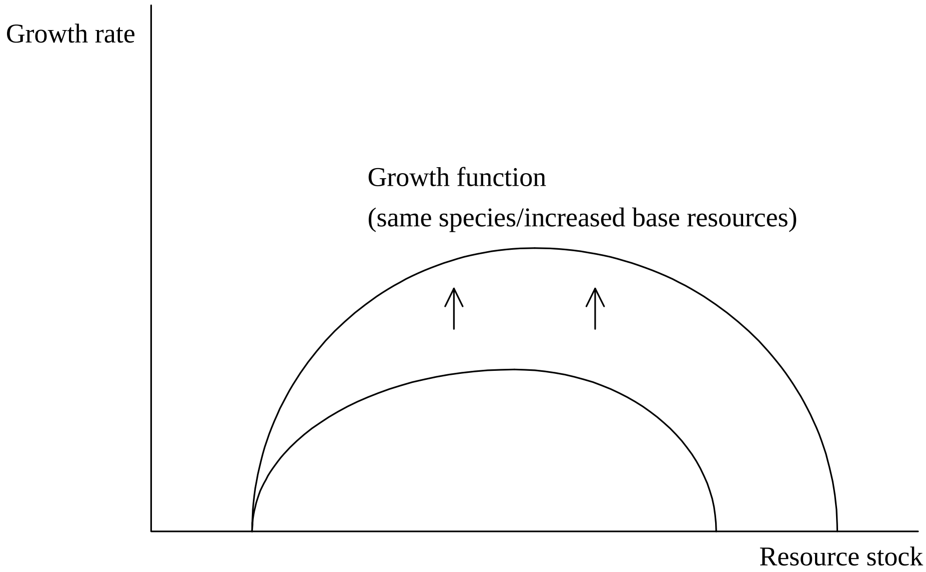


Figure 5.2 Upward shift of the growth function of the species resulted from increased base resources. Source: Swanson (1994), p. 61.

Furthermore, as Swanson indicated, the management factor is just as important for the survival of species as base resources. Even when substantial base resources have been allocated to certain species, management services, especially the establishment of property rights, are still required for their maintenance and protection. To demonstrate the importance of management in the decision process of the resource use, Swanson introduced a new variable M in the model which represents the level of the management services devoted by resource owner for protecting and managing renewable resources. The previous logistic growth function of species $F(x; R)$ which depends on both the resource stock level x and the amount of base resources R must be revised to adopt the new variable M, so that we have now a new logistic growth function of species $F(x; R, M)$ which depends additionally on, except the resource stock level x and the amount of base resources, the level of the management services. And just like the influence of R on growth function of species, an increase in M would raise the overall growth potential of species, and vice versa. After considering the opportunity cost incurred by the devotion of management services, the Swanson model can be modified again as follows:[11]

[11] In the original model, Swanson used an equation of motion for resource stock $\dot{x} =$

$$Max \quad \int_0^\infty [ph - c(x)h - r\rho_M M - r\rho_R R]\, e^{-rt} dt$$

$$s.t.\ \dot{x} = F(x; R, M) - h \tag{5.7}$$

where M: level of management services
ρ_M: unit price of management services
$F(x; R, M)$: natural reproduction rate of the resource stock with
$F(x; R, M) \geq 0$, $F_{xx} < 0$, $F_R > 0$ and $F_M > 0$.

This is a dynamic model with one state variable x and three control variable h, R and M. With the inclusion of the additional control variable M in the model, it yields accordingly an additional first-order condition regarding the optimal investment in management services:

$$M^* : \frac{\mu \cdot F_M}{\rho_M} = r. \tag{5.8}$$

This condition states that the marginal value product derived from allocation of management services to production of species must equal the incurred marginal costs in the steady state, or in other words, the resource owner will invest necessary management services for the growth of the target species only to the extent that the species can offer a competitive rate of return from this investment.

What condition (5.8) implies for the optimal allocation of management services and for the extinction risk of species, is in principle the same as the condition (5.6) does. In short, an increase in resource price would raise the relative rate of return for this species and thereby raise the optimal level of management services for the species. More management services would be then made available to the species because of its increased investment-worthiness. This would lead to an upward shift of the growth function of species and thereby decrease the extinction risk of the species. Simultaneously, many other species would also benefit from the increased allocation of management services, so that biodiversity is better conserved than before. On the contrary, if the price of the harvested species falls, a smaller amount

$F(x) - h$ and a utility function $S(h; R, M)$. However, we make here a modification by using the equation of motion for resource stock $\dot{x} = F(x; R, M) - h$ and the revenue function ph that accord with what are used in (5.5).

of management services would be allocated to the species as a result of its investment-unworthiness. The extinction risk of the harvested species and of all the other species existing in the same habitat would therefore rise, a situation in which biodiversity is worse conserved than ever. It follows that the usual open access state of poorly managed resources is in fact the result of the disinvestment in management services by resource owner as a result of the investment-unworthiness of these species.

To sum up, within the framework of the Swanson model, the only fundamental cause driving the extinction process is that the species is not worth retaining in the assets-portfolio of the resource owner. If the species can generate only a relative low rate of return and therefore cannot compete with the man-made assets, the resource owner would have no incentive to invest in the growth potential of the species. Many species are depleted to extinction, because the necessary base resources for their survival are converted to other forms of use which afford a higher rate of return. Some other species are subjected to inadequate property rights, such as open access regime, because they are not worth investing management services. Finally, some species with extremely high price-cost ratios and low growth rate are endangered, since the owner is willing to harvest the total population to cash in them and invest the funds in other forms of assets which have a higher growth rate than the species, as the Clark model demonstrated. Therefore, in total there are three alternative routes to extinction: base resource conversion, inadequate management and overexploitation. However, they are not really fundamental, but merely the proximate causes for observed extinction. In other words, they are merely different ´routes´ to extinction rather than the fundamental cause resulting in extinction. The only fundamental cause of extinction is the investment-unworthiness of the species. In this connection the Swanson model affords a more generalized explanation regarding the loss of biodiversity than the Clark model.

The policy implications of the Swanson model are as follows. To remedy the loss of biodiversity resulted from the investment-unworthiness of wild biological resources, human society needs to develop effective approaches to capture the values of the wild species given current human preferences, including consumptive and non-consumptive values. With regard to the consumptive value, as Swanson asserted, this would imply in many cases that we should try to increase the difference between resource price and harvest costs, rather than to minimize or even to eliminate it. In other words, the use of wild biological resources could be an effective instrument for conserving biodiversity and should be promoted. Swanson maintained therefore that conservation policies of demand destruction and supply criminalization which try to minimize the price-cost ratios, such as the prohibition of wildlife trade and hunting, are misleading. With reference to the protected area policies, the Swanson model implies that the ´fence and police´ policy prevailing in many protected areas may be misguided and should be reconsidered. For the sake of conserving biodiversity, we should seriously consider the question, whether wild biological resources in protected areas should be utilized to a greater extent than today. According to the Swanson model, this may in the long run raise the incentive of governments and local residents to invest more resources in the management of protected areas or even to expand the existing protected area networks, so long as the use of biological resources can afford a high enough rate of return from this investment.

The example of South Africa may support the rationale of the Swanson model. As a result of the promotion of the sustainable use of wildlife, especially of safari hunting and tourism, many farmers ceased cattle farming, returned their farms to natural habitats that are suitable to the survival of wildlife (increasing base resources for wild species, in the terminology of the Swanson model), and thereby built private reserves, because wildlife use is in many cases a form of land use that is more profitable than the traditional farming. The management efforts and devotion of land for wildlife subsistence invested by private individuals and enterprises has resulted in steady increase in wildlife populations (Grootenhuis and Prins, 2000) and increase in privately protected habitat area which is even greater than the total land area under the control of the National Park´s Board (Hearne and Mckenzie, 2000). This contributed not only to the conservation of wildlife, but also to the conservation of biodiversity.

5.4 Concluding remarks

The basic theme of the sustainable use approach is that, through creation of economic incentives, people will invest more resources in management of protected areas and thereby reduce simultaneously the illegal exploitation of renewable resources and the amount of wildlands which would be converted to land use of non-conservation alternatives. It is clear that, with the extensive explanation for the fundamental cause of biodiversity loss, the Swanson model provides an excellent theoretical foundation for the promotion of the sustainable use strategy as an instrument of biodiversity conservation, and is certainly a feasible starting point for further theoretical modeling. However, the Swanson model is still a rough model in the sense that, except asserting the importance of endogenizing the factors of land use competition and management, it did not deliberately investigate the dynamic interaction between the resource stock, base resources, management and harvest when considering the resource use and management issues. With its simple analysis, it suggested that a higher resource price/harvest cost ratio will enhance the investment-worthiness of the resource, and thereby contribute to the growth potential of the resource stock. However, it neglected the effect that an increased resource price/harvest cost ratio will simultaneously enhance the incentive of the resource owner to exploit resources and to cash in, as demonstrated by the Clark model. Therefore, whether an increased resource price/harvest cost ratio will lead to a net increase or decrease of the equilibrium resource stock, is a question that needs to be addressed. Finally, the Swanson model also did not consider the problem of illegal harvest which plays an important role in the resource management issues of the real world. Hence, more deliberate modeling is needed.

The sustainable use approach could be applied both inside and outside protected areas. The primary objective and task of this dissertation is to investigate whether and under which conditions the sustainable use approach is a feasible instrument for biodiversity conservation in existing protected areas. While the positive effect of the sustainable use approach on land use decisions outside existing protected areas is in principle uncertain and cannot be measured precisely, the use approach will result in direct impact on the population levels of the harvested species. Considering the fact that overexploitation in existing protected areas in exchange for future, unpredictable conservation benefits outside protected areas will hardly be accepted, the use approach should be applied in protected areas with particular caution. Hence, rather than concentrating on issues of land use competition, as Swanson did, we will confine the dissertation to addressing the related harvest and management issues of the sustainable use strategy applied in protected areas. Based on the findings of the Clark and the Swanson model, several new models will be developed in the next chapters.

Chapter 6
Use of renewable resources, poaching and anti-poaching: a simple bioeconomic model with one state variable and two control variables

6.1 Introduction

During the last two decades, one of the most serious controversy regarding the conservation issue involved the following questions: is sustainable use of renewable resources possible and appropriate in and around protected areas? Could such a strategy really help to protect biodiversity more effectively, or just on the contrary, it would degrade biodiversity more rapidly? To answer these questions, two often conflicting approaches, the preservation and the sustainable use approach, dominated the debate, as discussed in chapter 4.

In addition to the two somewhat extreme positions, many conservationists may agree that what is the best strategy will depend on a variety of biological, economic and social conditions at the specific site. There are seemingly no clear-cut answers, but, based on case studies arising from different circumstances, some general patterns and themes emerge (see Freese, 1997; Bennett and Robinson, 2000).[12] These case-study-based observations regarding harvest of renewable resources furnish the conceptual framework for generating and testing hypotheses and represent a crucial first step. Nonetheless, they are mostly fragmentary and lack theoretically consistent foundation. There are relatively few rigorously theoretical analysis to address the implication of the sustainable use strategy for biodiversity conservation. The existing analytical models were mainly developed by biologists and thereby based principally on biological points of view.[13] Therefore, it is necessary to develop models which incorporate economic rationale into the existing models of harvest.

In this context, Skonhoft and Solstad (1996) analyzed the conflict between illegal hunting and wildlife management within east Africa´s national

[12] A well-known example is that species with low intrinsic rates of population increase tend to be more vulnerable to harvest. However, this is not the only factor which would affect vulnerability of species (Bennett and Robinson, 2000).

[13] See, for example, McCullough (1984) and Robinson (2000).

parks in a bioeconomic model with one state variable (wildlife stock) and one control variable (anti-poaching effort). They concluded that, if the local people have the legal rights to reap the benefits of wildlife, more wildlife stock would be safeguarded, because people have the incentive to invest in wildlife stock. Hence, they argued for a shift of conservation policy from protectionism toward utilization of wildlife.

The Skonhoft and Solstad Model is in principle a variant of the Clark´s bioeconomic model (Clark, 1973; Clark and Munro, 1975). In this chapter we will further develop and adequately modify the existing models, in particular the Clark model and the Swanson model, and thereby to investigate deliberately the dynamic development process of resource stock, management efforts, harvest and poaching activity, under the assumption that people are allowed to use renewable resources in protected areas. To do this, a bioeconomic model with one state variable (resource stock) and two control variables (harvest rate, management effort) will be first constructed on the basis of the traditional bioeconomic model and the optimal control theory. An extended model which based on the simple model of this chapter will be then developed in next chapter. What we are concerned about is the question, whether and under which conditions the use of renewable resources in protected areas could really help to induce more human investment in biodiversity conservation and thereby protect them more effectively, as the optimistic conclusion of the Skonhoft and Solstad Model , or just on the contrary, this would lead to species extinction more rapidly. The models may help to offer more detailed answers for these questions and therefore more arguments for the judgement of present conservation policies. The dynamic interaction between control variables and state variable will be investigated deliberately. The policy implications of the models will also be discussed.

6.2 The model

In this section a nonlinear bioeconomic model with one state variable (resource stock) and two control variables (harvest rate, management effort) is employed on the basis of the traditional bioeconomic model and the optimal control theory. The necessary conditions for the optimal policy are also derived. The uniqueness and stability properties of the steady state solution of the model, and the relevant phase diagrams will be presented in the following sections, respectively.

As ever, poaching[14] is a serious problem so that anti-poaching become

[14] In many cases, poaching often means the illegal hunting of wildlife. However, for con-

the most important management issue of many, if not most, park authorities, especially in developing countries. To investigate the dynamic development process of resource stock, management, harvest and poaching activity resulted from the application of the sustainable use approach in protected areas, the present analysis will commence with addressing the economic decision faced by poachers. Milner-Gulland and Leader-Williams (1992), based on experiences of Luangwa Valley in Zambia, addressed the link between the economic decision faced by poachers, law enforcement and poaching activity. They found that several factors will influence the extent to which wildlife are illegally exploited. A greater anti-poaching effort and the following greater detection rate plays the most crucial role in reducing poaching activity. In addition, higher penalties and higher opportunity costs of poaching as a job also work in the same direction. On the other hand, a greater stock of wildlife will induce more poaching activity. In what follows, we will transform the analysis of Milner-Gulland and Leader-Williams into a formal poaching function which is prepared for the later construction of models.

Let $p(Y)$ represents the inverse black-market demand function of illegally exploited resources, where Y is the poached resource stock and $p_Y(Y) < 0$. $M(X, E)$ denotes the costs expended by poachers in exploiting per unit of resource stock, where X is the stock of renewable resources in a given protected area and E is the management effort of park authorities. It is assumed that a larger resource stock means a greater resource density, and thereby reduces the poaching costs, i.e., $M_X < 0$. On the contrary, more management efforts increase the difficulties of poaching activity and also the poaching costs, i.e., $M_E > 0$. In addition, a detection function $q(E)$ is here introduced to represent the probability that poaching activity is successfully detected, where $q_E > 0$ stresses the fact that more management efforts strengthen law enforcement and increase the detection rate. Finally, the severity of penalty given once the poacher has been caught, the variable V representing the fine for per unit poached resource stock, can also have an important effect on poaching activity.

Poachers do not own the legal property rights of resources, or in other words, natural resources are for poachers in an open access state. Therefore, it is reasonable for poachers to exploit resources as possible as they can, i.e., rather than to base on intertemporal profit maximization, their behavior is driven by the motive of exploiting resources until rent dissipates, as Gordon

venience, poaching will be used to mean the illegal exploitation of any kinds of renewable resources throughout this dissertation, including hunting of wildlife, fishing, logging and gathering of plants and of other nontimber products.

(1954) suggested. We define

$$\pi = [1 - q(E)]\,[p(Y) - M(X,E)]\,Y - q(E)YV \tag{6.1}$$

where π is the expected profit function of total poachers, which is the difference between expected revenues and expected costs. Expected revenues are given by the left term and are determined by multiplying successful harvest, $[1 - q(E)]\,Y$, by gross profits per unit of harvest, $[p(Y) - M(X,E)]$. Expected costs derived from law enforcement are given by multiplying detection rate, total harvest and fine per unit of harvest.

Poachers are attracted to exploit resources in protected areas in response to potential profits, and exploitation proceeds until their effort earns its opportunity cost. Their behavior determines the collectively optimal poaching level. The problem is straightforward and easily solved by setting expected profit function equal to zero. It yields

$$[1 - q(E)]\,[p(Y) - M(X,E)] - q(E)V = 0. \tag{6.2}$$

This relationship defines an implicit function $Y(X,E,V)$. The implicit function describes the behavior of poachers associated with poaching, given a resource stock, management effort of park authorities and fine level. It can be analyzed by the following procedure.

First, we differentiate the implicit equation (6.2) with E. After a routine calculation, it yields

$$\frac{dY}{dE} = \frac{1}{p_Y(Y)}\left(M_E(X,E) + \frac{q_E(E)(p(Y) - M(X,E) + V)}{1 - q(E)}\right), \tag{6.3}$$

and from given assumptions it is obvious that

$$\frac{dY}{dE} < 0 \tag{6.4}$$

under the fact that $p(Y)$ must be greater than $M(X,E)$.[15] Again, we differentiate the implicit equation (6.2) with X and V, respectively, and it yields

$$\frac{dY}{dX} = \frac{M_X(X,E)}{p_Y(Y)} > 0 \tag{6.5}$$

and

$$\frac{dY}{dV} = \frac{q(E)}{p_Y(Y)\,[1 - q(E)]} < 0 \tag{6.6}$$

[15] Otherwise, poaching activity will not exist.

under given assumptions.

The results (6.4) and (6.5) show that, other things being equal, a higher level of management effort of park authorities will reduce the poaching. This is because more management effort increase the unit exploitation costs of poachers and the detection rate. On the other hand, a higher resource stock will increase the poaching, as a result that higher resource stock reduces the unit exploitation costs of poachers. These results will be applied later for constructing a new bioeconomic model. In addition, a higher fine level will increase the exploitation costs of poachers, and thereby reduce the poaching.

We consider now the following scenario: certain people or certain organizations[16] are given the legal rights to exploit renewable resources in specific protected areas or in buffer areas around protected areas, and they are authorized to manage natural resources and human activities in those areas. Let $X = X(t)$ represent the biomass of renewable resources at time t. The population dynamics of the resource is assumed to be a pure compensation logistic function of the biomass level, i.e., the natural net growth function of the resource stock, $F(X)$,[17] is a strictly concave function of the biomass level with the following properties

$$F(X) > 0 \text{ for } 0 < X < \bar{X}, \quad F(0) = F(\bar{X}) = 0, \quad F''(X) < 0, \tag{6.7}$$

where $\bar{X}$ represents the carrying capacity of the environment.

Harvest rate $h(t)$ denotes the resource stock which is harvested by the resource owner at time t. The poaching function $Y(E(t))$[18] represents the resource stock which is illegally exploited by poachers at time t, where $E(t)$ is the management effort invested by resource owner and, as equation (6.4) showed,

$$Y'(E) < 0 \tag{6.8}$$

asserts the fact that a higher level of management effort will reduce the poached quantity of resources. In addition, it is assumed that

$$Y''(E) > 0 \tag{6.9}$$

[16]They might be individuals, local communities, private enterprises, park authorities, central governments or non-governmental conservation organizations.

[17]For convenience, the time notation will henceforth be omitted wherever possible.

[18]Rather than $Y = Y(X, E)$, it is assumed here that the poaching rate depends only on management effort level. A complete poaching function will be introduced in the next chapter for modifying the present model.

which explains the phenomenon of diminishing returns of management effort. Both h and E are nonnegative.

In addition to natural factors, the population dynamics of the resource is also subject to human interference. When legal harvest and poaching are introduced, the equation of motion for an exploited resource stock X can be stated as

$$\dot{X} \equiv \frac{dX}{dt} = F(X) - Y(E) - h. \tag{6.10}$$

Now let us consider the functional form of the utility function. It has long been recognized that wildlands and renewable resources may produce a range of valued products and services, and in turn provide benefits to human society (Barbier, 1992; Aylward,1992). Barbier (1992) suggested that the total economic value of wildlife and wildlands may comprise use value, which includes direct value, indirect value and option value, and non-use value. Under this definition the direct use value may includes benefits from harvesting, recreation, tourism, genetic material, education etc. Hence, as a function of harvest, the gross harvest profit function $U(h)$ usually used in the traditional bioeconomic model captures obviously only part of the total economic value and even only part of the direct use value, namely the consumptive value in the narrow sense. We use here the term ´consumptive value´ to refer to the direct use values provided by directly harvesting renewable resources. For convenience of analysis, we divide the economic value of renewable resources into consumptive value and non-consumptive value which refers to all forms of values that can not be classified as consumptive value. In other words, the resource stock remains in principle intact when it provides non-consumptive value.[19] In what follows, a new utility function

$$U(h, X) = U(h) + V(X) \tag{6.11}$$

which consider simultaneously the consumptive value and non-consumptive value is here introduced. The utility function is assumed to be additively separable for the sake of technical simplicity. The gross harvest profit function $U(h)$, which represents the total harvest revenues less the total harvest costs and comprises the consumptive value component of the utility function, depends on the harvest rate and is strictly increasing and concave in h,

[19] In some cases, for example usually happened in tourism, even non-consumptive use has potentially negative impacts on resource stock, although these impacts often can not be observed directly.

namely

$$U'(h) > 0, \qquad U''(h) < 0. \tag{6.12}$$

Under the assumption of $U'(h) > 0$, we can avoid the possibility of saturation in harvest for extremely large harvest level and the relevant technical complexity (Boyce, 1995). The concavity of $U(h)$ derives from the both assumptions of decreasing marginal revenues in harvest and increasing marginal harvest costs. Next, the non-consumptive utility function $V(X)$ which comprises the non-consumptive value component of the utility function depends on the level of the resource stock. We assume that $V(X)$ can be measured in monetary terms and is strictly increasing and concave in X:[20]

$$V'(X) > 0, \qquad V''(X) < 0. \tag{6.13}$$

It is here worth noting that, due to the public good characteristics of many of the non-consumptive values, the introduction of non-consumptive value does not mean that resource owner is able to appropriate the total economic value of renewable resources. In general, international conservation organizations may tend to take all values into consideration. National governments may, to certain extent, fail to capture part of the ecological function, existence and option value. And for private ranges or parks, it is likely that ecological function, option and existence value would not be taken into account in the process of decision-making.

The resource-management problem of the resource owner is to choose the harvest rate and the management effort to maximize the net present profits derived from protecting and harvesting the resource, subject to the constraint of biological dynamics. This task can be formally expressed as

$$Max \int_0^\infty \left[U(h) + V(X) - C(E)\right] e^{-rt} dt$$

$$s.t. \ \dot{X} = F(X) - Y(E) - h \tag{6.14}$$

where $C(E)$ and r denote the management cost function and the instantaneous discount rate, respectively. The management costs function owns the properties that

$$C'(E) > 0, \qquad C''(E) > 0, \tag{6.15}$$

[20]For example, one can imagine that the more spectacular the wildlife population in a national park is, the more tourists will be induced and in turn the more income generated. However, for per one unit of additional resource stock, the marginal utility will be decreasing. The same statement can be applied to cases in reference to other non-consumptive values, such as ecological functions, option value and existence value.

which implies that $C(E)$ is strictly increasing and convex in E.[21] The discount rate is assumed to be constant and $0 < r < 1$.

Next, the problem is analyzed by the application of the maximum principle. The current-value Hamiltonian of our case is

$$H = U(h) + V(X) - C(E) + \lambda \left[F(X) - Y(E) - h\right] \tag{6.16}$$

where λ is the current-value costate variable associated with the state variable X which gives the imputed marginal value of the resource stock. Assuming an interior solution, the first-order necessary conditions describing the optimization problem are given by equations (6.17)-(6.19) together with (6.10):

$$\frac{\partial H}{\partial h} = U'(h) - \lambda = 0 \tag{6.17}$$

$$\frac{\partial H}{\partial E} = -C'(E) - \lambda Y'(E) = 0 \tag{6.18}$$

$$\dot{\lambda} = r\lambda - \frac{\partial H}{\partial X} = \left[r - F'(X)\right] \lambda - V'(X). \tag{6.19}$$

Under given assumptions, the Hamiltonian is apparently concave in the state variables X and in the control variables h, E. The second order conditions are therefore satisfied. Along the optimal trajectory, the equation (6.17) means that the imputed marginal value, or the shadow price of an extra resource stock, λ, must be equal to the marginal profit of harvesting renewable resources, $U'(h)$. The equation (6.18) implies that the marginal gain derived from an extra unit of management effort, $-\lambda Y'(E)$, must equal the marginal cost of management input, $C'(E)$. Finally, equation (6.19) asserts that the summation of the change rate of the shadow price of the resource stock, $\dot{\lambda}$, the marginal non-consumptive value of the resource stock, $V'(X)$, and the gain derived from the marginal growth rate of the resource stock, $F'(X)\lambda$, must equal the opportunity cost when resource owner goes on to keep one unit of resource stock, $r\lambda$. The term $r\lambda$ represents the forgone ´interest´.

[21] According to author´s personal experiences in Taiwan, the assumptions of the concavity of $U(h)$ and the convexity of $C(E)$ are reasonable, because the market of renewable resources is usually relatively small and the market of manpower for conservation, especially at lower level, is typically localized.

6.3 Uniqueness of the steady state solution

In the following sections, we will investigate the properties of the steady state solution of the dynamic problem. Suppose that a steady state exists, the steady state solution can be determined under the conditions that the resource stock and the shadow price of the resource are constant, i.e., $\dot{X} = \dot{\lambda} = 0$. We can first verify the uniqueness of the steady state solution by proposition 1.

Proposition 1 *Under given assumptions with regard to the poaching function $Y(E)$, the utility function $U(h) + V(X)$, the management cost function $C(E)$ and the natural dynamics of the resource stock $F(X)$, the dynamic system (6.10) and (6.19) possess an unique steady state solution.*

Proof. In steady state, $\dot{X} = \dot{\lambda} = 0$. First, $\dot{\lambda} = 0$ gives

$$\lambda F'(X) + V'(X) = r\lambda$$

by applying the equation (6.19). Under given assumptions, we know that both $F'(X)$ and $V'(X)$ are monotonously decreasing in X. Therefore, the function $\lambda F'(X)+V'(X)$ is also monotonously decreasing in X. It is obvious that one unique X^* exists which satisfying the condition $\lambda F'(X^*)+V'(X^*) = r\lambda$ where X^* denotes the steady state solution of the resource stock X.

From (6.18), we know that

$$C'(E) = -U'(h)Y'(E). \tag{6.20}$$

Consider E as an implicit function of h. After substituting $E = E(h)$ for equation (6.20), and differentiating (6.20) with h, it yields

$$[C''(E) + U'(h)Y''(E)] \frac{dE}{dh} = -U''(h)Y'(E). \tag{6.21}$$

Under given assumptions, it can be easily shown that

$$\frac{dE}{dh} < 0. \tag{6.22}$$

Applying $E = E(h)$ for the case when $\dot{X} = 0$, it implies that

$$F(X^*) - Y(E(h)) - h = 0$$

in steady state. By defining

$$\Omega(h) = F(X^*) - Y(E(h)) - h \tag{6.23}$$

and differentiating $\Omega(h)$ with h, we obtain

$$\Omega'(h) = -Y'(E)E'(h) - 1 < 0, \tag{6.24}$$

because $E'(h) < 0$, as equation (6.22) shows. Therefore, $\Omega(h) = 0$ possesses an unique solution h^*. Since $E = E(h)$, $E'(h) < 0$, $\lambda = U'(h)$ and $U''(h) < 0$, it is obvious that both $E(h)$ and $U'(h)$ are monotonously decreasing function of h. It follows that $E^* = E(h^*)$ and $\lambda^* = U'(h^*)$ are also unique. Consequently, the steady state solution (X^*, λ^*, h^*, E^*) is unique. ■

6.4 Stability of the steady state solution

Proposition 2 *Under given assumptions with regard to the poaching function $Y(E)$, the utility function $U(h) + V(X)$, the management cost function $C(E)$ and the natural dynamics of the resource stock $F(X)$, the unique steady state solution of the dynamic system (6.10) and (6.19) is saddle point stable, if $F'(X)\,[r - F'(X)] - [-F''(X)\lambda - V''(X)]\,[-Y'(E)E'(\lambda) - h'(\lambda)] < 0$.*

Proof. By use of the result $\lambda = U'(h)$ of the equation (6.17), we can define h as an implicit function of λ. Since

$$\frac{dh}{d\lambda}\frac{d\lambda}{dh} = 1$$

or

$$\frac{dh}{d\lambda}U''(h) = 1$$

,it can be easily verified that

$$\frac{dh}{d\lambda} < 0. \tag{6.25}$$

Again, by applying the result $-C'(E) = \lambda Y'(E)$ of the equation (6.18), we can define E as an implicit function of λ. Since

$$\frac{dE}{d\lambda}\frac{d\lambda}{dE} = 1$$

and

$$\frac{d\lambda}{dE} = \frac{-C''(E)Y'(E) + C'(E)Y''(E)}{Y''(E)} > 0,$$

it yields

$$\frac{dE}{d\lambda} > 0. \tag{6.26}$$

After substituting $h(\lambda)$ for h and $E(\lambda)$ for E in equation (6.10), we observe now the following dynamic system of equations:

$$\begin{aligned} \dot{X} &= F(X) - Y(E(\lambda)) - h(\lambda) \\ \dot{\lambda} &= [r - F'(X)]\,\lambda. \end{aligned} \tag{6.27}$$

For the purpose of later analysis, we differentiate $\dot{X}$ and $\dot{\lambda}$ with X and λ, respectively. This yields

$$\begin{aligned} \frac{\partial \dot{X}}{\partial X} &= F'(X) \\ \frac{\partial \dot{X}}{\partial \lambda} &= -Y'(E)E'(\lambda) - h'(\lambda) \\ \frac{\partial \dot{\lambda}}{\partial X} &= -F''(X)\lambda - V''(X) \\ \frac{\partial \dot{\lambda}}{\partial \lambda} &= r - F'(X). \end{aligned} \tag{6.28}$$

A Taylor expansion of the dynamic system (6.27) at (X^*, λ^*) gives then

$$\begin{pmatrix} \dot{X} \\ \dot{\lambda} \end{pmatrix} = \begin{pmatrix} F'(X^*) & -Y'(E)E'(\lambda^*) - h'(\lambda^*) \\ -F''(X^*)\lambda^* - V''(X^*) & r - F'(X^*) \end{pmatrix} \begin{pmatrix} X - X^* \\ \lambda - \lambda^* \end{pmatrix} \tag{6.29}$$

, after the results of (6.28) are applied for substituting $\frac{\partial \dot{X}}{\partial X}$, $\frac{\partial \dot{X}}{\partial \lambda}$, $\frac{\partial \dot{\lambda}}{\partial X}$ and $\frac{\partial \dot{\lambda}}{\partial \lambda}$ in (6.29). Hence, the two eigenvalues $\frac{r \pm \sqrt{r^2 - 4A}}{2}$ can be easily derived where $A = F'(X^*)\,[r - F'(X^*)] - [-F''(X^*)\lambda^* - V''(X^*)]\,[-Y'(E)E'(\lambda^*) - h'(\lambda^*)]$. If $A < 0$, it is obvious that one of the eigenvalues is positive, and the other one is negative. Therefore, the steady state solution (X^*, λ^*) is a saddle point. ■

The saddle-point stability property of the steady state solution means that, given the initial value of the resource stock, it will always be possible for the resource owner to choose an optimal initial value of the harvest rate and the management effort which are on the stable trajectories that converge to the steady state equilibrium of the dynamic system.

6.5 Phase diagram analysis

6.5.1 Phase diagram (X, h)

In this section, we conduct a phase diagram analysis to investigate the qualitative properties of the solution of the dynamic problem. Let us observe first the phase diagram on the (X, h) plane.

First, we differentiate the necessary condition (6.17) with t and obtain

$$\dot{\lambda} = U''(h)\dot{h}. \tag{6.30}$$

Applying the results of equations (6.17) and (6.30) in (6.19), it yields that

$$\dot{h} = \frac{1}{U''}\left[U'\left(r - F'\right) - V'\right]. \tag{6.31}$$

By means of setting $\dot{h} = 0$, the isocline $\dot{h} = 0$ is the curve

$$U'\left(r - F'\right) - V' = 0 \tag{6.32}$$

as a result of the assumption $U'' < 0$. A total differentiation of (6.32) gives then

$$\frac{dh}{dX}\mid_{\dot{h}=0} = \frac{U'F'' + V''}{U''(r - F')} \tag{6.33}$$

Under given assumptions, it is clear that the term $U'F''+V''$ is negative. The denominator term $U''(r-F')$ is positive when X is extremely small, and it is negative when X is big enough because F' is monotonously decreasing and W' monotonously increasing in X. Accordingly, the gradient of the isocline $\dot{h} = 0$ on the (X, h) plane is negative on the left of X_r and positive on the right of X_r, where X_r denotes the resource stock so that $F'(X_r) = r$ (see Figure 6.1). It is worth noting that, in our case, the equilibrium resource stock must be greater than X_r, because of the fact that the non-consumptive value of the resource is taken into account.

Similarly, by the use of (6.10), it is obvious that

$$\dot{X} = F(X) - Y(E(h)) - h. \tag{6.34}$$

By setting $\dot{X} = 0$, the isocline $\dot{X} = 0$ is the curve

$$F(X) - Y(E(h)) - h = 0. \tag{6.35}$$

A total differentiating of (6.35) yields

$$\frac{dh}{dX} \Big|_{\dot{X}=0} = \frac{F'(X)}{1 + Y'(E)E'(h)}. \tag{6.36}$$

It is clear that the gradient of the isocline $\dot{X} = 0$ is strictly decreasing in X. The two isoclines determine then the unique steady state solution $(X^*,\ h^*)$.

Next, we examine the properties of the points which are not on the isoclines. As Figure 6.1 shows, the phase plane (X, h) can be divided into four isosectors by the two isoclines. By differentiating equation (6.31) with X, it yields

$$\frac{d\dot{h}}{dX} = \frac{-U'(h)F''(X) - V''(X)}{U''(h)} < 0 \tag{6.37}$$

under given assumptions. This means that, in the region on the left of the isocline $\dot{h} = 0$, h tends to increase, with a symbolic upward vertical pointing arrow. On the contrary, in the region on the right of the isocline $\dot{h} = 0$, h tends to decrease, with a symbolic downward vertical pointing arrow. Again, by differentiating equation (6.34) with h, it gives

$$\frac{d\dot{X}}{dh} = -Y'(E)E'(h) - 1 < 0. \tag{6.38}$$

Therefore, with a rightward horizontal pointing arrow, X tends to increase in the region below the isocline $\dot{X} = 0$. And, with a leftward horizontal pointing arrow, X tends to decrease in the region above the isocline $\dot{X} = 0$. These results outlined above show that the equilibrium point is a saddle point. The two solid trajectories in Figure 6.1, which denote the two stable trajectories, converge to the equilibrium point. Corresponding to each initial resource stock level, a unique corresponding value of harvest rate could be chosen on the stable trajectories. Hence, the following conclusion can be drawn that, on the optimal dynamic path, the resource stock and the harvest rate increase over time, if the initial resource stock level is less than the steady state resource stock. Contrarily, if the initial resource stock level is higher than the steady state resource stock, the resource stock and the harvest rate decrease over time on the optimal dynamic path. The economic meaning of this result is clear. The more resource stock people have, the more they would harvest without influencing the long-run survival of the resource, or in other words, in order to reach the steady state, people would harvest more than the steady state harvest rate when the resource is in relative abundance, in the sense that the resource stock is greater than the steady state resource stock. And they would harvest less than the steady state harvest rate when the resource is relatively scarce, in the sense that the resource stock is less than the steady state resource stock.

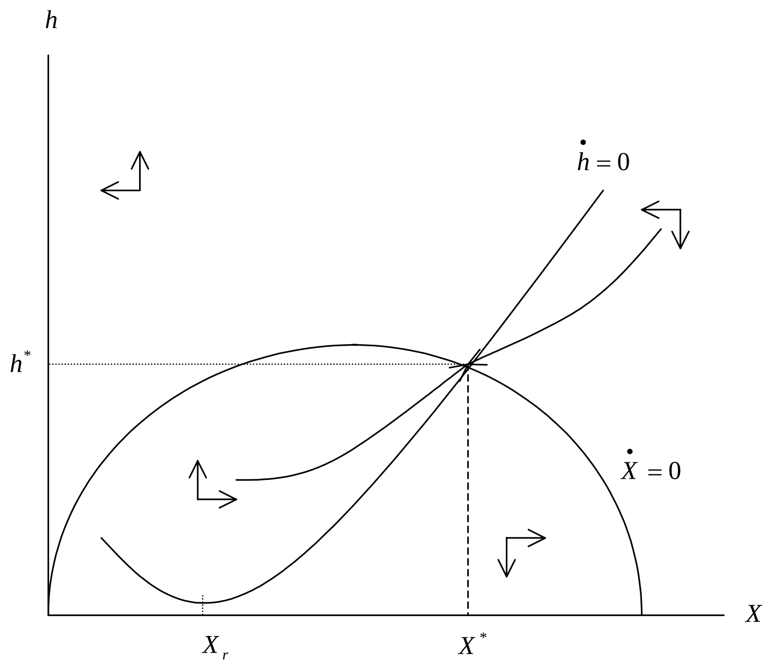


Figure 6.1 Phase diagram on the (X,h) plane.

6.5.2 Phase diagram (X, E)

We observe now the phase diagram on the (X, E) plane. A transformation of the equation (6.18) yields

$$\lambda Y'(E) = -C'(E). \tag{6.39}$$

Differentiating (6.39) with t and after a few transformations, we obtain

$$\dot{\lambda} = \frac{\left[-C''(E) - \lambda Y''(E)\right] \dot{E}}{Y'(E)}. \tag{6.40}$$

By substituting this result for $\dot{\lambda}$ in equation (6.19) and substituting $U'(h)$ for λ, it gives

$$\dot{E} = \frac{\left[\left[r - F'(X)\right] U'(h) - V'(X)\right] Y'(E)}{-C''(E) - U'(h) Y''(E)}. \tag{6.41}$$

Considering h as an implicit function of E and substituting $h(E)$ for h in (6.10), it yields

$$\dot{X} = F(X) - Y(E) - h(E). \tag{6.42}$$

We are now interested on the dynamic system (6.41) and (6.42). By means of setting $\dot{E} = 0$, the isocline $\dot{E} = 0$ is the curve

$$\left[r - F'(X)\right] U'(h(E)) - V'(X) = 0 \tag{6.43}$$

under given assumptions. A total differentiating of (6.43) gives

$$\frac{dE}{dX} \bigg|_{\dot{E}=0} = \frac{U'(h)F''(X) + V''(X)}{U''(h)h'(E)\left[r - F'(X)\right]}.$$

Under given assumptions, it is clear that the term $U'(h)F''(X) + V''(X)$ is negative. The denominator term $U''(h)h'(E)\left[r - F'(X)\right]$ is negative when X is extremely small, and it is positive when X is big enough because F' is monotonously decreasing in X. Accordingly, the gradient of the isocline $\dot{E} = 0$ on the (X, E) plane is positive on the left of X_r and negative on the right of X_r, where X_r denotes the resource stock so that $F'(X_r) = r$ (see Figure 6.2). Similarly, by setting $\dot{X} = 0$, the isocline $\dot{X} = 0$ is the curve

$$F(X) - Y(E) - h(E) = 0. \tag{6.44}$$

A total differentiating of (6.44) gives

$$\frac{dE}{dX} \bigg|_{\dot{X}=0} = \frac{F'(X)}{Y'(E) + h'(E)}. \tag{6.45}$$

The gradient of the isocline $\dot{X} = 0$ is negative when $F'(X) > 0$, and then strictly increasing in X. It becomes positive when $F'(X) < 0$. The two isoclines determine the unique steady state solution (X^*, E^*).

Again let us examine the properties of the points which are not on the isoclines. As Figure 6.2 shows, the phase plane (X, E) can be divided into four isosectors by the two isoclines. By differentiating equation (6.41) with X, it yields

$$\frac{d\dot{E}}{dX} = \frac{[-F''(X)U'(h) - V''(X)]\, Y'(E)}{-C''(E) - U'(h)Y''(E)} > 0 \tag{6.46}$$

under given assumptions. This means that, in the region on the left of the isocline $\dot{E} = 0$, E tends to decrease, with a symbolic downward vertical pointing arrow. On the contrary, in the region on the right of the isocline $\dot{E} = 0$, E tends to increase, with a symbolic upward vertical pointing arrow. Again, by differentiating equation (6.42) with E, it gives

$$\frac{d\dot{X}}{dE} = -Y'(E) - h'(E) > 0, \tag{6.47}$$

as a result of $h'(E) < 0$, according to (6.22). Consequently, with a leftward horizontal pointing arrow, X tends to decrease in the region below the isocline $\dot{X} = 0$. And, with a rightward horizontal pointing arrow, X tends to increase in the region above the isocline $\dot{X} = 0$. These results outlined above show that the equilibrium point is a saddle point. The two solid trajectories in Figure 6.2, which denote the two stable trajectories, converge to the equilibrium point. Corresponding to each initial resource stock level, a unique corresponding value of management effort could be chosen on the stable trajectories. Hence, the following conclusion can be drawn that, on the optimal dynamic path, the resource stock increases over time while the management effort input decreases, if the initial resource stock level is less than the steady state resource stock. Contrarily, if the initial resource stock level is greater than the steady state resource stock, the resource stock decreases over time on the optimal dynamic path while the management effort input increases. The economic meaning of this result is as follows. The more resource stock people have, the less management effort would be needed, or in other words, in order to reach the steady state, people would devote less management effort than the steady state management effort level when the resource is in relative abundance, in the sense that the resource stock is greater than the steady state resource stock. And they would invest more management effort than the steady state management effort level when the resource is relatively scarce, in the sense that the resource stock is less than the steady state resource stock. With reference to the poaching rate, we know that it will increase as the management effort input decreases, and vice versa. It follows that the poaching rate will change in the same direction as the resource stock does.

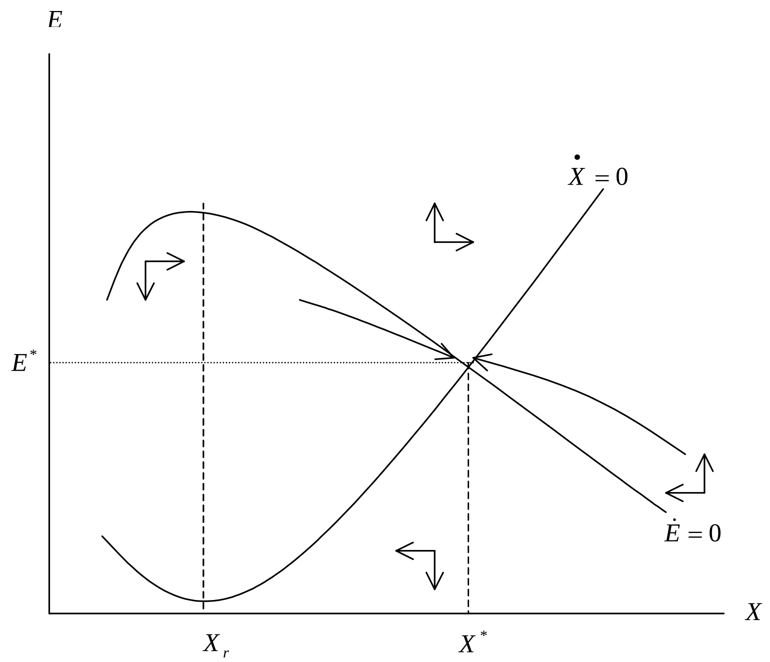


Figure 6.2 Phase diagram on the (X,E) plane.

6.6 Comparative static analysis

In order to investigate the influence of the exogenous variables on the equilibrium resource stock, we conduct here a comparative static analysis by introducing a particular specification for the natural growth function of the resource stock $F(X)$. We specify

$$F(X) = \rho X(1 - X) \tag{6.48}$$

where the exogenous coefficient ρ denotes the intrinsic growth rate of the resource stock. This means that the growth rate of the resource stock will approach ρ when the resource stock level is extremely small, i.e.

$$\lim_{X \to 0} \frac{F(X)}{X} = \rho \tag{6.49}$$

(Wacker and Blank, 1998), or in other words, the intrinsic growth rate of the resource stock is the highest growth rate that a population can reach, if it is not subject to food, space, resource competition and predation. This specification also implies that the carrying capacity of the environment $\bar{X}$ equals 1 and X_{MSY} equals $\frac{1}{2}$, where X_{MSY} denotes the stock level which can afford the Maximum Sustainable Yield. The fundamental concavity property of the Hamiltonian of the optimization problem and the uniqueness together with the stability of the steady state solution remain unchanged under this specification.

The specific version of the simple model results in the following dynamic system after some rearrangements:

$$\begin{aligned} \dot{X} &= \rho X(1-X) - Y(E(\lambda)) - h(\lambda) \\ \dot{\lambda} &= (r - \rho + 2\rho X)\,\lambda - V'(X). \end{aligned} \tag{6.50}$$

The system encompasses two endogenous variables, X and λ, and two exogenous variables, r and ρ. By taking the total differential of the system (6.50), it yields

$$\begin{aligned} &\begin{pmatrix} \rho - 2\rho X & -Y'E' - h' \\ 2\rho\lambda - V''(X) & r - \rho + 2\rho X \end{pmatrix} \cdot \begin{pmatrix} dX^* \\ d\lambda^* \end{pmatrix} \\ = &\begin{pmatrix} -X + X^2 & 0 \\ \lambda - 2X\lambda & -\lambda \end{pmatrix} \cdot \begin{pmatrix} d\rho \\ dr \end{pmatrix} \end{aligned} \tag{6.51}$$

By application of Cramer´s rule, the following results of the comparative static analysis can be derived:

$$\frac{dX^*}{dr} = \frac{A}{|J|}$$

$$\frac{dX^*}{d\rho} = \frac{B}{|J|}$$

where

$$J = \begin{pmatrix} \rho - 2\rho X & -Y'E' - h' \\ 2\rho\lambda - V''(X) & r - \rho + 2\rho X \end{pmatrix},$$

$$A = \lambda(-Y'E' - h') > 0$$

$$\text{and } B = (-X + X^2)(r - \rho + 2\rho X) - (\lambda - 2X\lambda)(-Y'E' - h').$$

Under the assumption of the proposition 2, we know that $|J| < 0$. It is also clear that $A > 0$ under given assumptions. It yields then

$$\frac{dX^*}{dr} < 0. \tag{6.52}$$

From (6.19), we know that $r - \rho + 2\rho X = \frac{V'(X)}{\lambda} > 0$ in steady state. The term $-X + X^2 = -\frac{F(X)}{\rho} < 0$. By applying these results, it can be easily shown that $B < 0$, if $X^* < \frac{1}{2}$ or $X^* = \frac{1}{2}$, and the sign of B is ambiguous , if $X^* > \frac{1}{2}$. This gives the following results:

$$\frac{dX^*}{d\rho} > 0, \text{ if } X^* < \frac{1}{2} \text{ or } X^* = \frac{1}{2}, \tag{6.53}$$

$$\text{and } \frac{dX^*}{d\rho} \text{ is ambiguous, if } X^* > \frac{1}{2}.$$

The outcome of (6.52) shows that an increase in the discount rate r will lower the equilibrium resource stock. This is because a higher discount rate raises the opportunity cost of holding resource stock, as the term $r\lambda$ of (6.19) shows, and constitutes motives for stock disinvestment. (6.53) demonstrates that an increase in the intrinsic growth rate of the resources ρ will raise the equilibrium resource stock, if the equilibrium resource stock is smaller than or equals to the maximum sustainable yield stock level. The underlying reason for this result is clear. The maximum sustainable yield stock level equals $\frac{1}{2}$ under the special specification. If $X^* < \frac{1}{2}$, an increase in ρ will raise the overall level of the marginal growth rate of the resources $F'(X^*) = \rho - 2\rho X^*$, and thereby raise the return of keeping resource stock. Hence, the resource owner will be willing to hold a higher resource stock. On the other hand, if $X^* > \frac{1}{2}$, an increase in ρ will lower the overall level of the marginal growth rate of the resources, and cause a disincentive for keeping resource stock. This leads to an ambiguously total effect.

6.7 A special case of the simple model

We investigate in this section a special case of the simple model, namely, that the non-consumptive value of the resource will not be taken into account. In this case, the utility function will have the functional form $U(h)$, while all other functions remain unchanged. It follows that the resource owner has the resource management problem

$$Max \int_0^\infty [U(h) - C(E)] \, e^{-rt} dt$$

$$s.t. \ \dot{X} = F(X) - Y(E) - h. \tag{6.54}$$

The relevant first-order necessary conditions can be easily derived:

$$\frac{\partial H}{\partial h} = U'(h) - \lambda = 0 \tag{6.55}$$

$$\frac{\partial H}{\partial E} = -C'(E) - \lambda Y'(E) = 0 \tag{6.56}$$

$$\dot{X} = F(X) - Y(E) - h \tag{6.57}$$

$$\dot{\lambda} = r\lambda - \frac{\partial H}{\partial X} = [r - F'(X)] \, \lambda. \tag{6.58}$$

These conditions have the same economic meaning as those presented in section 6.2. Under given assumptions, it can also be easily verified that one unique steady state solution exists, and the steady state solution is saddle point stable.

Proposition 3 *Under given assumptions with regard to the poaching function $Y(E)$, the utility function $U(h)$, the management cost function $C(E)$ and the natural dynamics of the resource stock $F(X)$, the dynamic system (6.57) and (6.58) possess an unique steady state solution.*

Proof. See appendix 6.1. ■

Proposition 4 *Under given assumptions with regard to the poaching function $Y(E)$, the utility function $U(h)$, the management cost function $C(E)$ and the natural dynamics of the resource stock $F(X)$, the unique steady state solution of the dynamic system (6.57) and (6.58) is saddle point stable.*

Proof. See appendix 6.2. ■

Next, by applying the same method used in section 6.5, the phase diagrams on the X-h plane and on the X-E plane can be derived, respectively (see Figure 6.3 and 6.4). The phase diagrams are similar to those of the simple model, except that the isocline $\dot{h} = 0$ in figure 6.3 is a vertical line through X^* which parallels the h-axis, and the isocline $\dot{E} = 0$ in figure 6.4 is a vertical line through X^* which parallels the E-axis. These differences can be attributed to the condition (6.58) which implies that

$$F'(X^*) = r. \tag{6.59}$$

It is worth noting that, in this special model, the steady state resource stock X^* is always less than the stock level which can afford the Maximum Sustainable Yield (MSY), because of the condition (6.59). In other words, the resource stock is worth being preserved, from the point of view of the resource owner, only when it´s marginal growth rate can compete with the discount rate. Obviously, this is not possible for a resource stock level greater than X_{MSY}, where X_{MSY} denotes the stock level which can afford the Maximum Sustainable Yield. Except these differences, figure 6.3 and 6.4 demonstrate similar interactions between X and h and between X and E on the stable trajectories, as demonstrated by figure 6.1 and 6.2.

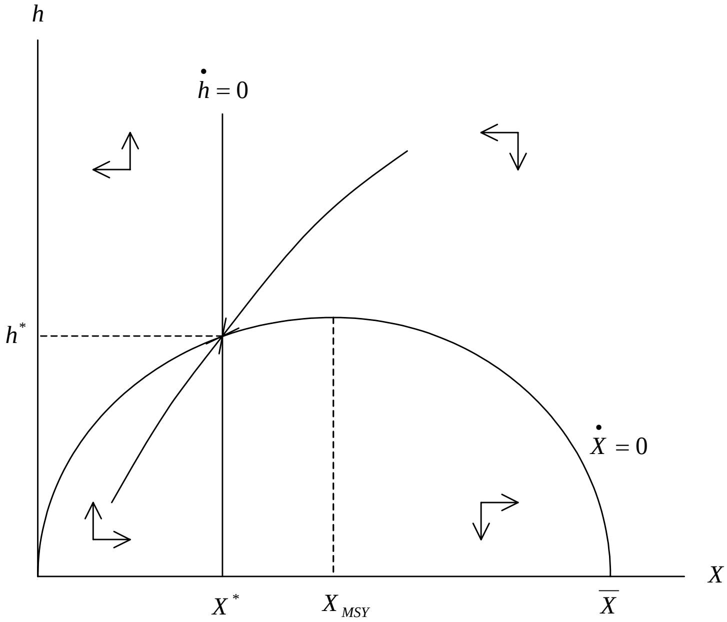


Figure 6.3 Phase diagram on the (X,h) plane.

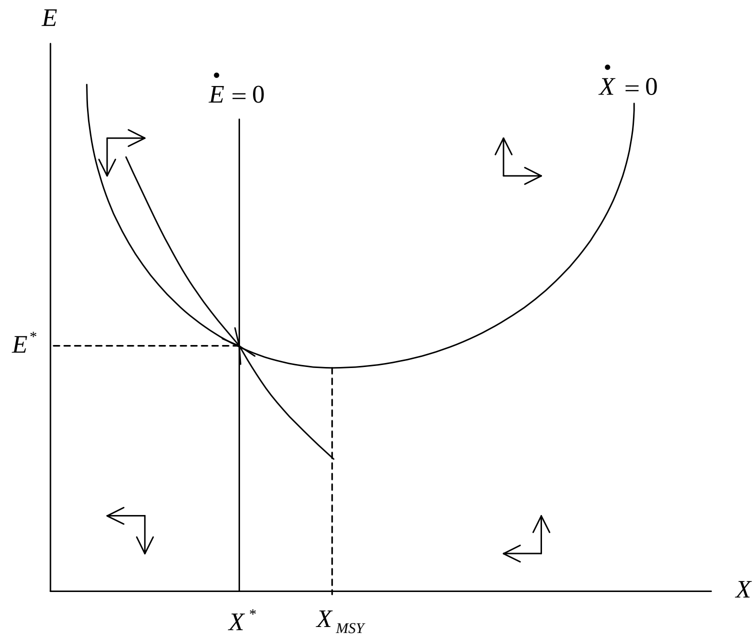


Figure 6.4 Phase diagram on the (X,E) plane.

Finally, let us examine the comparative static effects of the special model by using the specification of (6.48). Under this specification, the equilibrium resource stock level

$$X^* = \frac{\rho - r}{2\rho}, \tag{6.60}$$

as a result of the condition (6.59). Hence, the equilibrium resource stock depends on the two exogenous variables r and ρ. It can be easily verified that

$$\begin{aligned} \frac{\partial X^*}{\partial r} &= -\frac{1}{2\rho} < 0, \text{ and} \\ \frac{\partial X^*}{\partial \rho} &= \frac{2r(1-\rho)}{4\rho^2} > 0. \end{aligned} \tag{6.61}$$

These outcomes mean that an increase in discount rate will lower the equilibrium resource stock level, since the marginal growth rate of the resource stock $F'(X)$ must compete with a higher discount rate in steady state, and only a decrease in the equilibrium resource stock level can lead to a higher marginal growth rate. On the other hand, a higher intrinsic growth rate of the resource stock ρ will raise the overall level of $F'(X) = \rho - 2\rho X$, if $X < \frac{1}{2} = X_{MSY}$. This is exactly the case as our model, since $X^* = \frac{\rho - r}{2\rho} < \frac{1}{2}$ in our special model. Other things being equal, the resource owner will be willing to keep a higher equilibrium resource stock level, because the resource, as a kind of natural capital, becomes more productive.

6.8 Concluding remarks and policy implication

In this chapter we have developed a nonlinear bioeconomic model with one state variable (resource stock) and two control variables (harvest rate, management effort) to investigate the dynamic development process of resource stock, harvest rate, management effort input and poaching rate, under the premise that people are allowed to use and manage renewable resources in or around protected areas. The economic motive of the poachers is analyzed to construct the basis of the poaching function. Under some general assumptions with regard to the poaching function, the utility function, the management cost function and the natural dynamics of the resource stock, it can be verified that the steady state solution of the dynamic problem is unique and saddle point stable. The optimal time paths of the resource stock, harvest rate and management effort input are also depicted in relevant phase diagrams. By the help of a specification with reference to the natural growth function of the resource stock, we can identify two critical exogenous variables, the discount rate and the intrinsic growth rate of the resource stock, which will influence the equilibrium resource stock level. In addition, a special case of the model, which considers only the consumptive value of the resource, is also examined.

From the point of view of the conservation policy, the implication of the theoretical model is as follows. First, the quantity of the equilibrium resource stock depends on the intrinsic growth rate of resource and on the discount rate. The smaller the discount rate is, the higher the equilibrium resource stock level will be, and vice versa, while the comparative static effect of the intrinsic growth rate on the equilibrium resource stock is ambiguous, if the equilibrium resource stock is greater than the maximum sustainable yield stock level. However, the special case of the model in which the non-consumptive value of the resource is not considered demonstrates an unambiguous comparative static effect of the intrinsic growth rate of resource. Therefore, these two variables are good indicators for the assessment of the sustainable use strategy in specific cases. The use approach might potentially be more appropriate in a case with low discount rate and high intrinsic growth rate of resource than another cases with high discount rate and low intrinsic growth rate of resource.

Next, the impact of the sustainable use approach on conservation is double-edged, in the sense that the sustainable use approach will not necessarily result in a higher stock level of renewable resources. On the one hand, the sustainable use approach will theoretically contribute to better management of protected areas, decrease of poaching activity and the following increase in resource stock, if the initial resource stock is less than the equilibrium resource stock and no or only little management capacity exists initially.[22] On the other hand, if the initial resource stock is greater than the equilibrium resource stock, the use approach will inevitably lead to a decrease in resource stock through the adjustment of harvest rate until the equilibrium resource stock is reached, even though the resource owner will, to certain extent, invest simultaneously in management to control the poaching activity. Moreover, as the special case of our model predicts, it is notable that the equilibrium resource stock level is always smaller than the stock level which can afford the maximum sustainable yield. Certainly, in some cases, this equilibrium resource stock level X^* will not be necessarily very small in comparison with the carrying capacity $\bar{X}$, as the left-skewed logistic growth function in Figure 6.5 shows.[23] However, it is also absolutely possible that the equilibrium resource stock is small enough in some cases that people will be seriously concerned about the viability of the resource and the loss of it´s ecological functions. Even in the case of our original model, in which the non-consumptive value of the resource is taken into account so that the equilibrium resource stock is not necessarily smaller than the maximum sustainable yield stock, such possibility cannot be excluded, if the non-consumptive value of the resource is negligible. Apparently, in order to decide whether the use approach is appropriate in specific cases, we need to know, at least roughly, the relevant functional forms and parameter values, and simulate the possible scenarios before any decision is made. A similar trial will be conducted after the model presented here is extended in next chapter.

[22] An implicit but important premise for this conclusion is, that the use strategy can really generate positive or at least zero discounted net profit. Otherwise the use approach will not work at all.

[23] Such cases could happen if the resource stock increases at a relatively steady marginal growth rate, and the marginal growth rate will decrease apparently only when the resource stock comes near the carrying capacity of the environment. It happens usually in species in which animals do not breed until relatively late in life. Robinson and Redford (1991) suggested that, generally, wildlife species reach their maximum productivity when population densities are close to the range of 65% to 90% of carrying capacity.

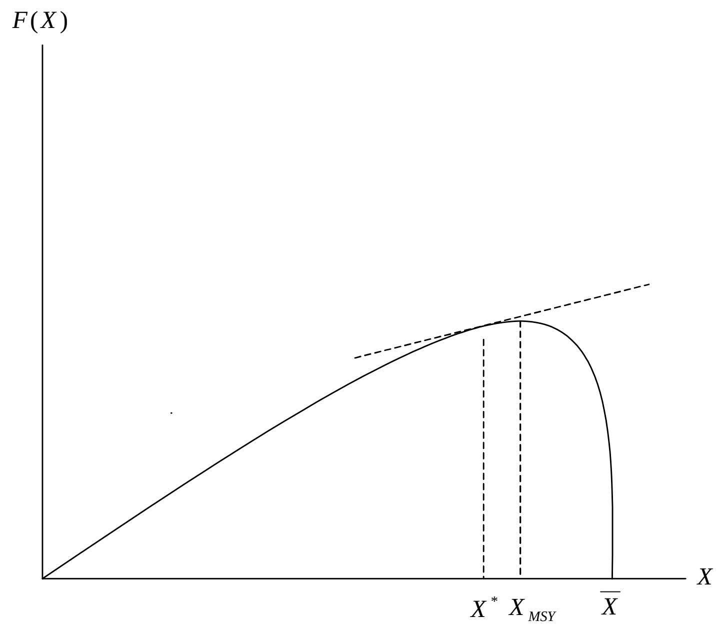


Figure 6.5 Left-skewed logistic growth function.

Appendix 6.1

From (6.58), we know that if $\dot{\lambda} = 0$, it gives

$$r - F'(X) = 0. \tag{A.6.1}$$

Therefore, the steady state solution of the resource stock X must satisfy the condition

$$F'(X^*) = r \tag{A.6.2}$$

where X^* denotes the steady state solution of the resource stock X. Since $F'(X)$ is a monotonously decreasing function of X, it is obvious that X^* is unique. From (6.56) and (6.55), we know that

$$C'(E) = -U'(h)Y'(E). \tag{A.6.3}$$

Consider E as an implicit function of h. Substituting $E = E(h)$ for equation (A.6.3), and differentiating (A.6.3) with h, it yields

$$[C''(E) + U'(h)Y''(E)]\,\frac{dE}{dh} = -U''(h)Y'(E).$$

Under given assumptions, it can be easily shown that

$$\frac{dE}{dh} < 0. \tag{A.6.4}$$

Applying $E = E(h)$ for equation (6.57), it implies that

$$F(X^*) - Y(E(h)) - h = 0$$

in steady state. By defining

$$\Omega(h) = F(X^*) - Y(E(h)) - h$$

and differentiating $\Omega(h)$ with h, we obtain

$$\Omega'(h) = -Y'(E)E'(h) - 1 < 0,$$

because $E'(h) < 0$, as (A.6.4) shows. Therefore, $\Omega(h) = 0$ possesses a unique solution h^*. And since $E = E(h)$, $E'(h) < 0$, $\lambda = U'(h)$ and $U''(h) < 0$, it is obvious that both $E(h)$ and $U'(h)$ are monotonously decreasing function of h. It follows that $E^* = E(h^*)$ and $\lambda^* = U'(h^*)$ are also unique. Consequently, the steady state solution $(X^*, \lambda^*, h^*, E^*)$ is unique.

Appendix 6.2

By use of the result of the equation (6.55), we can define h as a function of λ:

$$h = (U')^{-1}(\lambda) = h(\lambda), \tag{A.6.5}$$

and it can be easily verified that

$$\frac{dh}{d\lambda} < 0. \tag{A.6.6}$$

Again, by applying the equation (6.56), we define E as a function of λ:

$$E = E(\lambda). \tag{A.6.7}$$

Since

$$\frac{d\lambda}{dE} = \frac{-C''(E)Y'(E) + C'(E)Y''(E)}{Y''(E)} > 0,$$

it yields

$$\frac{dE}{d\lambda} > 0. \tag{A.6.8}$$

After substituting $h(\lambda)$ for h and $E(\lambda)$ for E in equation (6.57), we observe now the following dynamic system of equations:

$$\begin{aligned} \dot{X} &= F(X) - Y(E(\lambda)) - h(\lambda) \\ \dot{\lambda} &= [r - F'(X)]\,\lambda. \end{aligned} \tag{A.6.9}$$

We differentiate then $\dot{X}$ and $\dot{\lambda}$ with X and λ, respectively. This yields

$$\begin{aligned} \frac{\partial \dot{X}}{\partial X} &= F'(X) \\ \frac{\partial \dot{X}}{\partial \lambda} &= -Y'(E)E'(\lambda) - h'(\lambda) \\ \frac{\partial \dot{\lambda}}{\partial X} &= -F''(X)\lambda \\ \frac{\partial \dot{\lambda}}{\partial \lambda} &= r - F'(X). \end{aligned} \tag{A.6.10}$$

A Taylor expansion of the dynamic system (A.6.9) at (X^*, λ^*) gives then

$$\begin{pmatrix} \dot{X} \\ \dot{\lambda} \end{pmatrix} = \begin{pmatrix} F'(X^*) & -Y'(E)E'(\lambda^*) - h'(\lambda^*) \\ -F''(X^*)\lambda^* & r - F'(X^*) \end{pmatrix} \begin{pmatrix} X - X^* \\ \lambda - \lambda^* \end{pmatrix} \tag{A.6.11}$$

, after the results of (A.6.10) are applied for substituting $\frac{\partial \dot{X}}{\partial X}$, $\frac{\partial \dot{X}}{\partial \lambda}$, $\frac{\partial \dot{\lambda}}{\partial X}$ and $\frac{\partial \dot{\lambda}}{\partial \lambda}$ in (A.6.11). Moreover, we know that in steady state $F'(X^*) = r$, as (A.6.2) shows. Hence, the two eigenvalues $\frac{r \pm \sqrt{r^2+4A}}{2}$ can be easily derived where $A = -F''(X^*)\lambda^* \left[-Y'(E)E'(\lambda^*) - h'(\lambda^*)\right] > 0$ under given assumptions. It is obvious that one of the eigenvalues is positive, and the other one is negative. Therefore, the steady state solution (X^*, λ^*) is a saddle point.

Chapter 7
Management capital, use of renewable resources, poaching and anti-poaching: a bioeconomic model with two state and two control variables

7.1 Introduction

The capital theory plays an extremely important role in investigating the management problem of the renewable resources. We can consider the link between capital theory and management of the renewable resources in three different aspects. First, it has long been recognized that the exploitation problem of the renewable resources can be analyzed in a capital-theoretic framework (Scott, 1955; Clark and Munro, 1975; Clark, 1976). This recognition is founded on the fact that renewable resource stock can be treated as a capital stock, in the sense that it can generate a consumption flow, i.e. harvest, over time, and current harvest will influence the resource stock level, the potential for regeneration and the future harvest possibilities of the resource stock (Clark and Munro, 1975).

Next, it is notable that the harvest rate is subject to the available capital stock, both physical and human, utilized in exploiting the resource stock in investigating the harvest problem of the renewable resources. Therefrom the problem of capital accumulation arises, especially in the fishery. The usual single-state-variable (resource stock) and single-control-variable (harvest) model was then extended to a model which consider explicitly two capital stocks (Smith, 1968: Smith, 1969). In a dynamic linear-in-control-variables model involving two state variables (resource stock, capital stock) and two control variables (harvest rate, investment rate), Clark, Clarke and Munro (1979) depicted the dynamic path of an optimal policy and showed, that the variable denoting the capital stock can be eliminated from the analysis and the model can be reduced to the usual single-state-variable model if capital investment is assumed to be perfectly reversible. In other words, the single-state-variable model is in fact a special case of the general two-state-variables model. Similar attempt was followed by Boyce (1995) in a nonlinear two-state variable, two-control variable model with irreversible investment. He found that the nonlinearity in control variables has great impact on the optimal harvest and investment policy.

While the problem of capital accumulation utilized in exploiting resource stock has been studied deliberately by economists for a long time, an important dimension of the link between capital theory and management of the renewable resources remains almost intact. That is the problem of capital accumulation utilized in protecting renewable resources when the problem of poaching and anti-poaching is concerned. Among the comparatively few articles in economic literature dealing with the relevant management problem of natural resources, Skonhoft and Solstad (1996) regarded the anti-poaching effort as a flow variable and hence reduced the problem to a single-state-variable model. Katz (2000) argued that the existence of social capital, a technical terminology applied in modern sociology, plays an important role in natural resource management. She is probably one of the few economists who explicitly use the terminology ´capital´ and treat a certain kind of capital as an critical factor in the management practices of natural resources. However, her analysis is in principle based on case studies and lacks an internally consistent theoretic foundation.

In this chapter, rather than investigating the problem of capital accumulation utilized in exploiting resource stock, we will focus on the problem of capital accumulation utilized in protecting renewable resources. To do this, a new state variable ´management capital´ will be introduced and the simple model of chapter six will be extended to a model with two state variables (resource stock, management capital) and two control variables (harvest rate, investment rate). The dynamic interaction between control variables and state variables will be investigated deliberately. By the assistance of a comparative static analysis, the influence of exogenous parameters on the equilibrium resource stock will be studied. The policy implications of the model for the conservation issues will also be discussed.

7.2 Management capital

Both the Skonhoft-Solstad model and our simple model in chapter six regard the management effort for protecting renewable resources as a flow variable. Undoubtedly, such treatment avoid to great extent the technical complexity. Nonetheless, it also neglects the fact in conservation practices that management authorities usually need certain kinds of capital assets to offer a management service flow through time which can contribute to the conservation of natural resources. Hence, in addition to the traditional stock variable ´resource stock´ in bioeconomic models, we may consider a new stock variable to help modeling the interaction between the two stock variables, i.e. the resource stock and the capital stock utilized in protecting resource stock. For convenience of discussion, we introduce here a new terminology ´management capital´, which encompasses all kinds of capital assets that are necessary for conservation practices. Under the concept of the management capital, three important components could be classified as follows.

First, among the various types of capital assets, the conventional physical capital is undoubtedly necessary for conservation practices. This usually includes buildings, vehicles, equipments and sometimes aircraft. The measurement of the physical capital stock, investment and depreciation is relatively easy. The physical inputs component of the flow variable ´management effort´ in the simple model can be then viewed as an investment in physical capital.

Secondly, no one can deny the fact that human capital plays a critical role in conservation practices. Similar to that Romer (1990) suggested, we define human capital as all forms of intangible knowledge, know-how and other human skills that are rivalrous and excludable, in the sense that they are inherently tied to the physical object, i.e. the human body. In contrast, certain forms of knowledge or know-how can be neither rivalrous nor excludable, since they are stored on paper or computer systems. They will be then separated from the rival component of knowledge (or know-how) and classified into an another sort of capital. Under this definition, the human capital stock may be measured in criteria such as the years of formal education, on-the-job training and experiences (Romer, 1990). And a proportion of the labor inputs component of the flow variable ´management effort´ in the simple model can be treated as an investment in human capital.[24] What remains in the labor inputs devoted to management can be viewed as an investment in institution capital which is investigated afterward.

Finally, it has long been recognized that a successful resource management requires an adequate institutional base. Under the term institution, we may identify the two essential components, namely property rights (Bromley, 1994; Lant, 1994; Swallow and Bromley, 1995) and organizational issues (Wade, 1987; Murphree, 1994; Bromley, 1994), which can substantially influence the outcomes of resource management. While these two factors are viewed as exogenous variables in most of the resource-management-relevant economic literature, we asserts here that institution in the field of resource management arises and develops mainly on account of the intentional actions

[24]It is notable that some routine and nonprofessional labor inputs make little contribution to the accumulation of human capital, or in other words, the depreciation rate of this component of the human capital is quite high.

taken by people who react to market and/or other non-market incentives, such as financial profits and/or non-use value. Hence, institution should be treated as an endogenous rather than exogenous variable in our model. Furthermore, developing new and modifying existing institution requires resource inputs and hence incurs costs. These resource inputs can be viewed as an investment in the formation of a special component of the management capital, i.e. the institution. In addition, for simplicity the knowledge or know-how which is neither rivalrous nor excludable will be included in this category of management capital, if we explain their essence in a wider sense that they are a necessary component of institution.[25] It should be here recognized that, up to now, it is extremely difficult to measure the stock of the ´institution capital´ quantitatively, while certain kind of qualitative ranking may be possible. However, we asserts that the institution factor, as a concept, is so important that it should not be neglected just because of the difficulty of measurement. Besides, the measurement of the investment in institution capital is relatively easy. It can be measured in criterion such as the labor inputs in relevant institution-building issues, though some ambiguity arises when we try to differentiate the labor inputs in relevant institution-building issues from the labor inputs in human capital investment. For example, that a park officer executes authority can contribute simultaneously to his personal experiences and the development of institution. But in practice, at the aggregate level of management capital, it is not necessary to differentiate them from each other.

In sum, we argue in this chapter that it is the existence of the stock variable management capital, rather than the flow variable management effort, that can have positive impacts on the management of renewable resources. And people invest intentionally in management capital in reaction to certain market and/or non-market incentives. It follows that management capital stock and relevant investment are treated as endogenous variables in our extended model.

7.3 The extended model

In this section, a nonlinear bioeconomic model with two state variables (resource stock, management capital) and two control variables (harvest rate, investment rate) is developed on the basis of the simple model in chapter six. The necessary conditions for the optimal policy are derived. The uniqueness

[25] For example, a data-bank for the distribution of renewable resources in a specific region is critical in developing property rights. An another example is that formally internal instructions can contribute to the organizational operation of a park authority.

and stability properties of the steady state solution of the model will also be presented in section 7.4 and 7.5, respectively.

In comparison to the simple model, two fundamental modifications are introduced to develop the extended model. First, rather than the flow variable management effort, it is the stock variable management capital that can affect the management of renewable resources. And people invest intentionally in management capital in reaction to certain market and/or non-market incentives. As a result, the endogenous variable $K(t)$ denoting the stock of management capital at time t is introduced.[26] The equation of motion for the management capital is

$$\dot{K}(t) \equiv \frac{dK(t)}{dt} = I(t) - \delta K(t) \tag{7.1}$$

where $I(t)$ represents the investment rate in management capital at time t and $0 < \delta < 1$ is the depreciation rate of management capital which is assumed to be constant.

The second modification refers to that, according to the analysis in section 6.2, a more complete poaching function $Y(X, K)$ is introduced here that the poaching Y depends on both the resource stock X and the management capital stock K. For simplicity, we express the poaching function as an additively separable function of the two stock variables:

$$Y(X, K) = W(X) - Y(K). \tag{7.2}$$

The function $W(X)$ represents the resource stock which is illegally exploited and possesses the properties

$$W'(X) > 0, \qquad W''(X) > 0. \tag{7.3}$$

The convexity assumption of $W(X)$ asserts that, with an increasingly marginal poaching rate, a higher resource stock will induce a higher poaching rate. The function $Y(K)$ denotes the resource stock which is potentially ´rescued´ from poaching activity because of the devotion of management capital. It is notable that $Y(K)$ could be greater than $W(X)$. If this happens, it means that resource owner takes active actions to increase resource stock, for example by re-introducing species individuals in protected areas. It is assumed that $Y(K)$ is concave so that

$$Y'(K) > 0, \qquad Y''(K) < 0. \tag{7.4}$$

[26] As in chapter six, the time notation will henceforth be omitted wherever possible.

This means that a higher level of management capital will reduce the poaching rate, but it is limited by a decreasingly marginal effect for per unit of additional management capital.

Now let us consider the same scenario as in section 6.2, that certain people or certain organizations are given the legal rights to exploit renewable resources in specific protected areas or in buffer areas around protected areas, and they are authorized to manage natural resources and human activities in those areas. All assumptions and properties with regard to the natural population dynamics of the resource and the utility function remain unchanged, as shown in section 6.2.

In addition to natural factors, the population dynamics of resource is also subject to human interference. When legal harvest and poaching factor are taken into account, the equation of motion for an exploited resource stock X can be stated as

$$\dot{X} \equiv \frac{dX}{dt} = F(X) - W(X) + Y(K) - h. \tag{7.5}$$

The resource management problem of the resource owner is to choose the harvest rate and the investment rate to maximize the net present utility (in monetary terms) derived from protecting and harvesting the resource, subject to the constraint of dynamics of the resource stock and management capital. This task can be formally expressed as

$$Max \int_0^\infty [U(h) + V(X) - C(I)]\, e^{-rt} dt \tag{7.6}$$

$$\begin{aligned} s.t.\ \dot{X} &= F(X) - W(X) + Y(K) - h \\ \dot{K} &= I - \delta K \end{aligned}$$

where $C(I)$ and r denote the investment cost function and the instantaneous discount rate, respectively. The investment cost function owns the properties that

$$C'(I) > 0, \qquad C''(I) > 0, \tag{7.7}$$

which implies that $C(I)$ is strictly increasing and convex in I.[27] The discount

[27]For resource owner, the marginal cost of the investment in physical capital remains probably constant, because a resource management authority constitutes only a tiny fraction of the whole physical capital market, for example the Jeep market. However, the marginal cost of the investment in human and institution capital is likely to be increasing, because the market of manpower for conservation is small and, especially at lower level, is typically localized.

rate is assumed to be constant and $0 < r < 1$. The optimization problem has hence two state variables, resource stock and management capital stock, and two control variables, harvest rate and investment rate, both of which are to be chosen optimally over time.

Next, the problem is analyzed by the application of the maximum principle. The current-value Hamiltonian of our case is

$$\begin{aligned} H &= U(h) + V(X) - C(I) + \lambda\left[F(X) - W(X) + Y(K) - h\right] \\ &\quad + \mu\left[I - \delta K\right] \end{aligned} \tag{7.8}$$

where λ is the current-value costate variable associated with the state variable X which gives the imputed marginal value of the resource stock, and μ is the current-value costate variable associated with the state variable K which gives the imputed marginal value of the management capital stock. Assuming an interior solution, the first order necessary conditions describing the optimization problem are given by equations (7.9)-(7.12) together with (7.1) and (7.5):

$$\frac{\partial H}{\partial h} = U'(h) - \lambda = 0 \tag{7.9}$$

$$\frac{\partial H}{\partial I} = -C'(I) + \mu = 0 \tag{7.10}$$

$$\dot{\lambda} = r\lambda - \frac{\partial H}{\partial X} = \left[r - F'(X) + W'(X)\right]\lambda - V'(X) \tag{7.11}$$

$$\dot{\mu} = r\mu - \frac{\partial H}{\partial K} = (r + \delta)\mu - \lambda Y'(K). \tag{7.12}$$

Under given assumptions, the Hamiltonian is apparently concave in the state variables X, K and in the control variables h, I. The second order conditions are therefore satisfied. Along the optimal trajectory, the equation (7.9) means that the imputed marginal value, or the shadow price of an extra resource stock, λ, must be equal to the marginal profit of harvesting one unit of renewable resources, $U'(h)$. The equation (7.10) implies that the shadow price of an extra management capital stock, μ, must equal the marginal cost of investment in management capital, $C'(I)$. Equation (7.11) indicates that the change rate of the shadow price of the resource stock, $\dot{\lambda}$, plus the marginal non-consumptive value of the resource stock, $V'(X)$, must equal the opportunity cost when resource owner goes on to keep one unit of resource stock, $[r - F'(X) + W'(X)]\,\lambda$. The opportunity cost includes the component of forgone ´interest´, $r\lambda$, and the loss derived from marginal poaching rate as a result of the increased resource stock, $W'(X)\lambda$, minus the gain derived from the marginal growth rate of the resource stock, $F'(X)\lambda$. Finally, the equation (7.12) implies that the change rate of the shadow price of the management capital stock, $\dot{\mu}$, plus the gain derived from the reduction of the poaching rate as a result of the devotion of an extra unit of management capital, $\lambda Y'(K)$, must be equated with the opportunity cost when resource owner devotes one unit of management capital stock to the protection of resources. The opportunity cost comprises the component of the ´interest cost´, $r\mu$, and the capital loss as a result of the depreciation, $\delta\mu$.

7.4 Uniqueness of the steady state solution

In the following sections, we will investigate the properties of the steady state solution of the dynamic problem. Suppose that a steady state exists, the steady state solution can be determined under the conditions that the resource stock, the management capital stock, the shadow price of the resource and the shadow price of the management capital are constant, i.e., $\dot{X} = \dot{K} = \dot{\lambda} = \dot{\mu} = 0$. First, the uniqueness of the steady state solution can be verified by proposition 5.

Proposition 5 *Under given assumptions with regard to the poaching function $W(X) - Y(K)$, the utility function $U(h) + V(X)$, the investment cost function $C(I)$ and the natural dynamics of the resource stock $F(X)$, the dynamic system (7.1), (7.5), (7.11) and (7.12) possess an unique steady state solution.*

Proof. In steady state, $\dot{X} = \dot{K} = \dot{\lambda} = \dot{\mu} = 0$. First, $\dot{\lambda} = 0$ gives

$$\lambda F'(X) - \lambda W'(X) + V'(X) = r\lambda \tag{7.13}$$

by applying equation (7.11). Under given assumptions, both $F'(X)$ and $V'(X)$ are monotonously decreasing and $W'(X)$ is monotonously increasing in X. Therefore, the function $\lambda F'(X) - \lambda W'(X) + V'(X)$ is also monotonously decreasing. It is obvious that one unique X^* exists which satisfying the condition $\lambda F'(X^*) - \lambda W'(X^*) + V'(X^*) = r\lambda$.

Next, after applying the results of equations (7.9) and (7.10) in (7.12), $\dot{\mu} = 0$ yields

$$(r + \delta)C'(I) - U'(h)Y'(K) = 0. \tag{7.14}$$

In addition, $\dot{K} = 0$ in steady state implies that

$$I = \delta K \tag{7.15}$$

by applying equation (7.1). After substituting δK for I in (7.14), h can be defined as an implicit function of K. A total differentiation of equation (7.14) with K yields then

$$\delta(r + \delta)C''(I) - U'(h)Y''(K) = U''(h)h'(K)Y'(K) \tag{7.16}$$

and it can be easily shown that

$$h'(K) < 0 \tag{7.17}$$

under given assumptions with regard to $C(I), U(h)$ and $Y(K)$.

Finally, by applying the results of the previous discussion about X^* and $h(K)$ in equation (7.5), $\dot{X} = 0$ gives

$$F(X^*) - W(X^*) + Y(K) - h(K) = 0.$$

Then we define a function

$$\Theta(K) = Y(K) - h(K) + A$$

where $A = F(X^*) - W(X^*)$ is a constant. It follows that

$$\Theta'(K) = Y'(K) - h'(K) > 0$$

by the application of (7.17) and of given assumption with regard to $Y(K)$. Hence, $\Theta(K)$ is monotonously increasing in K and $\Theta(K) = 0$ has a unique solution K^*. According to equations (7.15) and (7.17), this result ensures that the steady state solution of I and h is unique. It follows that $\lambda^* = U'(h^*)$ and $\mu^* = C'(I^*)$ are also unique. Consequently, the steady state solution $(X^*, K^*, h^*, I^*, \lambda^*, \mu^*)$ of the dynamic system is unique. ■

It is here worth noting the equilibrium resource stock. The special case of the simple model in chapter six indicated that the equilibrium resource stock level is always smaller than the stock level X_{MSY} which can afford the maximum sustainable yield, because of the condition $F'(X^*) = r$, as (6.59) showed. However, according to (7.13), the equilibrium resource stock in the extended model is determined by the condition $[r - F'(X^*) + W'(X^*)]\,\lambda - V'(X^*) = 0$. In comparison to (6.59), two new factors, i.e. $W'(X^*)\lambda$ and $V'(X^*)$ can also influence the steady state resource stock. The marginal poaching effect, $W'(X)\lambda$, leads to a decrease in the equilibrium resource stock, since greater resource stock induces more poaching and thereby increases the cost of holding on resource stock. The marginal non-consumptive-value effect, $V'(X)$, results in an increase in the equilibrium resource stock, because greater resource stock raises the non-consumptive value, and thereby increases the benefit derived from holding on resource stock as an asset. The interaction between these two effects makes it difficult to determine whether the equilibrium resource stock is greater or smaller than X_{MSY}. Finally, it depends on the relative strength of these two effects. In any case, it is in the extended model possible that the equilibrium resource stock is greater than X_{MSY}, if renewable resources can generate sufficiently great marginal non-consumptive value so that the marginal non-consumptive-value effect dominates the steady state solution.

7.5 Stability of the steady state solution

Now, we concentrate on the stability of the steady state solution. By applying the result $\lambda = U'(h)$ of the equation (7.19), h can be defined as a function of λ:

$$h = (U')^{-1}(\lambda) = h(\lambda). \tag{7.18}$$

it can be easily verified that

$$\frac{dh}{d\lambda} < 0. \tag{7.19}$$

Again, by applying the result $\mu = C'(I)$ of the equation (7.10), I can be defined as a function of μ:

$$I = (C')^{-1}(\mu) = I(\mu). \tag{7.20}$$

it also can be shown that

$$\frac{dI}{d\mu} > 0. \tag{7.21}$$

After substituting $h(\lambda)$ for h in (7.5) and $I(\mu)$ for I in (7.1), we observe now the following dynamic system of equations:

$$\begin{aligned}
\dot{X} &= F(X) - W(X) + Y(K) - h(\lambda) \\
\dot{K} &= I(\mu) - \delta K \\
\dot{\lambda} &= [r - F'(X) + W'(X)]\,\lambda - V'(X) \\
\dot{\mu} &= (r+\delta)\mu - \lambda Y'(K).
\end{aligned} \tag{7.22}$$

The Jacobian of the dynamic system (7.22) is stated as

$$J = \begin{bmatrix}
\frac{\partial \dot{X}}{\partial X} & \frac{\partial \dot{X}}{\partial K} & \frac{\partial \dot{X}}{\partial \lambda} & \frac{\partial \dot{X}}{\partial \mu} \\
\frac{\partial \dot{K}}{\partial X} & \frac{\partial \dot{K}}{\partial K} & \frac{\partial \dot{K}}{\partial \lambda} & \frac{\partial \dot{K}}{\partial \mu} \\
\frac{\partial \dot{\lambda}}{\partial X} & \frac{\partial \dot{\lambda}}{\partial K} & \frac{\partial \dot{\lambda}}{\partial \lambda} & \frac{\partial \dot{\lambda}}{\partial \mu} \\
\frac{\partial \dot{\mu}}{\partial X} & \frac{\partial \dot{\mu}}{\partial K} & \frac{\partial \dot{\mu}}{\partial \lambda} & \frac{\partial \dot{\mu}}{\partial \mu}
\end{bmatrix} \tag{7.23}$$

and after some routine calculations it yields

$$J = \begin{bmatrix}
F' - W' & Y' & -h' & 0 \\
0 & -\delta & 0 & I' \\
-\lambda(F'' - W'') - V'' & 0 & r - F' + W' & 0 \\
0 & -\lambda Y'' & -Y' & r+\delta
\end{bmatrix}. \tag{7.24}$$

The value of the determinant can be derived:

$$\begin{aligned}
|J| = {} & \delta(r+\delta)\,[\lambda(F'' - W'') + V'']\,h' \\
& -\delta(r+\delta)(F' - W')(r - F' + W') \\
& -\lambda\,[\lambda(F'' - W'') + V'']\,Y''h'I' \\
& +\lambda(F' - W')(r - F' + W')Y''I' \\
& -[\lambda(F'' - W'') + V'']\,(Y')^2 I'.
\end{aligned} \tag{7.25}$$

Next, we define a function G as

$$G = \begin{vmatrix} \frac{\partial \dot{X}}{\partial X} & \frac{\partial \dot{X}}{\partial \lambda} \\ \frac{\partial \dot{\lambda}}{\partial X} & \frac{\partial \dot{\lambda}}{\partial \lambda} \end{vmatrix} + \begin{vmatrix} \frac{\partial \dot{K}}{\partial K} & \frac{\partial \dot{K}}{\partial \mu} \\ \frac{\partial \dot{\mu}}{\partial K} & \frac{\partial \dot{\mu}}{\partial \mu} \end{vmatrix} + 2 \begin{vmatrix} \frac{\partial \dot{X}}{\partial K} & \frac{\partial \dot{X}}{\partial \mu} \\ \frac{\partial \dot{\lambda}}{\partial K} & \frac{\partial \dot{\lambda}}{\partial \mu} \end{vmatrix}$$

The value of G can be derived as follows:

$$\begin{aligned}
G = {} & (F' - W')(r - F' + W') - [\lambda(F'' - W'') + V'']\,h' \\
& -\delta(r+\delta) + \lambda Y''I'.
\end{aligned} \tag{7.26}$$

The local stability property of the dynamic system (7.22) is determined by the eigenvalues of the Jacobian J. According to the general formula developed by Dockner (1985), we can calculate the eigenvalues of the Jacobian associated with the optimal control problems with two state variables:

$$\xi_{1,2,3,4} = \frac{r}{2} \pm \left\{ \left(\frac{r}{2}\right)^2 - \frac{G}{2} \pm \left[\left(\frac{G}{2}\right)^2 - |J| \right]^{\frac{1}{2}} \right\}^{\frac{1}{2}} \tag{7.27}$$

where r is the discount rate and $|J|$ is the determinant of J. And Dockner (1985) showed that two of the eigenvalues are positive and the other two negative, i.e. the steady state solution is saddle point stable, if $|J| > 0$ and $G < 0$. By applying this result, the local stability property of the dynamic system (7.22) can verified under further assumptions.

Proposition 6 *Under given assumptions with regard to the poaching function $W(X) - Y(K)$, the utility function $U(h) + V(X)$, the investment cost function $C(I)$ and the natural dynamics of the resource stock $F(X)$, the unique steady state solution of the dynamic system (7.22) is saddle point stable, if $U'(F'' - W'') + V'' < U''(F' - W')(r - F' + W')$.*

Proof. Since $h = (U')^{-1}(\lambda) = h(\lambda)$ and $\lambda = U'(h)$ according to (7.18) and (7.9), and since $\frac{dh}{d\lambda}\frac{d\lambda}{dh} = 1$, it can be easily verified that $\frac{dh}{d\lambda} = \frac{1}{U''(h)}$ because $\frac{d\lambda}{dh} = U''(h)$. The condition $U'(F'' - W'') + V'' < U''(F' - W')(r - F' + W')$ can be then written as $\frac{U'(F''-W'')+V''}{U''} > (F'-W')(r-F'+W')$ or equivalently as $[\lambda(F'' - W'') + V'']\, h' > (F' - W')(r - F' + W')$.

Let us first determine the sign of $|J|$. If $[\lambda(F'' - W'') + V'']\, h' > (F' - W')(r - F' + W')$, it can be easily shown that the sum of the first two terms on the right-hand side of (7.25)

$$\delta(r + \delta)\, [\lambda(F'' - W'') + V'']\, h' - \delta(r + \delta)(F' - W')(r - F' + W') > 0$$

is positive because of $\delta(r + \delta) > 0$. And the sum of the third and fourth terms

$$-\lambda\, [\lambda(F'' - W'') + V'']\, Y''h'I' + \lambda(F' - W')(r - F' + W')Y''I'$$

is also positive since $Y'' < 0$ and $I' > 0$, according to (7.21). Finally, the fifth term $-\, [\lambda(F'' - W'') + V'']\, (Y')^2 I'$ cannot be negative since $F'' < 0$, $W'' > 0$ and $V'' < 0$. Accordingly, we can show that $|J| > 0$.

Next, we investigate the sign of G. Under the same premise, it is obvious that the sum of the first two terms on the right-hand side of (7.26) is negative. The third term $-\delta(r+\delta)$ is clearly negative. And the fourth term $\lambda Y''I'$ is also negative under given assumptions. Hence, $G < 0$. The results $|J| > 0$ and $G < 0$ guarantee that two of the eigenvalues of J are positive and the other two negative (Dockner, 1985). The steady state solution of the dynamic system (7.22) is therefore saddle point stable. ■

The saddle-point stability property of the steady state solution implies that, given the initial value of the resource and management capital stock which are close to the steady state, it will always be possible for the resource owner to choose a pair of optimal initial values of the harvest rate and the investment rate which are on the stable trajectories that converge to the steady state equilibrium of the dynamic system.

7.6 Comparative static analysis

We conduct here a comparative static analysis to investigate the influence of permanent changes in the exogenous parameters on the equilibrium resource stock. To study the impact of as many parameters as possible, a special version of the extended model will be considered by introducing particular specifications for $F(X)$, $W(X)$, $V(X)$ and $Y(K)$. In what follows, we specify

$$\begin{aligned} F(X) &= \rho X(1-X) \\ W(X) &= \alpha X \\ V(X) &= \beta X \\ Y(K) &= \gamma K. \end{aligned} \tag{7.28}$$

The exogenous coefficient ρ denotes the intrinsic growth rate of the resource stock and, as discussed in section 6.6. The constant coefficient α represents the marginal poaching rate and is an indicator for measuring the intensity of poaching activity. The constant coefficient β denotes the marginal non-consumptive value of resource stock. Finally, the constant γ is a efficiency coefficient which measures the effect of one unit extra management capital on the poaching rate. The linearity assumption with regard to $W(X)$, $V(X)$ and $Y(K)$ is somewhat unrealistic, but it allows us to study the effects of more exogenous parameters than before. It can be easily verified that the fundamental concavity property of the Hamiltonian of the optimization problem and the uniqueness together with the stability of the steady state solution remain unchanged under these specifications.

The specific version of the extended model results in the following dynamic system after some rearrangement:

$$
\begin{aligned}
\dot{X} &= \rho X(1-X) - \alpha X + \gamma K - h(\lambda) \\
\dot{K} &= I(\mu) - \delta K \\
\dot{\lambda} &= (r - \rho + 2\rho X + \alpha)\,\lambda - \beta \\
\dot{\mu} &= (r+\delta)\mu - \lambda\gamma.
\end{aligned}
\tag{7.29}
$$

The system encompasses four endogenous variables, X, K, λ, μ, and six exogenous variables, r, α, β, γ, δ, ρ. By taking the total differential of the system (7.29), it yields

$$
\begin{bmatrix}
\rho - 2\rho X - \alpha & \gamma & -h' & 0 \\
0 & -\delta & 0 & I' \\
2\rho\lambda & 0 & r - \rho + 2\rho X + \alpha & 0 \\
0 & 0 & -\gamma & r+\delta
\end{bmatrix}
\cdot
\begin{bmatrix}
dX^* \\ dK^* \\ d\lambda^* \\ d\mu^*
\end{bmatrix}
\tag{7.30}
$$

$$
=
\begin{bmatrix}
-X + X^2 & 0 & X & 0 & -K & 0 \\
0 & 0 & 0 & 0 & 0 & K \\
\lambda - 2X\lambda & -\lambda & -\lambda & 1 & 0 & 0 \\
0 & -\mu & 0 & 0 & \lambda & -\mu
\end{bmatrix}
\cdot
\begin{bmatrix}
d\rho \\ dr \\ d\alpha \\ d\beta \\ d\gamma \\ d\delta
\end{bmatrix}
$$

By application of Cramer´s rule, the following results of the comparative static analysis can be derived.

Proposition 7 *Under special specifications of (7.29) with regard to the poaching function, the utility function and the natural dynamics of the resource stock, and under given assumptions with regard to the investment cost function, $\frac{dX^*}{d\rho} > 0$ if the equilibrium resource stock is less than or equals to the maximum sustainable yield stock level, and $\frac{dX^*}{d\rho}$ is ambiguous if the equilibrium resource stock is higher than the maximum sustainable yield stock level.*

Proof. See appendix 7.1. ■

The outcome of proposition 7 shows that an increase in the intrinsic growth rate of the resources ρ will raise the equilibrium resource stock, if the equilibrium resource stock is smaller than or equals to the maximum sustainable yield stock level. The underlying reason for this result is clear. As in section 6.6 discussed, the maximum sustainable yield stock level equals $\frac{1}{2}$ under the special specification. If $X^* < \frac{1}{2}$, an increase in ρ will raise the overall level of the marginal growth rate of the resources $F'(X^*) = \rho - 2\rho X^*$, and thereby raise the return of keeping resource stock. Hence, the resource owner will be willing to hold a higher resource stock. On the other hand, if $X^* > \frac{1}{2}$, an increase in ρ will lower the overall level of the marginal growth rate of the resources, and cause a disincentive for keeping resource stock. However, the first term in the right-hand side of the first equation of (A.7.5) $\delta(r+\delta)(X-X^2)\,(r-\rho+2\rho X+\alpha)$ is positive. This leads to an ambiguously total effect.[28]

Proposition 8 *Under special specifications of (7.29) with regard to the poaching function, the utility function and the natural dynamics of the resource stock, and under given assumptions with regard to the investment cost function, the following comparative static effects are derived:* $\frac{dX^*}{dr} < 0$, $\frac{dX^*}{d\alpha} < 0$, $\frac{dX^*}{d\beta} > 0$, $\frac{dX^*}{d\gamma} > 0$ *and* $\frac{dX^*}{d\delta} < 0$.

Proof. See appendix 7.1. ■

Proposition 8 demonstrates some unambiguous comparative static effects. First, an increase in the discount rate r and in the marginal poaching rate α will lower the equilibrium resource stock, and vice versa. This is because a higher discount rate raises the opportunity cost of holding resource stock, as the term $r\lambda$ of (7.11) shows, and constitutes motives for stock disinvestment. The same rationale applies also to the effect of a permanent change in the marginal poaching rate. On the contrary, an increase in the marginal non-consumptive value of resource stock β raises the benefit of holding resource stock, as (7.11) demonstrates, and encourages the resource owner to hold a higher stock level. The coefficient γ measures the efficiency of management capital against the poaching activity, and a higher γ implies that management capital can dampen poaching more effectively than before and thereby

[28]Up to now, empirical studies generally support the conclusion that species with low intrinsic growth rate are less resilient to harvest (e.g., Bodmer, 1995a; Bodmer, 1995b; Bodmer et al., 1997a; Bodmer et al., 1997b; Bodmer and Puertas, 2000; Fa et al., 1995; Fitzgibbon et al., 1995; Clayton and Milner-Gulland, 2000; Lee, 2000; Peres, 2000). This may support the conjecture that the term $\delta(r+\delta)(X-X^2)\,(r-\rho+2\rho X+\alpha)$ in the right-hand side of the first equation of (A.7.5) is big enough so that, in any case, we can obtain an unambiguous total effect $\frac{dX^*}{d\rho} > 0$.

contribute to a greater resource stock. Finally, an increased depreciation rate of management capital δ will reduce the equilibrium resource stock. The intuition is that, other things being equal, a higher depreciation rate causes more depreciation of the given management capital stock, and this in turn weakens the strength of anti-poaching action and results in a smaller equilibrium resource stock. With reference to the comparative static effects of exogenous variables on the equilibrium management capital stock, appendix 7.1 shows that it is not possible to determine the signs of these effects as a result of the ambiguous term $\rho - 2\rho X - \alpha$ in every expressions of (A.7.6).

7.7 Concluding remarks and policy implications

In this chapter we address the important role played by management capital in conservation praxis. We argue that it is the existence of the stock variable management capital, rather than the flow variable management effort, that can have positive impacts on the management of renewable resources. People invest intentionally in management capital in reaction to certain market and/or non-market incentives. Accordingly, in comparison to the traditional one-state-variable bioeconomic models in which no man-made capital exists, and the two-state-variable bioeconomic models which investigate the problem of capital accumulation utilized in exploiting resource stock, we introduce a new stock variable to help modeling the interaction between the two stock variables, i.e. the resource stock and the capital stock utilized in protecting resource stock. A nonlinear bioeconomic model with two state variables (resource stock, management capital) and two control variables (harvest rate, investment rate) is then developed on the basis of the simple model in chapter six, and under the premise that people are allowed to use legally the renewable resources in or around protected areas. In addition to the management capital, the important modification with regard to the functional forms of the poaching function is also made. Under some assumptions with reference to the poaching function, the utility function, the investment cost function and the natural dynamics of the resource stock, it can be verified that the steady state solution of the dynamic problem is unique and saddle point stable. By introducing a specific version of the extended model, we can identify six critical exogenous parameters, the discount rate, the intrinsic growth rate of the resource stock, the marginal poaching rate, the marginal non-consumptive value of the resource stock, the efficiency coefficient of the management capital and the depreciation rate of the management capital, which will influence the equilibrium resource stock level. Some critical comparative static effects are thereby found.

Several important outcomes of the extended model and the relevant implications for conservation policy are here worth noting. First, the special case of the simple model in chapter six concludes that the equilibrium resource stock level is in any case smaller than the maximum sustainable yield stock level. However, this conclusion can not be applied in the extended model (also not in the simple model). In fact, it is difficult to know whether the equilibrium resource stock is greater or smaller than X_{MSY} in the extended model as a result of the introduction of the marginal poaching effect and the marginal non-consumptive-value effect, as discussed in section 7.4. In any case, it can happen in the extended model that the equilibrium resource stock is greater than X_{MSY}, if renewable resources can generate sufficiently great marginal non-consumptive value so that the marginal non-consumptive-value effect dominates the steady state solution. For some cases in which the species have a maximum sustainable yield stock level close to the carrying capacity, this implies that the equilibrium resource stock level may be quite high, if the non-consumptive value of the species are highly appreciated, and/or if some other conditions discussed later are appropriate. Certainly, under some inappropriate conditions, the possibility can not be excluded that the equilibrium resource stock is small enough in some cases that people will be seriously concerned about the viability of the resource and the loss of its ecological functions. What these conditions are will be investigated later.

By the application of the comparative static analysis in section 7.6, some important parameters affecting the equilibrium resource stock and sustainability are identified. Of the six parameters, the intrinsic growth rate of species is a well-known biological factor. The outcome of the comparative static analysis shows that an increase in the intrinsic growth rate will raise the equilibrium resource stock, if the equilibrium resource stock is smaller than or equals to the maximum sustainable yield stock level. In the cases which the equilibrium resource stock is greater than the maximum sustainable yield stock level, the comparative static effect is ambiguous. Nonetheless, empirical studies generally support the conclusion that species with low intrinsic growth rate are less resilient to harvest. Accordingly, we may generally conclude that an increase in the intrinsic growth rate will raise the equilibrium resource stock, and vice versa.

The comparative static analysis addresses also the comparative static effects of the other parameters on the equilibrium resource stock. In sum, the lower the discount rate, the marginal poaching rate and the depreciation rate of management capital, and the higher the marginal non-consumptive value and the efficiency coefficient for the management capital is, the higher the equilibrium resource stock will be. Accordingly, we can use these parameters as indicators for evaluating the success probability of a sustainable use project before or when it is practiced. The sustainable use strategy may potentially be more appropriate in sites with more positive indicators, namely high marginal non-consumptive value, intrinsic growth rate and efficiency coefficient for the management capital, and low discount rate, marginal poaching rate and depreciation rate of management capital, than those sites with less positive indicators.

Appendix 7.1

By keeping all exogenous variables constant except a certain one, the equation (7.30) can be rewritten as

$$JM_i = d_i,\ i = \rho, r, \alpha, \beta, \gamma, \delta \tag{A.7.1}$$

where

$$J = \begin{bmatrix} \rho - 2\rho X - \alpha & \gamma & -h' & 0 \\ 0 & -\delta & 0 & I' \\ 2\rho\lambda & 0 & r - \rho + 2\rho X + \alpha & 0 \\ 0 & 0 & -\gamma & r+\delta \end{bmatrix}, \tag{A.7.2}$$

$$\begin{aligned} M_\rho &= (\frac{dX^*}{d\rho}, \frac{dK^*}{d\rho}, \frac{d\lambda^*}{d\rho}, \frac{d\mu^*}{d\rho})' \\ M_r &= (\frac{dX^*}{dr}, \frac{dK^*}{dr}, \frac{d\lambda^*}{dr}, \frac{d\mu^*}{dr})' \\ M_\alpha &= (\frac{dX^*}{d\alpha}, \frac{dK^*}{d\alpha}, \frac{d\lambda^*}{d\alpha}, \frac{d\mu^*}{d\alpha})' \\ M_\beta &= (\frac{dX^*}{d\beta}, \frac{dK^*}{d\beta}, \frac{d\lambda^*}{d\beta}, \frac{d\mu^*}{d\beta})' \\ M_\gamma &= (\frac{dX^*}{d\gamma}, \frac{dK^*}{d\gamma}, \frac{d\lambda^*}{d\gamma}, \frac{d\mu^*}{d\gamma})' \\ M_\delta &= (\frac{dX^*}{d\delta}, \frac{dK^*}{d\delta}, \frac{d\lambda^*}{d\delta}, \frac{d\mu^*}{d\delta})' \end{aligned} \tag{A.7.3}$$

and

$$\begin{aligned} d_\rho &= (-X + X^2, 0, \lambda - 2X\lambda, 0)' \\ d_r &= (0, 0, -\lambda, -\mu)' \\ d_\alpha &= (X, 0, -\lambda, 0)' \\ d_\beta &= (0, 0, 1, 0)' \\ d_\gamma &= (-K, 0, 0, \lambda)' \\ d_\delta &= (0, K, 0, -\mu)'. \end{aligned} \tag{A.7.4}$$

The steady state value of X, K, λ and μ are denoted by asterisks. By application of Cramer´s rule, the comparative statics can be derived:

$$M_{ij} = \frac{|J_{ij}|}{|J|}$$

where M_{ij} is the jth element of vector M_i, and J_{ij} is the matrix J with its column j substituted by the vector i. As a result of the assumption of proposition 6, it is clear that $|J| > 0$. Therefore, the signs of M_{ij} are determined by the signs of $|J_{ij}|$. Some routine calculations yield that

$$\begin{aligned}
|J_{\rho 1}| &= \delta(r+\delta)(X-X^2)(r-\rho+2\rho X+\alpha) \\
&\quad -\delta(r+\delta)h'(\lambda-2X\lambda)+I'\gamma^2(\lambda-2X\lambda) \\
|J_{r1}| &= \delta(r+\delta)\lambda h' - I'\mu\gamma(r-\rho+2\rho X+\alpha) - I'\gamma^2\lambda \\
|J_{\alpha 1}| &= -\delta(r+\delta)X(r-\rho+2\rho X+\alpha)+\delta(r+\delta)\lambda h' - I'\lambda\gamma^2 \\
|J_{\beta 1}| &= -\delta(r+\delta)h' + I'\gamma^2 \\
|J_{\gamma 1}| &= \delta(r+\delta)K(r-\rho+2\rho X+\alpha)+I'\lambda\gamma(r-\rho+2\rho X+\alpha) \\
|J_{\delta 1}| &= (r-\rho+2\rho X+\alpha)\left[-I'\gamma\mu-(r+\delta)\gamma K\right].
\end{aligned} \tag{A.7.5}$$

From (7.29), we know that $r-\rho+2\rho X+\alpha = \frac{\beta}{\lambda} > 0$ in steady state. And (7.19) and (7.21) show that $h' < 0$ and $I' > 0$, respectively. By applying these results, it can be easily shown that $|J_{\rho 1}| > 0$, if $X < \frac{1}{2}$ or $X = \frac{1}{2}$, and the sign of $|J_{\rho 1}|$ is ambiguous , if $X > \frac{1}{2}$. This applies also to $\frac{dX^*}{d\rho}$, since $|J| > 0$. Similarly, it is obvious that $|J_{r1}| < 0$, $|J_{\alpha 1}| < 0$, $|J_{\beta 1}| > 0$, $|J_{\gamma 1}| > 0$ and $|J_{\delta 1}| < 0$. It follows that $\frac{dX^*}{dr} < 0$, $\frac{dX^*}{d\alpha} < 0$, $\frac{dX^*}{d\beta} > 0$, $\frac{dX^*}{d\gamma} > 0$ and $\frac{dX^*}{d\delta} < 0$.

In the same way, the following determinants can be expressed as

$$\begin{aligned}
|J_{\rho 2}| &= \gamma I'\left[2\rho\lambda\left(X^2-X\right)-(\lambda-2X\lambda)(\rho-2\rho X-\alpha)\right] \\
|J_{r2}| &= I'\lambda\gamma(\rho-2\rho X-\alpha)+2\rho\lambda I'\mu h' \\
&\quad +I'\mu(\rho-2\rho X-\alpha)(r-\rho+2\rho X+\alpha) \\
|J_{\alpha 2}| &= I'\gamma\left[\lambda(\rho-2\rho X-\alpha)+2\rho\lambda X\right] \\
|J_{\beta 2}| &= I'\gamma(\rho-2\rho X-\alpha) \\
|J_{\gamma 2}| &= -2\rho\lambda^2 I'h' - 2\rho\lambda\gamma K I' \\
&\quad -\lambda I'(\rho-2\rho X-\alpha)(r-\rho+2\rho X+\alpha) \\
|J_{\delta 2}| &= \left[K(r+\delta)+I'\mu\right]\left[(\rho-2\rho X-\alpha)(r-\rho+2\rho X+\alpha)\right] \\
&\quad +2\rho\lambda K(r+\delta)h' + 2\rho\lambda\mu I'h'.
\end{aligned} \tag{A.7.6}$$

The term $\rho-2\rho X-\alpha$ exists in all of the above expressions. As a result of the

fact that it is not possible to determine the sign of the term $\rho-2\rho X-\alpha$, the comparative static effects $\frac{dK^*}{d\rho}$, $\frac{dK^*}{dr}$, $\frac{dK^*}{d\alpha}$, $\frac{dK^*}{d\beta}$, $\frac{dK^*}{d\gamma}$ and $\frac{dK^*}{d\delta}$ are ambiguous, if no further assumption is applied.

Chapter 8
Management capital, use of renewable resources, poaching and anti-poaching: a general bioeconomic model

In this chapter we will further investigate the interaction between use of renewable resources, management capital accumulation, resource stock and poaching activities in a more general model. To do this, the extended model of chapter 7 will be generalized in the sense that, instead of applying the additively separable poaching function $W(X) - Y(K)$ and utility function $U(h) + V(X)$ for sake of technical simplicity, a general poaching function $W(X, K)$ and utility function $U(h, X)$ will be introduced in this chapter. The necessary conditions for the optimal policy will be derived. The existence property of the steady state solution of the model will also be presented. By application of computer simulation, the relevant phase diagrams and the impacts of exogenous parameters on the equilibrium resource stock will be studied. The implications of the model for conservation policy will be addressed in section 8.5 and 8.6.

8.1 The general model

In comparison to the extended model, two fundamental modifications are here introduced to develop the general model. First, a more general poaching function $W(X, K)$, rather than the additively separable poaching function in the extended model, is applied here that the poaching rate depends on both resource stock and management capital stock. The function $W(X, K)$ represents the resource stock which is illegally exploited and possesses the following properties

$$W_X > 0,\ W_{XX} > 0,\ W_K < 0,\ W_{KK} > 0, W_{XK} < 0 \tag{8.1}$$

$$\text{and} \quad W_{XX}W_{KK} - W_{XK}^2 > 0.$$

The convexity assumption of $W(X, K)$ in X implies that, with an increasingly marginal poaching rate, a higher resource stock will induce a higher poaching rate. The convexity assumption of $W(X, K)$ in K means that a higher level of management capital will reduce the poaching rate, but it is limited by a decreasingly marginal effect for per unit of additional management capital.

The second modification involves the functional form of the utility function, namely, the additively separable utility function in the extended model is here replaced by the general utility function $U(h, X)$,which simultaneously considers the consumptive value and non-consumptive value of renewable resources. We assume that $U(h, X)$ can be measured in monetary terms and is strictly increasing and concave in both X and h

$$U_X > 0,\ U_{XX} < 0,\ U_h > 0,\ U_{hh} < 0,\ U_{Xh} > 0 \tag{8.2}$$

$$\text{and } U_{hh}U_{XX} - W_{Xh}^2 > 0$$

where U_X represents the marginal non-consumptive utility generated by an additional unit of resource stock, and U_h denotes the marginal gross harvest profit from an additional unit of harvest which in turn equals the differential between the unit resource price and the unit harvest cost. In addition, it is assumed that

$$\lim_{X \to \bar{X}} U_X(h, X) = 0 \tag{8.3}$$

, meaning that the marginal non-consumptive utility of resource stock equals zero when the resource stock approaches the carrying capacity.

Following the previous modifications, the equation of motion for the exploited resource stock X must also be modified, if legal harvest and poaching activities are taken into account:

$$\dot{X} \equiv \frac{dX}{dt} = F(X) - W(X, K) - h. \tag{8.4}$$

The functional forms of the other functions, including the equation of motion for management capital,

$$\dot{K} = I - \delta K \tag{8.5}$$

and the investment cost function, $C(I)$, and the relevant assumptions remain unchanged. The meanings of the notations δ and r and their properties remain also unchanged.

Now let us consider the same scenario as considered in section 7.3, that certain people or organizations are given the legal rights to exploit renewable resources in specific protected areas or in buffer areas around protected areas, and they are authorized to manage natural resources and human activities in those areas. The resource management problem of the resource owner is to decide the optimal harvest rate and the investment rate to maximize the net present utility (in monetary terms) derived from protecting and harvesting the resource, subject to the constraint of dynamics of the resource stock and management capital. This problem can be formally stated as

$$Max \int_0^\infty [U(h,X) - C(I)]\, e^{-rt} dt \tag{8.6}$$

$$\begin{aligned} s.t.\ \dot{X} &= F(X) - W(X,K) - h \\ \dot{K} &= I - \delta K. \end{aligned}$$

The optimization problem has therefore two state variables, resource stock and management capital stock, and two control variables, harvest rate and investment rate, both of which are to be chosen optimally over time. Next, the problem can be analyzed by the application of the maximum principle. The corresponding current-value Hamiltonian is

$$H = U(h,X) - C(I) + \lambda\,[F(X) - W(X,K) - h] + \mu\,[I - \delta K] \tag{8.7}$$

where λ is the current-value costate variable associated with the state variable X which gives the imputed marginal value of the resource stock, and μ is the current-value costate variable associated with the state variable K which gives the imputed marginal value of the management capital stock. Assuming an interior solution, the first order necessary conditions describing the optimization problem are given by equations (8.8)-(8.11) together with (8.4) and (8.5):

$$\frac{\partial H}{\partial h} = U_h - \lambda = 0 \tag{8.8}$$

$$\frac{\partial H}{\partial I} = -C_I + \mu = 0 \tag{8.9}$$

$$\dot{\lambda} = (r - F_X + W_X)\lambda - U_X \tag{8.10}$$

$$\dot{\mu} = (r + \delta)\mu + W_K \lambda. \tag{8.11}$$

Under given assumptions, the Hamiltonian is apparently concave in the state variables X, K and in the control variables h, I. The second order conditions are therefore satisfied. Along the optimal trajectory, equation (8.8) shows that the imputed marginal value, or the shadow price of an extra resource stock, λ, must be equal to the marginal gross profit of harvesting one unit of renewable resources, U_h. The equation (8.9) means that the shadow price of an extra management capital stock, μ, must equal the marginal cost of investment in management capital, C_I. Equation (8.10) implies that the change rate of the shadow price of the resource stock, $\dot{\lambda}$, plus the marginal non-consumptive value of the resource stock, U_X, must be equal to the opportunity cost when resource owner goes on to keep one unit of resource stock, $(r - F_X + W_X)\lambda$. The opportunity cost includes the component of forgone ´interest´, $r\lambda$, and the loss derived from marginal poaching rate as a result of the increased resource stock, $W_X\lambda$, minus the gain derived from the marginal growth rate of the resource stock, $F_X\lambda$. Finally, the equation (8.11) indicates that the change rate of the shadow price of the management capital stock, $\dot{\mu}$, plus the gain derived from the reduction of the poaching rate as a result of the devotion of an extra unit of management capital, $-W_K\lambda$, must be equated with the opportunity cost when resource owner devotes one unit of management capital stock to the protection of resources. The opportunity cost comprises the component of the ´interest cost´, $r\mu$, and the capital loss as a result of the depreciation, $\delta\mu$.

8.2 Existence of the steady state solution

Now let us verify the existence property of the steady state solution by virtue of proposition 9.[29]

Proposition 9 *Under given assumptions with regard to the poaching function $W(X, K)$, the utility function $U(h, X)$, the investment cost function $C(I)$ and the natural dynamics of the resource stock $F(X)$, the dynamic system (8.4), (8.5), (8.10) and (8.11) possess at least one steady state solution.*

Proof. In steady state, $\dot{X} = \dot{K} = \dot{\lambda} = \dot{\mu} = 0$. Consider both h and K as

[29] The proof of the proposition 9 is inspired by the discussion on existence property of the steady state equilibrium of the two-state-and-two-control-variables model developed by Li and Lőfgren (1998).

an implicit function of X. By dividing $\dot{\lambda}$ by λ, we can define a function

$$\Gamma(X) = \frac{\dot{\lambda}}{\lambda} = (r - F_X + W_X(X, K(X))) - \frac{U_X(h(X), X)}{U_h(h(X), X)}. \tag{8.12}$$

Furthermore, let $\underline{X}$ be the resource stock level which satisfies the condition $F_X(\underline{X}) - W_X(\underline{X}, K(\underline{X})) = r$. When $\dot{\lambda} = 0$, it also implies that $\Gamma(X) = 0$, and the following result can be derived from (8.12):

$$F_X(X^*) - W_X(X^*, K(X^*)) = r - \frac{U_X(h(X^*), X^*)}{U_h(h(X^*), X^*)} < r$$

where X^* denotes the equilibrium resource stock level. Since F_X is monotonically decreasing and W_X monotonically increasing in X, it can be easily verified that $X^* > \underline{X}$.

Finally, at $X = \underline{X}$, we know $\Gamma(X) = -\frac{U_X(h(X), X)}{U_h(h(X), X)} < 0$. When X approaches the carrying capacity $\bar{X}$, it can be shown that $\Gamma(X) = (r - F_X + W_X(X, K(X))) > 0$, since it is assumed in (8.3) that $\lim_{X \to \bar{X}} U_X(h, X) = 0$. Under the assumption that $\Gamma(X)$ is a continuous function of X, the results that $\Gamma(X) < 0$ at $\underline{X}$ and $\Gamma(X) > 0$ at $\bar{X}$ guarantee that there must be at least one $X^* \in (\underline{X}, \bar{X})$ which satisfies the condition $\Gamma(X) = 0$. Therefore, the dynamic system possesses at least one steady state solution. ■

8.3 Phase diagram analysis: computer simulation

As generally recognized, it is not possible to analytically depict phase diagrams for nonlinear differential equations in models involving multiple state and control variables because of the interdependence of the variables (Li and Lőfgren, 1998). Hence, with the help of computer simulation, we apply numerical methods in this section to conduct the phase diagram analysis and to investigate the qualitative properties of the solution of the dynamic problem. To do this, a modified version of the computer program originally developed by Martin Quaas (personal communication) is completed by using *MATHEMATICA* (see appendix 8.1). [30] The basic procedure of the computer simulation is demonstrated as follows.

[30] I thank Martin Quaas, of the Interdisciplinary Institute for Environmental Economics of University of Heidelberg, Germany, for his generosity to share his idea with me. All remaining errors are mine alone.

First, the following functional forms are specified:

$$\begin{aligned}
F(X) &= \rho X(1-\frac{X}{100}) \\
U(h,X) &= 100X^{\frac{1}{2}}h^{\frac{1}{2}} \\
C(I) &= \frac{1}{40}I^2 \\
W(X,K) &= \frac{X^2}{2K}
\end{aligned} \tag{8.13}$$

where $\rho = 1$. In addition, we specify a discount rate $r = 0.05$ and a depreciation rate of the management capital $\delta = 0.5$. By using the similar technique applied in sections 7.6.1 and 7.6.2, we can transform the differential equation system (8.4), (8.5), (8.10) and (8.11) into the following system:

$$\begin{aligned}
\dot{X} &= F(X) - W(X,K) - h \\
\dot{K} &= I - \delta K \\
\dot{h} &= \frac{1}{U_{hh}}((r - F'(X) + W_X)U_h - U_X - U_{hX}\dot{X}) \\
\dot{I} &= \frac{1}{C'}((r+\delta)C' + W_K U_h).
\end{aligned} \tag{8.14}$$

Under these specifications, the steady state solution (X^*, K^*, h^*, I^*) of the differential equation system (8.14) can be obtained, with $X^* = 52.1514$, $K^* = 202.945$, $h^* = 18.253$ and $I^* = 101.472$. The Jacobian of (8.14) evaluated at the steady state solution can also be obtained:

$$J = \begin{bmatrix} -0.3 & 0.0330176 & -1 & 0 \\ 0 & -0.5 & 0 & 1 \\ -1.26 & 0.0577807 & 0.35 & 0 \\ -2.67538 & 0.55 & 1.52879 & 0.55 \end{bmatrix}. \tag{8.15}$$

The eigenvalues are $\zeta_1 = 1.27642$, $\zeta_2 = -1.22642$, $\zeta_3 = 0.815694$ and $\zeta_4 = -0.765694$, with respective eigenvectors $e_1 = (0.66, -0.49, 0.12, -0.86)$, $e_2 = (0.38, -0.70, 0.33, 0.51)$, $e_3 = (-0.03, -0.60, 0.02, -0.80)$ and $e_4 = (-0.12, -0.96, -0.09, 0.25)$. Therefore, the differential equation system (8.14) is saddle point stable.

Next, we determine a neighboring point s_0, which is located on the convergent saddle point trajectory, to the steady state solution by setting $s_0 = s^* + \sum_{i=1}^{4} g_i e_i$, where $s^* = (X^*, K^*, h^*, I^*)$ is the steady state solution, and g_i are arbitrary tiny numbers, for example 0.01. To ensure that s_0 is located on the convergent saddle point trajectory, we specify $g_i = 0$ for i with $\zeta_i > 0$. Using s_0 as the initial point at $t = t_0$, and solving the differential equations (8.14) by numerical method, we can obtain all the points on the convergent saddle point trajectory from t_0 to t_n in a way of time reverse where $t_0 > t_n$, and finally draw them on the phase diagrams. Similarly, the convergent saddle point trajectory from an another direction can be drawn by specifying g_i as a negative tiny number, for example -0.01. Accordingly, both the phase diagram projected on the (X, h) plane conditional on the optimality of the other variables and the phase diagram projected on the (K, I) plane conditional on the optimality of the other variables can be depicted, as figures 8.1 and 8.2 show, respectively.

The two trajectories ´o.p.´ in Figure 8.1 and 8.2, which represent the stable saddle point trajectories, converge to the equilibrium point. It means that, on the (X, h) plane, corresponding to each initial resource stock level, an unique corresponding value of harvest rate could be chosen on the stable trajectory. It follows that, on the optimal dynamic path, the resource stock and the harvest rate increase over time, if the initial resource stock level is less than the steady state resource stock. On the other hand, if the initial resource stock level is higher than the steady state resource stock, the resource stock and the harvest rate decrease simultaneously over time on the optimal dynamic path. The economic meaning of this outcome can be explained as follows. The more resource stock people have, the more they would harvest without influencing the long-run survival of the resource, or in other words, in order to reach the steady state, people would harvest more than the steady state harvest rate when the resource is in relative abundance, in the sense that the resource stock is higher than the steady state resource stock. And they would harvest less than the steady state harvest rate when the resource is relatively scarce, in the sense that the resource stock is less than the steady state resource stock.

Similarly, on the (K, I) plane, corresponding to each initial management capital stock level, a unique value of investment rate could be chosen on the stable trajectories. On the optimal dynamic path, the management capital stock increases while the investment rate decreases over time, if the initial management capital stock level is less than the steady state management capital stock. On the contrary, if the initial management capital stock level is higher than the steady state stock level, the management capital stock decreases over time on the optimal dynamic path while the investment rate increases. The economic meaning of this result can be explained as follows.

The less management capital stock people have, the more investment would be needed in order to accelerate capital accumulation and thereby to reach the steady state. In other words, people would devote less investment than the steady state investment level when the management capital is in relative abundance, in the sense that the management capital stock is greater than the steady state capital stock. And they would invest more than the steady state investment rate when the management capital is relatively scarce, in the sense that the management capital stock is less than the steady state capital stock.

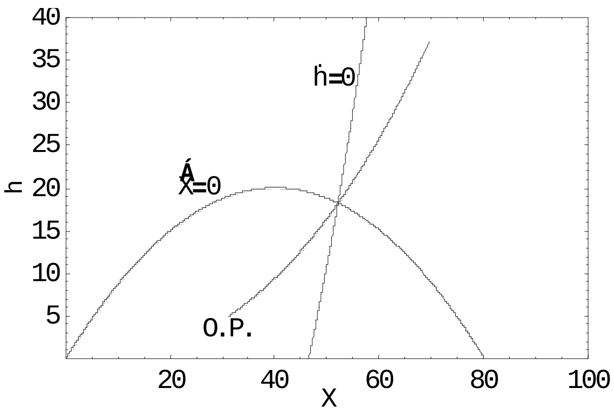


Figure 8.1. Phase diagram on the (X, h) plane

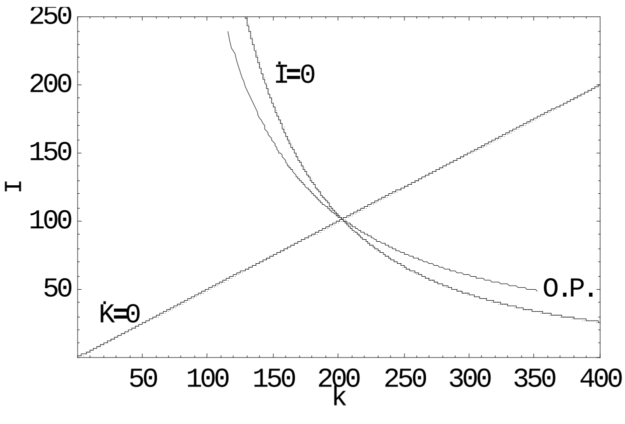


Figure 8.2. Phase diagram on the (K, I) plane

With reference to the time path of the poaching rate, we know that, ceteris paribus, it will decrease when the management capital stock increases, and vice versa. However, the poaching rate depends also on the resource stock level. Thus, unlike the interaction between resource stock and harvest rate or between management capital and investment rate, the development trend of the poaching in general can not be determined when the resource and management capital stock varies. It depends mainly on the initial conditions with regard to resource and management capital stock. To understand this, we may consider the following different scenarios.

First, if the initial resource stock is in relative abundance but management capital is scarce, the resource stock will then decrease while management capital stock increase. It follows that the poaching rate will decrease according to the assumptions of (8.1). On the contrary, if the initial resource stock level is relatively low but management capital is in relative abundance, the resource stock will then increase while management capital stock decrease. This will induce a higher poaching rate. In an another scenario that both the initial resource and management capital are scarce, they will increase simultaneously. An increase in resource stock will induce more poaching, whereas an increase in management capital curtails poaching rate. Therefore, the net effect on poaching rate is ambiguous and depends on the relative strength of the effects derived from resource and management capital stock. It is possible that the building of management regime effectively contributes to the growth of resource stock and simultaneously effectively dampens poaching activity to such a low level so that it results in a net effect of a decrease in poaching rate. However, it may also happen that a conservation project is so successful that the recovery of resource stock induces more poaching than before, although the devotion of management capital has to certain extent slowed down the growth of poaching activity. A similar but reverse rationale can be applied to the case in which both the initial resource and management capital stock level are relatively high.

The previous discussion suggests that, as an indicator for evaluating the success of a management regime, the equilibrium resource stock may be more appropriate than the poaching rate. The conclusion outlined here is opposed to the point of view expressed by Lewis and Phiri (1998). They suggested in a case study in Zambia that wire snare counts, a proxy for poaching activity, can be treated as an indicator for evaluating the success of the community-based conservation projects. However, we asserts here that a reduction in poaching does not necessarily imply the success of a conservation project, if no or only few renewable resources exist in reserves and poachers are therefore not interested on them at all. Likewise, the variation of the management capital stock does not represent the success or failure of a conservation project.

For example, a conservation project may fail, because inappropriate exogenous conditions dominate the project and lead to a low equilibrium resource stock level, while the management capital stock increases as a result of the low initial management capital stock.[31] After all, the renewable resources themselves, rather than the poaching rate, management capital, investment or harvest rate, are the ultimate concern of the conservation policy. From the aspect of conservation, whether the equilibrium resource stock can satisfy the ecological criteria of a sound ecosystem, is the ultimate indicator for judging the extent of success of a conservation policy.

8.4 Comparative static analysis: computer simulation

In addition to the phase diagram analysis, we can also make use of computer simulation to conduct a comparative static analysis, and thereby to investigate the influence of permanent changes in the exogenous parameters on the equilibrium resource stock. To study the impact of as many parameters as possible, some special parameters will be introduced in the functional forms of $F(X)$, $W(X,K)$, $U(h,X)$ and $C(I)$. In what follows, we specify

$$\begin{aligned} F(X) &= \rho X(1-\frac{X}{100}) \qquad (8.16)\\ U(h,X) &= 2\beta X^{\frac{1}{2}}\tau h^{\frac{1}{2}} \\ C(I) &= \frac{1}{2}\sigma I^2 \\ W(X,K) &= \frac{\alpha X^2}{\gamma K}. \end{aligned}$$

The exogenous parameter ρ, as usual, denotes the intrinsic growth rate of the resource stock. The functional form of $F(X)$ shows that the carrying capacity of the environment equals 100. The parameters α, β, γ, σ and τ represents exogenous social, economic, cultural, natural or institutional factors which influence marginal poaching rate, marginal non-consumptive value of the resource, marginal efficiency of the management capital, marginal cost of investment in management capital and marginal gross profit of the harvest, respectively. For convenience, we name these parameters α, β, γ, σ and τ the poaching coefficient, the non-consumptive value coefficient, the efficiency coefficient of management capital, the cost coefficient of investment and the gross profit coefficient of harvest, respectively.

The marginal poaching rate, W_X, measures the impact of one extra unit resource stock on the poaching rate, and it can be influenced by many exogenous factors. For example, compared to protected areas located in the

[31] These exogenous conditions will be investigated in the next section.

heart of the Amazon rain forests, an area of easy access (say because it is in close vicinity to a highway) would have an overall higher marginal poaching rate, or in other words, the poaching function would have a greater α coefficient.[32] As a result of the same geographic factor, it might also have an overall lower marginal efficiency of the management capital, W_K, which measures the influence of one extra unit management capital on the poaching rate, or a smaller γ. The marginal non-consumptive value of the resource, U_X, which represents the influence of one extra unit of resource stock on human utility level, is usually dominated by aesthetic or cultural factors. For example, the panda bear has generally much higher overall level of marginal non-consumptive value, or greater β, than most kinds of snake. Other things being equal, beautiful scenery of a national park often contributes to overall higher level of marginal non-consumptive value of the wild species living in that park. Similarly, some institutional factors have great impact on the marginal gross profit of the harvest, U_h, which denotes the gross profit (unit resource price minus unit harvest cost) derived from one extra unit of harvest. For example, an absolute ban on hunting and trading of wildlife, if it is successful, usually leads to an overall lower level of marginal gross profit of the harvest of wild species, or a smaller τ, because the resource owner cannot use and sell their resources, so that the marginal gross profit of harvesting wildlife will be lower or even become zero. Finally, the parameter σ determines the overall level of the marginal cost of investment in management capital, C_I. In a country with higher price level or higher unit labor costs, for example, the parameter σ would be potentially greater and leads to a higher overall level of the marginal cost of investment in management capital.

Together with the discount rate r and the depreciation rate of the management capital δ, the differential equations system encompasses four endogenous variables, X, K, h, I, and six exogenous parameters, r, δ, α, β, γ, σ, ρ and τ. After repeated tests under different parameter combinations, the results of computer simulation show unambiguously that, other things being equal, the steady state resource stock will increase, when β, γ, ρ and τ are increased, and when r, δ, α and σ are decreased (see Table 8.1). Apart from the two new parameters σ and τ introduced in this chapter, all results of the computer simulation are consistent with those found by comparative static analysis in section 7.6.

[32] Another example is, as Hurt and Ravn (2000) indicated, poaching pressures are positively correlated with the human population density throughout Africa. The more people living in the vicinity to wilderness areas, the more heavy the poaching pressure.

Table 8.1 Results of the comparative static analysis: computer simulation.

	r	δ	α	β	γ	σ	ρ	τ
Comparative static effect on equilibrium resource stock	$-$	$-$	$-$	$+$	$+$	$-$	$+$	$+$

With reference to the comparative static effect of the cost coefficient of investment on the equilibrium resource stock, the model result is understandable. A decrease in the cost coefficient of investment will reduce the overall level of the marginal cost of investment in management capital, thereby induce more investment in management capacity, and in turn result in a higher level of equilibrium resource stock. Of the new comparative static results in this section, what especially worth noting is the impact of an variation of the gross profit coefficient of harvest on equilibrium resource stock. Our model suggests that, other things being equal, an increase in the gross profit coefficient of harvest will lead to a higher equilibrium resource stock level. This result is consistent with what asserted by Swanson (1994), and contrary to the conclusion drawn by Clark (1973). In his model, Swanson asserted that an increase in the price/harvest cost ratio (unit resource price divided by unit harvest cost) will contribute to the growth of the population of the harvested species, and this conclusion is the reverse of Clark´s. The reason for this difference is, that both Swanson´s and our models take the factor of the evolution of management capacity into account, while the Clark model did not. In the context of the Clark model, an increase in the price/harvest cost ratio enhance the incentive to harvest resources, without a concomitant increase in the resource stock resulting from the devotion of a higher management capital. However, in the context of our model, a higher gross profit coefficient of harvest results in increased shadow price of resource stock λ, and this in turn raise the shadow price of management capital stock μ, as equation (8.11) indicates. Therefore, more capital will be devoted to the management of resources, and finally result in a higher equilibrium resource stock.

Can the conclusion be drawn from our model results that a higher gross profit coefficient of harvest (or a higher price/harvest cost ratio in the context of the Clark model and the Swanson model) can contribute to the conservation of harvested species? It does not necessarily, since the model result is based on the premise of ´other things being equal´. As discussed previously, the gross profit coefficient of a harvested species is determined by many exogenous factors. Sometimes, we may try to manipulate the gross profit coefficient in favor of our policy objectives through rearranging the institutional framework which dominates the gross profit coefficient. However, in the real world, there is usually a certain correlation between different parameters, especially between the gross profit coefficient and the poaching coefficient. For example, in order to raise the gross profit coefficient of wildlife, we may lift the ban on wildlife hunting and trading prevailing in many countries. But lifting the ban will probably influence the poaching coefficient, usually raise it simultaneously, because both legal resource owner and poachers are motivated to harvest wildlife after the ban is removed. Therefore, we should consider the situation in which both the gross profit coefficient and the poaching coefficient are increased or decreased. Analytically, the final result must be ambiguous, since the increased gross profit coefficient raises, and the increased poaching coefficient reduces the equilibrium resource stock, and we do not know exactly which effect will dominate the final outcome. Nonetheless, with the help of computer simulation, we can make simulations under different scenarios to explore, when the equilibrium resource stock will increase or decrease if the gross profit coefficient and the poaching coefficient vary simultaneously and in the same direction.

As an example, let us make use of the functional specifications of (8.16), and specify $\rho = 1$, $\delta = 0.5$, $r = 0.05$, $\beta = 50$, $\sigma = 0.05$ and $\gamma = 2$ for running simulation program. Table 8.2 demonstrates some of the simulation results under different τ-α combinations. For example, under the initial scenario $\tau = 0.1$ and $\alpha = 0.1$, it yields a equilibrium resource stock $X^* = 58.79$. When τ and α are increased to 3 and 0.2 respectively, the equilibrium resource stock increases also to 61.78. In fact, as long as τ is increased to 3 and α is smaller or equal to 0.5, the equilibrium resource stock will increase, compared to the initial scenario. On the other hand, when τ is increased to 3 and α is increased to 0.6 or more, the equilibrium resource stock will be smaller than that under the initial scenario. The ´break even´ point for α, in the sense that at which the equilibrium resource stock remains unchanged, lies somewhere between 0.5 and 0.6. In general, we can conclude that, whether the equilibrium resource stock will increase or decrease depends on the extent to how τ and α positively correlate to each other.[33] If the increase in τ is accompanied by only slight increase in α, it will result in a higher equilibrium resource stock. Contrarily, if the increase in τ is accompanied by substantial increase in α, it will possibly lead to a decrease in the equilibrium resource stock.

[33]It is evident that the comparative static effect of a shift of τ will be unambiguous, if there is negative correlation between τ and α.

Table 8.2 Computer simulation results under different τ-α combinations

	$\tau = 0.1$	$\tau = 0.5$	$\tau = 1$	$\tau = 1.5$	$\tau = 2$	$\tau = 2.5$	$\tau = 3$
$\alpha = 0.1$	58.79	61.32	62.07	62.43	62.66	62.83	62.96
$\alpha = 0.2$	55.34	59.22	60.39	60.96	61.32	61.58	61.78
$\alpha = 0.3$	52.56	57.51	59.00	59.74	60.21	60.54	60.80
$\alpha = 0.4$	50.17	56.00	57.78	58.66	59.22	59.63	59.93
$\alpha = 0.5$	48.03	54.64	56.68	57.69	58.33	58.79	59.15
$\alpha = 0.6$	46.10	53.39	55.60	56.79	57.51	58.02	58.42
$\alpha = 0.7$	44.33	52.23	54.71	55.94	56.73	57.30	57.73
$\alpha = 0.8$	42.69	51.14	53.81	55.15	56.00	56.61	57.09
$\alpha = 0.9$	41.15	50.11	52.96	54.39	55.31	55.97	56.47
$\alpha = 1$	39.72	49.13	52.15	53.67	54.64	55.34	55.88

It seems that the somewhat ambiguous conclusion can help policy-makers consider conservation issues in their decision making process, if they know the correlation between τ and α. But in the real world it does not help so much, because in many cases no one knows exactly, ex ante at least, the extent to which how τ and α positively correlate to each other. Just therefrom many prevailing controversy about conservation policy arise. In next section, we will illustrate the policy implications of our comparative static analysis with two important empirical examples.

8.5 Policy implications of the comparative static analysis with regard to the gross profit coefficient of species and the poaching coefficient: two examples

8.5.1 Debate on conservation and consumptive use of the African elephant

During the last two decades, the consumptive use and trading of wildlife were one of the most controversial conservation issues. The experiences of the African elephant conservation typify many elements of the ongoing debate surrounding sustainable use approach versus preservation approach. Hence we use the example here to demonstrate what our model would imply for this case.

During the 1980s, the populations of the African elephant throughout Africa has experienced a massive slaughter to an unacceptable extent. Driven by the extraordinary high price of the elephant ivory in the international wildlife products market, poachers slaughtered more than half of the elephant population in Africa within ten years. According to estimates, poaching has led to a significant decrease in African elephant population, from about 1.3 million in 1979 to six hundred thousand in 1989 (ITRG, 1989). This raised serious public concern about the survival of the African elephant. To save the African elephant, conservation organizations initiated worldwide campaigns aimed at banning the ivory trade. Finally, after some attempts to regulate the ivory trade had failed to dampen poaching activities, the general assembly of the Convention on International Trade in Endangered Species of Flora and Fauna (CITES) decided in 1989 that the African elephant are up-listed, from CITES Appendix II, which allows controlled and monitored trading, to Appendix I which absolutely bans any trading of species concerned (Duffy, 2000).

The logic behind the ban on ivory trade is easy to understand. By destroying the market for ivory, the ivory price will dramatically fall, and this, together with the lack of access to legal market, in turn constitutes a disincentive for poachers to poach again. In fact, at least in the beginning several years after ban was imposed, the ban seemed to have, to certain extent, achieved its objectives. The ivory price has dramatically fallen, poaching activities decreased (Duffy, 2000), and population of the African elephant remained stable.

However, the influence of the trade ban on the poaching rate is not as clear-cut as it was supposed to be by the proponents of the ban. Research by the IUCN African Elephant Specialist Group suggested that, it was mainly the increased efforts on law enforcement, rather than the ban *per se*, which contributed to the decline of poaching at the beginning stage of ban (Dublin et al., 1995). In addition, Zimbabwe claimed that poaching has virtually rose after the ban as a result of a higher illegal market ivory price (Duffy, 2000).

Meanwhile, some active opponents of the ivory ban, mostly the Southern African countries such as South Africa, Zimbabwe, Botswana and Namibia, continued to argue that the ban has virtually punished them for their sound wildlife conservation policy. They asserted that, through wise consumptive use of elephant in which the trading of ivory played a critical role, they have protected their elephant populations and wilderness more effectively than those countries which banned the consumptive use of elephant, and thereby they virtually conserved a sound and stable elephant population, even in the 1980s.[34] In fact, Zimbabwe, as one of the most active proponents of the

[34] For example, in 1989, the Kruger National Park in South Africa culled 350 elephants to prevent overpopulation. The park authority sold the ivory and hides, and earned US$2.5 million which constituted 10% of its annual budget (Rasker et al., 2000).

sustainable use of wildlife, has increased its elephant population since 1981 (Rasker et al., 2000). Using their successful experiences in wildlife conservation, these countries continued to call for lifting of the total ban on ivory trade. In 1997, they succeeded in down-listing their elephant populations to CITES Appendix II, which meant that a restricted ivory trade with Japan was allowed in 1999 under the premise that only legally harvested ivory can be traded and the harvest should be sustainable. The pro-ban conservationists criticized the restricted lifting of ban because of the fear of losing control of ivory trade and the concomitant resurgence of poaching. They argued that, given the circumstances that it is extremely difficult to differentiate illegal ivory from legal ivory, the illegal ivory trading will continue by way of laundering under the cloak of the legal trading (Duffy, 2000).

The brief retrospect about the conservation of the African elephant shows how complex the issue in the reality is. The central question is, whether the ban on ivory trade, as a whole, is advantageous or disadvantageous to the conservation of the African elephant, or more precisely, whether the ban results in an increase or a decrease in elephant population. We are here especially interested in the implication of our model for this question.

In Africa, except in some private reserves, the elephant populations are, de jure, owned by state. Hence we can treat state government as the resource owner in this case. The ban has succeeded in reducing ivory price (a smaller τ in our terminology) and thereby reducing poaching (a smaller α) almost throughout Africa (although its influence on poaching is not as significant as it is supposed to be). For the countries with very low management capacity and in which access to elephant is virtually open, the decline of the overall poaching level is undoubtedly a good news for elephant conservation. However, for those countries which actively manage their elephant populations for consumptive or non-consumptive use, they are faced a smaller α, advantageous for elephant conservation on the one hand, and simultaneously a smaller τ, disadvantageous for elephant conservation on the other hand. According to our model findings, the ´net´ increase or decrease in elephant population derived from the ban will be ambiguous, unless we know, at least roughly, how τ and α correlate to each other. In reality, it is usually difficult to obtain these informations because it needs extensive field studies. Therefore, given current available informations, neither the ban nor lifting of the ban can be unambiguously justified. Any change of 180 degrees of the current policy will take too many risks. Some experiments with regard to lifting the ban are worth being done, but only on an adaptive, trial and error basis. In any case, decisions about whether and where the ban should be lifted depend primarily on the correlation between ivory price and poaching rate.

In addition, our model suggests that we may try to raise τ, while keep α unchanged, to get a greater equilibrium resource stock. From this perspective, the safari hunting in Africa is a good alternative which will promote elephant conservation through consumptive use. According to the recent study made by Hurt and Ravn (2000), sport hunters usually pay a licence fee of US$10,000 for a single elephant, and the total price of an average 21 day safari hunting, where elephant is the main trophy, is more than US$40,000. In general, the greatest cash return on a single elephant from consumptive use is usually the licence fee paid by sport hunters. Therefore, the introduction of safari hunting substantially enhances the overall level of the marginal consumptive value of elephant, or in other words raises τ. Most importantly, the high price holds only for the elephant owner. For poachers, their cash return from killing an elephant remains unchanged, and this guarantees that the poaching coefficient α remains also unchanged. In this case, the introduction of safari hunting will unambiguously lead to a greater equilibrium elephant population. This may, at least partly, explains why South Africa and Zimbabwe, as two of the most active proponents of the sustainable use of elephant and major safari hunting destinations, have a healthy and increasing elephant population. Therefore, we may conclude that, from the perspective of elephant conservation, safari hunting is a feasible policy option.

In fact, the same rationale holds also for other game species. As a result of the boom of wildlife use, especially safari-hunting, almost every private ranches and reserves in South Africa have re-introduced wildlife (Hurt and Ravn, 2000), and this has resulted in steady increase in wildlife populations (Grootenhuis and Prins, 2000) and increase in privately protected habitat area which is even greater than the total land area under the control of the National Park´s Board (Hearne and Mckenzie, 2000). Similar trend with reference to populations of wildlife species and land area devoted to wildlife use can also be found in Zimbabwe where more than 30% of the country´s land area are devoted to some form of wildlife use (Kock, 1996). Our model and the experiences in Southern Africa clearly demonstrate the safari hunting´s conservation component. The more lucrative safari hunting is, the more it can contribute to the conservation of the utilized species, and most importantly, to the conservation of habitats and biodiversity sharing habitats with the target species. According to our model, any consumptive use strategy of renewable resources, which can enhance the overall level of the marginal consumptive value of resources and simultaneously keep poaching unchanged (or even reduce poaching), will be adequate option for conservation policy instrument, provided the other parameters discussed in section 8.4 are in principle positive or at least neutral from the perspective of conservation.

8.5.2 Conservation and consumptive use of wildlife in Taiwan

Taiwan had once a rich and diverse fauna world. But in the past several decades, severe habitat degradation and poaching have significantly reduced population levels of almost all wildlife species, especially those of traditionally important game species. Given that more than 50% of the land area of Taiwan is still covered by healthy forests, poaching is, to great extent, responsible for the disappearance of wildlife. For Taiwanese, the so-called ´wild meat´, which means the meat and any eatable parts of wildlife, are valuable delicacies. The prices of wild meat are usually much higher than those of domestic animals. The high prices drive many people, indigenous or non-indigenous, to go hunting for commercial purpose, while some indigenous people retain their hunting tradition for subsistence or cultural purposes. Given the de facto open access state of wild species, those commercial hunters rapidly harvested almost all of the wildlife resources of Taiwan.

To save wildlife from going extinction, an absolute ban on hunting of wildlife was imposed by the central government in 1973. In 1989, the more strict Wildlife Conservation Law, which follows the strict preservation model and may be one of the most strict wildlife law throughout the world, was enacted. According to the Wildlife Conservation Law, all wildlife species[35] should be classified into two categories: (1) Protected Species, which include endangered, rare, valuable and other conservation-deserving species; (2) General Wildlife, which include all wildlife species not included in Protected Species (WCL, 1994: Article 4). All wildlife species and their products are prohibited from being disturbed, abused, hunted, killed, traded, exhibited, displayed, owned, imported, exported, raised or bred, except in some special cases (WCL, 1994: Article 16, 18, 21 and 24).[36] Hence, any use of

[35] The Article 3 of the Wildlife Conservation Law defines wildlife as ´any animal living in a natural habitat, including mammals, birds, reptiles, amphibians, fish, insects and other kinds of animals.´ (WCL, 1994: Article 3).

[36] The Wildlife Conservation Law permits hunting or utilization of wildlife species only in the following cases: ´(1) when population size exceeds the carrying capacity of the area; or (2) for academic research or educational purposes and with proper approval from the NPA.´(WCL, 1994: Article 18) and ´(1) danger to public safety or human life; (2) damage to crops, poultry, livestock or aquaculture; (3) being a disese vector of zoonoses or other pathogens; (4) danger to the safety of air transportation; (5) for traditional cultural or ritual hunting, killing or utilization needs of Taiwan aborigines living in reserved areas;

wildlife for commercial or subsistence purposes is virtually prohibited by the Wildlife Conservation Law.

The enactment of the Wildlife Conservation Law and the concomitant increased efforts on law enforcement against poaching has succeeded in damping poaching during the last decade. Populations of most of the important game species have, to certain extent, recovered (Pei, 2001). However, the absolute ban on wildlife use also induced the protest of the indigenous people who call for legal hunting rights for both economic and cultural purposes. The fact that some indigenous subsistence hunters were treated as poachers by the Wildlife Conservation Law further intensified the conflicts between indigenous people and government authorities, though usually only slight punishments were inflicted on these hunters because of their special identity. Given that protest continues to intensify and that populations of some important game species have apparently recovered, there are recently more and more people who suggest modifying the Wildlife Conservation Law (and/or the National Park Act) to allow legal hunting and utilization of wildlife, and thereby to promote both conservation and sustainable development projects initiated by indigenous communities (Liu, 2000). On the other hand, like the debate prevailing throughout the world, many people are averse to this idea because they fear the concomitant resurgence of poaching if the hunting ban is lifted (Chang, 2001).

Should the ban on wildlife hunting and utilization be lifted in Taiwan? Again, we assert here on the basis of the model findings that the answer to this question is not simple yes or no. It depends, except the other six parameters discussed in section 8.4, primarily on the influences of the policy options on the marginal gross profit of harvest of wildlife and on the poaching rate. If it involves the reopening of hunting and of trading of wildlife meat and products, poaching will probably boom again while the marginal gross profits of harvest of wildlife are increased, because it is practically impossible to differentiate illegal from legal hunted wild meat and wildlife parts, regardless of in the market or on the dishes of the so-called ´wild meat restaurant´.[37] According to author´s personal observations and of Chang (2001), even under the control of the Wildlife Conservation Law, wild meat restaurants are still very popular in the remote country, and the ´raw material´ of their delicacies are, to great extent, illegal hunted wildlife. Therefore, it is

(6) other reasons approved by the authorities.´ (WCL, 1994: Article 21). With reference to cultural or ritual hunting, each indigenous tribe is allowed to hunt twice in a year.

[37] ´Wild meat restaurant´ is a special kind of restaurant in Taiwan which sell primarily delicacies made from wild meat and wild species of plants.

reasonable to conjecture that, once legal trading of wildlife meat and parts is reopened, the poaching rate will significantly increase under the cloak of the legal trading. Given the current circumstance that the management capacity of areas out of national parks is deficient and access to wildlife is practically open, the negative impact on wildlife populations will be inevitable in national forests and nature reserves. For those community based conservation projects aimed at sustainable use of wildlife and initiated by local people, the significant increase in poaching rate will reduce their success probability, although, according to our model, the higher marginal gross profit of harvest of wildlife may compensate the negative effect derived from increased poaching rate. But even legal hunting and trading of wildlife can contribute to an increase in wildlife populations in some private protected areas, it cannot offset the population loss occurred in national forests and nature reserves, because the great majority of the wilderness remains under the control of the state.[38] It follows clearly that, from the perspective of wildlife conservation, the combination of reopening hunting and trading of wildlife meat and parts may not be an adequate policy option under current circumstances of Taiwan.

To promote community based conservation projects, or to mitigate the conflicts between indigenous people and state authorities, we may consider the policy option of legalizing sport hunting (and fishing), while the other forms of consumptive use of wildlife are still not allowed. As discussed in subsection 8.5.1, sport hunting can significantly enhance the marginal gross profit of harvest of target species, while the poaching rate remains unchanged.[39] This, as a whole, will result in an increase in equilibrium population of the target species, and simultaneously promote the protection of habitat. In Taiwan, some wildlife species which are resilient to harvest and have relatively abundant populations, such as Formosan wild boar (*Sus scrofa taivana*), Formosan hare (*Lepus siensis formosanus*), Formosan giant flying squirrel (*Petaurista granis*) and Formosan white-faced flying squirrel (*Petaurista lena*), can be adequate game species. Some heavily harvested big game species, such as Formosan Reeve´s muntjac (*Muntiacus reevesi micrurus*), Formosan sambar (*Cervus unicolor swinhoei*) and Formosan serow (*Naemorhedus swinhoei*), are also valuable and adequate game

[38] The state-owned wilderness in Taiwan can be roughly divided into the following three categories: national parks, nature reserves and national forests. In general, only national parks are practically strictly protected. Nature reserves and national forests, de jure, should also be protected but practically not as a result of the low management capacity of the authorities concerned.

[39] If those poachers, usually skilled hunters, can be transformed into legal hunting guides or rangers through legalizing sport hunting, the poaching rate will probably decrease.

species, but we still need scientific assessments to identify their population levels before they are hunted. From the perspective of wildlife conservation, sport hunting should be encouraged, and may be the only feasible form of sustainable wildlife use in today´s Taiwan.

8.6 Concluding remarks and some implications for conservation policy

In this chapter we have investigated the relevant properties of the general model. By use of the computer simulation, the relevant phase diagrams and the impacts of exogenous parameters on equilibrium resource stock have been studied. The comparative static effects of the general model are similar to those of the extended model presented in chapter 7, which is a special case of the general model. The implications of the comparative static analysis for specific species conservation policy have also been explored with two examples. In addition, we assert here that the comparative static effects of the general model can be used to assess the feasibility of the sustainable use strategy applied in specific areas. We will show how this works in the case study of the next chapter.

8.6.1 Some remarks

There are still some points that are noteworthy. First, like the policy implication of the simple model, the impact of the sustainable use approach on conservation is double-edged in the sense that the sustainable use approach will not necessarily result in a higher stock level of renewable resources and/or management capital than before. In some cases, it can cause a lower stock level of renewable resources and/or management capital. Moreover, the impact of the sustainable use approach on conservation is here far more complex than that in the simple model. Figure 8.1 demonstrates, that the sustainable use approach will theoretically contribute to an increase in resource stock through the adjustment of harvest rate and management capital stock, if the initial resource stock is smaller than the equilibrium resource stock. Otherwise, if the initial resource stock level is higher than the equilibrium resource stock level, the use approach will inevitably lead to a decrease in resource stock until the equilibrium resource stock is reached. The time path of the accumulation of management capital exhibits a similar pattern, as figure 8.2 shows. On the one hand, the management capital stock will increase through the adjustment of investment rate, if the initial stock level is lower than the equilibrium stock level. On the other hand, if the initial management capital stock level is higher than the equilibrium stock level, the management capital stock will decrease steadily until the equilibrium stock level is reached. From the previous discussions it follows that the sustainable use approach will not necessarily result in a better conservation status of renewable resources.

How the accumulation of the management capital influence the growth of the resource stock is a crucial problem in which we are interested. The general model demonstrates that an increase in management capital will, ceteris paribus, reduce the poaching rate and thereby contribute indirectly to the accumulation of resource stock, but it does not guarantee an simultaneous increase in resource stock. On the contrary, a decrease in management capital will cause a higher poaching rate and thereby contribute indirectly to the detriment of resource stock, but it does not necessarily lead to an simultaneous decrease in resource stock. Whether the resource stock will grow or decrease depends on the relative size of the initial resource stock compared to the equilibrium resource stock, and on the corresponding adjustment of the harvest rate. This conclusion does not imply that the accumulation of the management capital does not play any role in the accumulation process of the resource stock. It means that the accumulation of management capital influences but can not definitely determine the accumulation process of the resource stock. The previous discussion suggests that the variation of management capital stock is not an appropriate indicator for evaluating the performance of the sustainable use approach as a conservation instrument.

We are also interested in the question how the poaching activity will develop after the sustainable use approach is applied. The discussion in section 8.3 exhibits that in the case which initial resource stock level is lower than and management capital stock is higher than equilibrium stock level, and in the case which initial resource stock level is higher than and management capital stock is lower than the equilibrium stock level, the development trend of poaching rate can be derived. However, in the cases which both initial resource stock and management capital stock is greater than the equilibrium stock, and which both initial resource stock and management capital stock is smaller than the equilibrium stock, the development trend of poaching rate is ambiguous. In any case, as section 8.3 concluded, the variation of the poaching rate is not an appropriate indicator for evaluating the success of the sustainable use approach. From the aspect of conservation, whether the equilibrium resource stock can satisfy the ecological criteria of a sound ecosystem is the ultimate indicator for judging the extent of success of a conservation policy. Under the general premise that the more the equilibrium resource stock closes to the carrying capacity, the better it would be for the whole ecosystem, we can use the equilibrium resource stock as an indicator for judging the extent of success of the use approach as a conservation instrument, and accordingly turn our attention to the factors which affect the size of the equilibrium resource stock and the corresponding policy implications.

8.6.2 Policy implications with regard to the intrinsic growth rate

By the application of the comparative static analysis, some important parameters affecting the equilibrium resource stock are identified. Of the eight parameters, the intrinsic growth rate of species is a well-known biological factor. The outcome of the comparative static analysis shows that an increase in the intrinsic growth rate will unambiguously raise the equilibrium resource stock. Based on this conclusion, some policy implications with reference to conservation can be drawn.

At the individual species level, the higher the intrinsic growth rate of the species is, the more appropriate the use approach will be for the management of the species. For species with low intrinsic growth rates, a specially cautious attitude toward the harvest problem must be taken. In general, long-lived and slow-reproducing species, such as primates, elephants, whales and sharks, have low intrinsic growth rates and may be particularly vulnerable to harvest (Mangel et al., 1996).[40] Moreover, the intrinsic growth rate is affected in principle by two factors, i.e. Body size and Phylogeny. Larger animals tend to have lower rates (Eisenberg, 1980), and, as a group, primates and carnivores have generally lower intrinsic growth rates than expected from body size, whereas ungulates and rodents have higher rates for their body size (Robinson and Redford, 1986; Bennett and Robinson, 2000b).[41]

Next, as a whole, various habitat types can be characterized by different reproductive capacities of total biomass. Table 8.3 illustrates estimates of the mean net primary production and mean plant biomass for various habitat types of the world.[42] Under the premise that the production/biomass ratio, just like the intrinsic growth rate at the individual species level, can be treated as an appropriate indicator for the reproductive rate of the whole plant biomass of specific habitats, we may conclude that, from the viewpoint

[40] An example for the mahogany tree with low rate of increase and the possible fate of its sustainable harvest see Gullison (1998).

[41] For more detailed estimates about the intrinsic growth rate of individual species of neotropical animals see Robinson and Redford (1991), and Robinson and Bennett (2000).

[42] Stiling (1992) defined the concept net primary production as gross primary production minus energy lost by plant respiration. And gross primary production is equivalent to the energy fixed in photosynthesis. For more details about net primary production see Stiling (1992) and Ricklefs (1990).

of plant conservation, the higher the production/biomass ratio of habitats is, the more appropriate the use approach will be for the management of the habitats. A comparison of the production/biomass ratios of the earth´s major ecosystem types shows that, in general, the production/biomass ratio is inversely related to the overall degree of forest cover. For instance, open grasslands, including Savannah and temperate grassland, have much higher production/biomass ratios than forest ecosystems. This may suggest that, compared to open grasslands, forest ecosystems are particularly vulnerable to overharvesting of plant communities. Such a general comparison neglects the considerable differences in reproductive rates of plants between specific species and regions, but this deficiency does not override the general patterns it reveals.

With reference to the total faunal biomass and the relevant reproductivity of the earth´s major ecosystem types, there is nowadays no similar general estimates like those of plant biomass, but some estimates in tropical ecosystems regarding game biomass, in which people are especially interested, reveal also specific patterns. A comparison of large mammal biomasses[43] at various tropical ecosystems made by Robinson and Bennett (2000) showed that, in general, the overall standing mammalian biomass has a negative relation to the degree of forest cover, and this difference in mammalian biomass can in principle be accounted for by the difference in ungulate biomass. It lacks definite estimates regarding the overall reproductive rate of mammalian biomass. However, based on the estimates of the components of the mammalian communities in tropical ecosystems[44], some important facts emerge. The mammalian biomass of the tropical forests, especially those in tropical rain forests, encompasses a much higher proportion of primates which have generally low intrinsic growth rates, than open grasslands or habitats with a mosaic of forest and grassland do. Contrarily, the mammalian communities of the open grasslands are made up of almost only ungulates and rodents which have generally much higher intrinsic growth rates than primates have. Therefore, the conjecture that the weighted mean intrinsic growth rate of the total mammalian biomass of tropical ecosystems is inversely related to the degree of forest cover should be reasonable. This may suggest that, compared to tropical open grasslands, tropical forest ecosystems are particularly vulnerable to overharvesting of mammalian communities. Again, it is notable that such a general conclusion neglects the considerable differences in

[43]The large mammals are defined as the species with over 1 kg adult body mass. This definition includes most of the important game species.

[44]For more detailed data about the components of the mammlian biomass see Robinson and Bennett (2000), pp. 17-18.

reproductivity between specific species and regions, though this deficiency does not override the general patterns it reveals.[45]

8.6.3 Policy implications with regard to some other parameters

The comparative static analysis addresses also the comparative static effects of the other parameters on the equilibrium resource stock. In sum, the lower the discount rate, the poaching coefficient, the cost coefficient of investment and the depreciation rate of management capital, and the higher the non-consumptive value coefficient, the gross profit coefficient of harvest and the efficiency coefficient of management capital are, the higher the equilibrium resource stock will be. Accordingly, we can use these parameters as indicators for evaluating the success probability of a sustainable use project before or when it is brought into practice. The sustainable use strategy may potentially be more appropriate in sites with more positive indicators than those sites with less positive indicators. Based on this conclusion, the following important policy implications emerge.

For convenience of discussion, we divide roughly all the countries or regions into two categories: the developed and developing countries (regions). The two groups have markedly different features in some of the previous parameters. First, compared to the developed countries, the developing countries are generally characterized by higher levels of discount rate as a result of the prevailing poverty (Bodmer et al., 1997b), the uncertainty with regard to tenure, markets and population levels of exploited species (Freese, 1997),[46] and high levels of inflation rate. As Clark (1973) demonstrated in his pioneer paper, a high discount rate has a destructive effect on the sustainable use of resources, since the relatively slow-reproductive resources will potentially never generate a competitive return on owner´s investment, and then the rapid depletion of the resources becomes a rational option. Given the generally prevailing high discount rate in developing countries, it is difficult for the species with low intrinsic growth rates that the use strategy can create strong enough incentives so that biological over-harvesting would not happen.[47]

[45] For more examples discussing the differences in wildlife production between various tropical forest types see Hart (2000) and Peres (2000).

[46] As Freese (1997) explained, uncertainty with regard to tenure, markets and population levels of exploited species increases the risk premium of holding resource stock and raises accordingly the discount rate. The developed countries have similar but less serious problems than developing countries, because property rights are generally secured and population levels of exploited species are, though not perfectly, far more known than in developing countries.

[47] Milner-Gulland and Mace (1998) illustrated an example of a tragically very high dis-

In addition, as a result of poverty and deficiency of legal enforcement, the natural resources in developing countries are generally under a much greater pressure of poaching activity than in developed countries. This implies that the developing countries have generally a higher poaching coefficient. Moreover, it is in principle undoubted that the non-consumptive value of renewable resources is much more appreciated by the people in developed countries than in developing countries. It follows that the non-consumptive value coefficients of the utility function in the developed countries should be generally higher than those in developing countries.

In sum, compared to the developing countries, the developed countries have some more appropriate socio-economic conditions when we are concerned about the feasibility of the sustainable use approach as a conservation instrument. In other words, given the social and economic conditions, the success probability of the sustainable use approach might be relatively low in developing countries which are the focus of the current conservation practice and of the debates about conservation issues. This might imply that a more conservative attitude toward the application of the sustainable use approach should be taken in developing countries, and the enthusiasm of some international organizations, local people and conservationists for the use approach should be questioned. The previous policy implication holds especially for the tropical rain forests where the overall reproductive rate of faunal and floral communities is low. If we additionally consider the fact that tropical rain forests species are generally characterized by high diversity and low densities (Owen, 1992), it is worthwhile being particularly cautious of reconsidering the use approach to prevent from local or even global extinctions of species. Contrarily, in developed countries a more active attitude toward the application of the sustainable use approach might be taken to supplement the traditional preservation approach.[48] In tropical open grasslands where the overall reproductive rate of plant and mammalian biomass is high, depending on the socio-economic circumstances, a neutral attitude could be taken. Generally speaking, as a result of the relatively low probability of success in developing countries, the use approach can hardly create systems of a scale sufficient to preserve large portions of ecosystems in developing countries which are the focus of the current international conservation campaign.[49] At most, it

count rate happened in the Ache, an indigenous group living in Paraguay. The high discount rate led rapidly to the depletion of natural resources, after they received legal title to their reservations in 1988. According to author´s personal experiences, this is not an accidental example. The indigenous people in Taiwan have similar problems.

[48] A retrospect about the successful experiences of sustainable harvest of wildlife in North America see Shaw (1991).

[49] The experience of South Africa may be an exception. As a result of the boom of

could play a supplementary role in the whole conservation policy. Whether this implies that we should re-emphasize the importance of the preservation approach as several decades ago did and apply it more intensively when creating new protected areas, or that another alternative approach should be developed, is a critical question worth investigating.

Certainly, the previous general conclusion neglects the considerable differences in socio-economic and biological conditions between various countries, regions and habitat types. In certain circumstances the sustainable use approach can succeed even in developing countries and in cases in which the intrinsic growth rate of the harvested species is low, such as the successful use and conservation of the African elephant in Southern Africa (Roth, 1997a). For some species with high intrinsic growth rates, such as Wild boar and feral pigs, the use approach becomes an old tradition around the world and even a necessity to control the damage to agriculture they cause (Roth, 1997b), and the populations remain relatively abundant almost irrespective of under which socio-economic circumstances.[50] Therefore, whether the use or the preservation approach is appropriate, depends always on the site- and species-specific conditions. Nonetheless, the general conclusion affords a fundamental direction for the rethinking of the conservation policies.

---

the wildlife use, especially safari-hunting, about 4,000 private ranches and reserves have devoted totally over 80,000 km^2 land area to wildlife, compared with less than 10,000 km^2 in 1979. This is also remarkable for its extent, compared with the total land area of about 28,000 km^2 under the control of the National Park´s Board (Hearne and Mckenzie, 2000).

[50]There are a few exceptions to this rule. Some of the localised Asian pig species are certainly overharvested so that their populations decrease to a alarmingly low levels. For detailed list of these species see Roth (1997b).

Table 8.3 Primary production and plant biomass of the earth´s major ecosystem types

Ecosystem Type	M.N.P.P.[1]	M. B.[2]	P/B Ratio[3](%)
Continental			
Tropical rain forest	2200	45.0	4.89
Tropical seasonal forest	1600	35.0	4.57
Temperate evergreen forest	1300	35.0	3.71
Temperate deciduous forest	1200	30.0	4.00
Boreal forest	800	20.0	4.00
Woodlands and shrubland	700	6.0	11.66
Savannah	900	4.0	22.50
Temperate grassland	600	1.6	37.50
Tundra and alpine	140	0.6	23.33
Desert and semidesert scrub	90	0.7	12.86
Cultivated land	650	1.0	65.00
Swamp and marsh	2000	15.0	13.33
Lake and stream	250	0.02	1250.00
Marine			
Open ocean	125	0.003	4166.00
Upwelling zones	500	0.02	2500.00
Continental shelf	360	0.01	3600.00
Algal beds and reefs	2500	2.0	125.00
Estuaries	1500	1.0	150.00

Source: Columns 1-3 From Whittaker (1975). Column 4 from personal calculation.

Note 1. M.N.P.P.=Mean Net Primary Production. Units are dry grams per square meter per year.

Note 2. M.B.=Mean Biomass. Units are kilograms per square meter per year.

Note 3. P/B Ratio=M.N.P.P./M.B..

Appendix 8.1: Computer program

```
ρ = 1;
δ = 5/10;
r= 5/100;
α = 1;
β = 50;
σ = 1/20;
γ = 2;
τ = 1;

f[x_] := ρ x(1 − x/100) ;
u[x_,h_] := 2β x^ (1/2) τ h^(1/2) ;
c[i_] := (1/2) σ i^2;
w[x_,k_] := α (x^2) / (γk) ;

dx[x_,k_,h_,i_] =f[x] −w[x,k] −h;
dk[x_,k_,h_,i_] =i−δk;
dh[x_,k_,h_,i_] =
  (D [u [x,h] , h] (r-D [f [x] , x] + D [w [x,k] , x]) − D [u [x,h] , x] − D [u [x,h] , h, x] dx [x,k,h,i]) /
    D[u [x,h] , {h,2}] ;
di[x_,k_,h_,i_] = (D [c [i] , i] (r+δ) + (D [u [x,h] , h] D [w [x,k] , k])) /D[c [i] , {i,2}] ;

NSolve[{dx [x,k,h,i] == 0, dk [x,k,h,i] == 0, dh [x,k,h,i] == 0, di [x,k,h,i] == 0}, {x,k,h,i}]

%[[3]] //N

xs=x/.%18 [[3]]
ks=k/.%18 [[4]]
hs=h/.%18 [[1]]
is=i/.%18 [[2]]

jm={{D[dx [x,k,h,i] ,x], D[dx [x,k,h,i] ,k], D[dx [x,k,h,i] ,h], D[dx [x,k,h,i] ,i]},
     {D[dk [x,k,h,i] ,x], D[dk [x,k,h,i] ,k], D[dk [x,k,h,i] ,h], D[dk [x,k,h,i] ,i]},
     {D[dh [x,k,h,i] ,x], D[dh [x,k,h,i] ,k], D[dh [x,k,h,i] ,h], D[dh [x,k,h,i] ,i]},
     {D[di [x,k,h,i] ,x], D[di [x,k,h,i] ,k], D[di [x,k,h,i] ,h], D[di [x,k,h,i] ,i]}}/.
     {x→xs, k→ks, h→hs, i→is};

TableForm[jm]

evs=Eigevalues[jm]
```

```
evcts=Eigenvectors[jm]

e1=evcts[[2]]

e2=evcts[[4]]

lsg=Fuction[{x0, k0, h0, i0},
   solc=NDSolve[{
       x´[t] ==dx[x [t] , k [t] , h [t] , i [t]],
       x[0] ==x0,
       k´[t] ==dk[x [t] , k [t] , h [t] , i [t]],
       k[0] ==k0,
       h´[t] ==dh[x [t] , k [t] , h [t] , i [t]],
       h[0] ==h0,
       i´[t] ==di[x [t] , k [t] , h [t] , i [t]],
       i[0] ==i0,}, {x, k, h, i}, {t, -7.5, 0}];
   scx[t_] :=Evaluate[x [t] /.solc [[1, 1]]] ;
   sck[t_] :=Evaluate[x [t] /.solc [[1, 2]]] ;
   sch[t_] :=Evaluate[x [t] /.solc [[1, 3]]] ;
   sci[t_] :=Evaluate[x [t] /.solc [[1, 4]]] ;
  Table[{t, scx [t] , sck [t] , sch [t] , sci [t] }, {t, -7.5, 0, 1/100}]];

lsg[xs, ks, hs, is] ;

Clear[xn1, hn1] ;
xn1=Join[Table[{x, Evaluate[f [x] − w [x,ks]]}, {x,0,xs-1/100,1/100}],
        Table[{x, Evaluate[f [x] − w [x,ks]]}, {x,0,xs-1/100,xs+1/100,1/100}],
          Table[{x, Evaluate[f [x] − w [x,ks]]}, {x,0, xs+1/100,100,1/100}]];
hnlfunc[y_] :=Evaluate[(2x (r-D [f [x] ,x] + D [w [x,k] ,x]) − f [x] + w [x,k]) /.{x → y, k → ks}]
hnl=Table[{x, hnlfunc [x] }, {x, 1, 100, 1/100}] ;

Clear[kn1, in1] ;
kn1=Join[Table[{k,δk}, {k, 0, ks-1/100, 1/100}] ,
Table[{k,δk}, {k, ks-1/100, ks+1/100, 1/100}] ,Table[{k,δk}, {k, ks+1/100, 400, 1/100}]];
inlfunc[x_] :=Evaluate[-D [u [x,h] , h] D [w [x,k] , k] /σ (r+δ))/.{x → xs, h → hs, k → x}]
in1=Table[{k, inlfunc [k] },{k, 1/1000, 400, 1/100}] ;

Clear[sc]
sc[h1_,h2] :=sc[h1,h2] =lsg[xs+h1 e1[[1]] +h2 e2[[1]] ,
      ks+h1 e1[[2]] +h2 e2[[2]] , hs+h1 e1[[3]] +h2 e2[[3]] ,
      is+h1 e1[[4]] +h2 e2[[4]] ;
```

```
xh[h1_,h2_] :=Transpose[{Transpose [h1,h2]] [[2]], Transpose[sc [h1,h2]] [[4]]}]

Show[
 ListPlot[xn1, PlotRange→ {{0, 100}, {0, 40}}, PlotJoined→True,
 DisplayFunction→Identity, Axes→False, Frame→True,
 FrameLabel→ {´´x´´, ´´h´´}],
 ListPlot[hn1, PlotRange→All, PlotJoined→True,
  DisplayFunction→Identity],
 ListPlot[xh[1/300, 1/300], PlotJoined→True, Plotstyle→Hue[0] ,
  DisplayFuction→Identity],
 ListPlot[xh[−1/100, −1/100], PlotJoined→True, Plotstyle→Hue[0.8] ,
  DisplayFuction→Identity],

Graphics[Text[StyleForm[´´ẋ=0´´, FontSize→10, FontWeight→´´Bold´´],
{30,22},{1,1},Frame→True, DisplayFunction→Identity],
Graphics[Text[StyleForm[´´ḣ=0´´, FontSize→10, FontWeight→´´Bold´´],
{51,35},{0,1},Frame→True, DisplayFunction→Identity],
Graphics[Text[StyleForm[´´o.p.´´, FontSize→10, FontWeight→´´Bold´´],
{37, 5},{1,1},Frame→True, DisplayFunction→Identity],
 DisplayFunction→ $DisplayFunction];

Ki[h1_,h2] :=Transpose[{Transpose [sc [h1,h2]] [[3]], Transpose[sc [h1,h2]] [[5]]}];
Show[
  ListPlot[kn1, PlotRange→ {{0, 400}, {0, 250}}, PlotJoined→True,
   DisplayFunction→Identity, Axes→False, Frame→True,
  FrameLable→ { ´´k´´, ´´I´´}],
   ListPlot[in1, PlotRange → All, PlotJoined → True, DisplayFunction → Identity] ,
    ListPlot[ki[1/120, 1/120], PlotJoined→True, Plotstyle→Hue[0] ,
      DisplayFuction→Identity],
    ListPlot[ki[−1/30, −1/30], PlotJoined→True, Plotstyle→Hue[0.8] ,
      DisplayFuction→Identity],


Graphics[Text[StyleForm[´´k̇=0´´,FontSize→10, FontWeight→´´Bold´´],
{50,44},{1,1},Frame→Ture, DisplayFunction→Identity],
Graphics[Text[StyleForm[´´İ=0´´,FontSize→10,FontWeight→´´Bold´´],
{165,220},{0,1},Frame→Ture, DisplayFunction→Identity],
Graphics[Text[StyleForm[´´o.p.´´,FontSize→10,FontWeight→´´Bold´´],
{400, 60},{1,1},Frame→True, DisplayFunction→Identity],
DisplayFunction→ $DisplayFunction];
```

Chapter 9
Case study: sustainable use and conservation of renewable resources in Danayiku Nature Park at Shan-Mei, Taiwan

In this chapter we will investigate a community-based conservation program in Danayiku Nature Park at Shan-Mei, Taiwan, which was initiated to promote sustainable use and conservation of renewable resources. As the most successful community-based conservation project in Taiwan, Danayiku Nature Park at Shan-Mei has attracted intensive attention of public media, and was usually highly praised as an excellent model for conservation and rural community development. Many rural communities, indigenous or non-indigenous, are therefore enthusiastic about imitating this successful experience. What we here particularly concerned about is the question, which exogenous factors contributed to the success of Danayiku Nature Park. Based on extensive field studies between 1999 and 2002 and the theoretical framework developed in previous chapters, some assessments about the feasibility of the sustainable use strategy in Shan-Mei and in vicinal several indigenous communities can be made. The basic theme is, that the success of Danayiku Nature Park can be attributed to some feasible exogenous conditions that other communities do not possess. This explains to a great extent why sustainable use strategy scored a success in Shan-Mei, while similar projects in vicinal communities failed or did not work so well like Shan-Mei did.[51]

9.1 Background

The 2,200-hectare (ha) Shan-Mei, named ´Saviki´ in Tsou, is an indigenous village of the Tsou[52] located in the A-Li-Shan Township of the Chia-Yi County, a remote montane area of southwest Taiwan (approximately 120.4° east and 23.2° north). It is located in the west part of the A-Li-Shan Mountain Range, and is not too distant from the A-Li-Shan National Scenic Area, one of the popular destinations for tourists, and the Yushan National Park (see figure 9.1). Elevations of the village vary between 400 and 1200m. The

[51] Some accidental factors, such as the personalities of community leaders, may have influenced the outcomes of community-based conservation projects. But these are beyond the scope of our discussion.

[52] Tsou is one of the ten officially recognized indigenous tribes of Taiwan.

village includes about 800 ha. aboriginal reserves and 1400 ha. national forest lands which are covered mainly by subtropical forests. Almost all of the residents are Tsou people. According to Li and Tang (1999), there were about 530 residents in Shan-Mei, but only half of them practically lived in the village in 1999. Therefore, like almost all indigenous villages in Taiwan, Shan-Mei was faced the severe problem of population drain caused by scarce employment opportunities.

The main occupation of the village residents is farming, especially bamboo plantation supplemented by seasonal job opportunities provided by tea plantation. Compared to the Chinese people mainly living on the plain,[53] the income level and standard of living of the village residents are relatively low. The construction of Tai-18 highway passing through A-Li-Shan Mountain Range led to extensive tea and beetle nut plantation in this area since 1980s. The prosperous tea plantation in the last two decades had once significantly improved the standard of living of village residents, because it needed intensive labor inputs and thereby provided many seasonal job opportunities for indigenous people. However, the collapse of prices of tea and bamboo in recent years again put pressure upon the local economy. Nowadays, village residents usually still have to find part time jobs to make ends meet. Getting into debt is a usual phenomenon. To great extent, all these factors are responsible for the prevailing social problems throughout indigenous communities in this area, such as poverty, unemployment, low education level and excessive drinking.

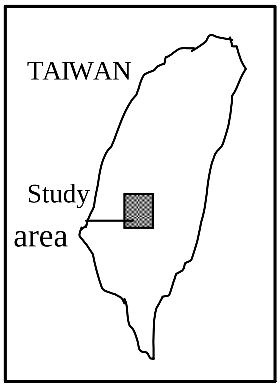
In spite of the prevailing economic and social problems, residents of Shan-Mei accidentally preserved some of the Tsou traditions, while the other Tsou communities rapidly lost their cultural traditions in the last several decades (Li and Tang, 1999). Compared to other Tsou communities, Shan-Mei preserved stronger tribal identity. Individuals are more obedient to the communal decisions and in principle willing to work voluntarily for the community. Chief and elders of Shan-Mei still have great influence on the process of decision-making about public affairs. The traditional fishing territories were still well-defined until poaching boomed. All these have significantly influenced the outcome of the conservation project in the future.

For Shan-Mei, the construction of the Tai-18 highway brought both opportunities and challenges. On the one hand, based on its beautiful scenery and the advantage of adjoining the A-Li-Shan National Scenic Area and the

[53]The Chinese people refer in this chapter to those who originally stem from mainland China.

Yushan National Park, Shan-Mei has good chance to develop tourism after the traffic infrastructure was improved.[54] On the other hand, for poachers coming from outside world, it became more easily to have access to the natural resources of Shan-Mei that can be found primarily in Danayiku. Danayiku is the valley of the Danayiku river, a tributary of the Zeng-Wen river, which originates from the Central Mountain Range (see figure 9.1). The 18 kilometers long Danayiku river is famous among Tsou people for its abundant wildlife and fresh water fish resources. For residents of Shan-Mei, it was the most important origin of protein. However, since the end of the World War II, the Chinese poachers first heavily reduced wildlife populations, and then the subsistence hunters of Tsou also depleted wildlife resources in Danayiku. Even under such situation, the fresh water fish remained abundant until 1970s. The use rights of fresh water fish resources in Danayiku belonged to five major families of Shan-Mei, including Du, Chuang, An, Wang and Yang. Each family owned certain part of the Danayiku river, and had the right to manage and use fresh water fish resources. It is interesting to note that Danayiku river was not only private property. Sometimes communal fishing festival was held by using traditional fishing methods, and the harvests were allocated to all community individuals. Therefore, Danayiku river was, to certain extent, also the community common. The whole property system worked well until the construction of the Tai-18 highway at the middle of 1970s. Thereafter, Chinese poachers came into Shan-mei, used modern technology like chemical poison and electrofishing, and thereby significantly reduced fish populations. The local residents also learned to use modern technology. They soon exhausted all biological resources of the Danayiku river. At the beginning of 1980s, the Danayiku river was virtually dead, and the traditional property right system was broken down (Cheng-Sheng Gau; Cheng-An Chuang, pers. comm.).

[54] In fact, there were private enterprises that were interested in the potential of developing tourism in Shan-Mei and wanted to purchase lands. But these proposals were refused by villagers because of their consensus of retaining autonomous development (Cheng-Sheng Gau, the Village Chief of Shan-Mei, pers. comm.).



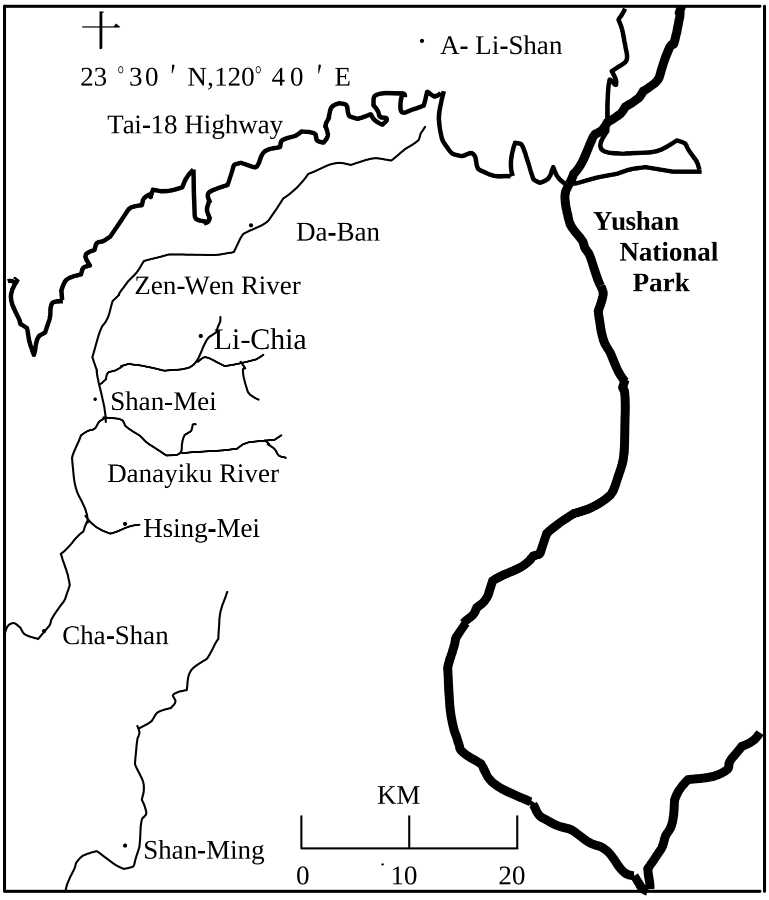


Figure 9.1 Location of the study area.

9.2 Project history and evolution

In 1985, primarily driven by poverty prevailing in Shan-Mei, some leaders and young people of the village tried to find an alternative for improving the standard of living. Considering the proven fact that agriculture had failed to generate enough income for residents, they turned their attention to the possibility of developing tourism. A team named ´Tourism Research Group´ in 1985, which later was transformed into the ´Tourism Promotion Committee´ in 1987, was authorized to assess different alternatives. At the community assembly held in October, 1989, after years long severe debates and discussions within community, they finally decided to initiate a community-based conservation program which aimed at developing ecotourism based on the sustainable use of natural resources, especially the fresh water fish resources of Danayiku (Li and Tang, 1999; Cheung-Mei Yang, the former general secretary of SMCDS, pers. comm.).[55]

The conservation program included several important components. First, five major families that owned the property rights of fishing field in Danayiku agreed to donate their traditional rights to the Shan-Mei village. Secondly, Danayiku was designated as a protected area managed by Shan-Mei, and any exploitation of wildlife and fresh water fish resources is banned in Danayiku, as long as it is not permitted by the management authority. Thirdly, anyone who violates the ban on hunting and fishing will be fined US$286 to 1429,[56] and a double fine will be imposed on community leaders who violate the ban. Fourthly, all male residents older than 18 and younger than 50 years old were obligated to patrol Danayiku by turns, in principle one patroller in daytime, and two at night (Li and Tang, 1999; Cheng-Sheng Gau, pers. comm.).

In November, 1989, about 13,000 kooye minnow, *Varicorhinus barbatulus* (*Pellegrin*), the traditionally most valued fish species by Tsou and most popular fish species found in Danayiku, were re-introduced in Danayiku. The intensive manpower input and strict enforcement of ban worked. Poaching still lasted, but decreased significantly. The kooye minnow population recovered dramatically. In only two years, the population increased to an estimated total number of 1.5 million in Danayiku (Cheng-Sheng Gau, pers. comm.). Meanwhile, villagers worked voluntarily for improving the infrastructure of

[55] The reason why a community-based conservation program was chosen had primarily to do with the personal experiences and ideas of community leaders. It is here omitted because it is beyond the scope of our discussion. For relevant details see Li and Tang (1999).

[56] In 2002, one US dollar exchanges approximately for 35 NT dollar.

the village and of Danayiku. To more effectively handle the increased management issues, the Shan-Mei Community Development Society (SMCDS) was founded in June, 1993. SMCDS are authorized to execute the conservation program in Danayiku, and also responsible for tourism and community development. Most adults of villagers are members of SMCDS, and have the right to participate in the process of decision-making (Cheung-Mei Yang, pers. comm.).

In August, 1994, Danayiku was opened to develop sport fishing. Anglers were charged for angling in Danayiku. At the same time, SMCDS succeeded in persuading the owners of about 200 ha. private lands to abandon their farms and bamboo plantation in Danayiku, and leave the lands recovering from human interference (Cheng-Sheng Gau, pers. comm.). In January, 1995, SMCDS declared the opening of the Danayiku Nature Park (DNYKNP). The park includes the area that extends 3 KM on both sides of the 18 KM long Danayiku river, and a section of 6 KM long riverside area along the Zeng-Wen river. Thereafter, tourists were allowed to enter into Danayiku after an entrance fee was charged. The main tourist attraction is the spectacular kooye minnow population which can hardly be found in wilderness of Taiwan. The revenues derived from sport fishing and entrance fee are used to pay the management costs, mainly staff salaries, and investment costs in physical capital. If there is a surplus, those money will be distributed to finance welfare works, including old-age pension, grants for students and subsidies for marriage, birth and death. Part of revenues was also spent to promote preservation and development of the culture of the Tsou (Wen, 2000; Cheung-Mei Yang, pers. comm.).

The success of the conservation program in DNYKNP soon attracted public attentions. Numerous official awards and reports of public media made it much more famous than in the beginning years. Both governmental donations and tourists began to pour in. Since 1998, the number of tourists increased dramatically, and this significantly improved the financial foundation of SMCDS (see section 9.3). Nowadays, DNYKNP is generally recognized as the most successful community-based conservation project in Taiwan.

9.3 Resource use

Based on the historical experiences, the Tsou people of Shan-Mei believe that sustainable use of renewable resources is reasonable by nature and should be encouraged. Accordingly, all interviewers speak bluntly that, rather than conservation motive, it was economic motive that drove Shan-Mei to initiate conservation project aimed at the sustainable use of renewable resources. The target species that is protected and harvested is kooye minnow, $Varicorhinus$ $barbatulus$ ($Pellegrin$). Kooye minnow, which is called ´Luska aku´ in Tsou meaning ´real fish´, is highly valued by the Tsou people, and was one of the main protein origins of the Tsou. It is also one of the most important game species of fresh water sport fishing in Taiwan. The market prices of kooye minnow are extremely high, varying from US$22.9 to 38.6 per kilogram (Cheng-Sheng Gau; Cheng-An Chuang, pers. comm.).

Three kinds of consumptive use of kooye minnow are allowed in DNYKNP. First, the section of 6 KM long riverside area along the Zeng-Wen river is designated as the sport fishing zone. The fishing season begins in November and ends in May. Anglers have to pay a charge between US$1.43 to 2.86 per hour, depending on number of anglers. Sport fishing is currently the most important form of consumptive resource use. In addition, infant fishes were sometimes caught for sale to aquatic breeder. Usually once each year, adult fishes will be caught, and then distributed to elders of the village. But these two forms of use play a so minor role that they almost can be neglected.

Fish viewing has been the primary non-consumptive use form of kooye minnow since the opening of DNYKNP. In fact, it is by far more important than any of the consumptive uses because it generates currently more than 90% of the total revenues of DNYKNP. In DNYKNP, about 20 ha. land are designated as the fish viewing zone and opened to tourists. Tourists currently have to pay an entrance fee of US$2.86 on weekend and holidays and of US$2.29 on weekdays (for child US$1.71 and 1.14, respectively). As a result of the dramatic increase in number of tourists in recent years that usually exceeded the carrying capacity of DNYKNP, an upper limit of daily tourist number might be introduced in the future.

9.4 Performance of the Danayiku Nature Park

Since 1989, villagers of Shan-Mei have actively protected natural resources in Danayiku, developed use strategies aimed at capturing both consumptive and non-consumptive values of resources, and devoted manpower and physical investment to improving management capacity. The case is therefore suitable for being analyzed under the theoretical framework developed in previous chapters. In the following discussion, we attempt to briefly describe the performance of DNYKNP in terms of fish population level, harvest quantity, physical capital stock, investment in physical capital, labor input, tourist number, and revenues of DNYKNP.

Figure 9.2 demonstrates the trend of kooye minnow population levels in DNYKNP in the last years. The original statistics about fish population are transformed by author into estimates of population biomass (kg), based on the premise suggested by Cheng-Sheng Gau (pers. comm.) that the mean weight of fish (including adult and infant fish) is 40 gram. After 13,000 fishes were re-introduced in 1989, the total population biomass reached a level of 60,000 kg in two years, and remained stable until 1996. In August, 1996, the typhoon Herb destroyed the whole A-Li-Shan area with a rainfall of more than 1,000 mm. at one night. DNYKNP lost about two thirds of its fish population as a result of the flood. Nonetheless, the fish population recovered rapidly thereafter. In recent two years, it increased to the level of slightly more than 60,000 kg, because fish population spilled from the core conservation area into the upstream area of DNYKNP (Cheng-Sheng Gau, pers. comm.). Several interviewers asserted that the current fish population has approached the level of natural state like several decades ago, but not yet totally recovered (Cheung-Mei Yang; Yue-Mei Chuang; Cheng-Sheng Gau; Cheng-An Chuang, pers. comm.). The reason why fish population does not totally recover can be attributed partly to the fact that fish population spilled from the core conservation area into the unprotected section of the Zeng-Wen river, and partly to the fact that poaching, controlled by insider of village, still lasted (anonymous interviewer; pers. comm.). In general, according to the population trend of fish, we can assert that DNYKNP has succeeded in protecting its target species. Moreover, previous statistics reveal that the intrinsic growth rate of kooye minnow is extremely high, although currently there is no relevant scientific assessment that can precisely measure the intrinsic growth rate of kooye minnow.

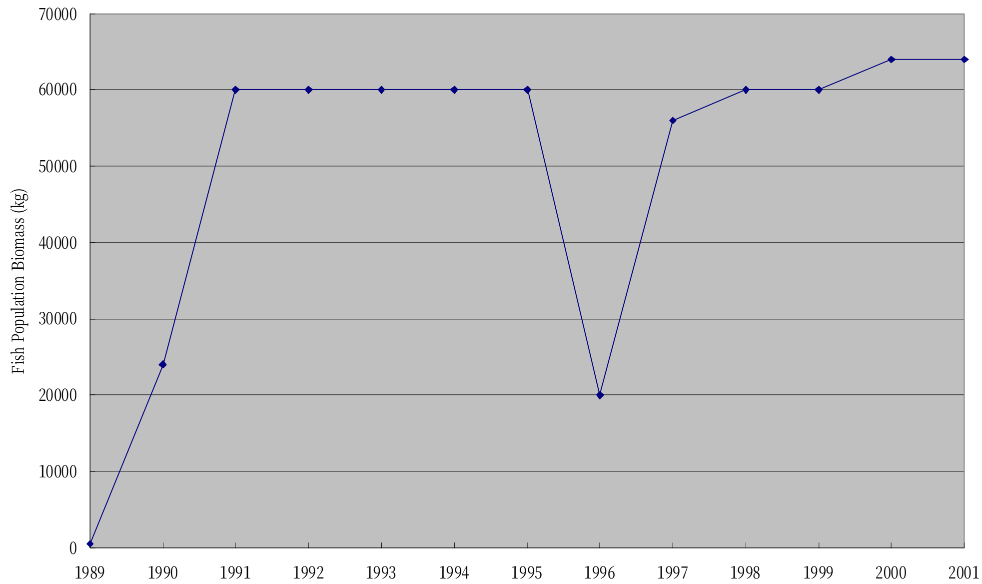


Figure 9.2 Fish population biomass of DNYKNP. Source: Cheng-Sheng Gau (pers. comm.) and personal calculation.

Figure 9.3 shows the harvest quantity of kooye minnow. The original statistics about revenues of sport fishing are transformed by author into estimates of harvest quantity, based on the assumption provided by SMCDS that fishing fee averages US$1.43 per hour and each angler has an average harvest of 600 gram each day (8 hours). Before 1994, all consumptive uses of kooye minnow were banned. After sport fishing and other forms of consumptive uses were allowed in 1994, the harvest quantity increased steadily. The trend was temporarily interrupted by the typhoon Herb. As a result of the serious loss caused by the typhoon, all consumptive uses were forced to be stopped in 1997. However, as fish population recovered, harvest increased again in recent years. In sum, the conclusion drawn from figure 9.2 and 9.3 is apparently consistent with that drawn from our theoretical models, that harvest and resource stock correlate positively to each other.

Based on the fact that the kooye minnow population steadily remains at a high, relatively stable level, and that the annually harvested quantity constitutes only less than 2% of the total population biomass so that fish population can easily recover from the effects of harvest, it can be asserted that the current use form of kooye minnow is sustainable.

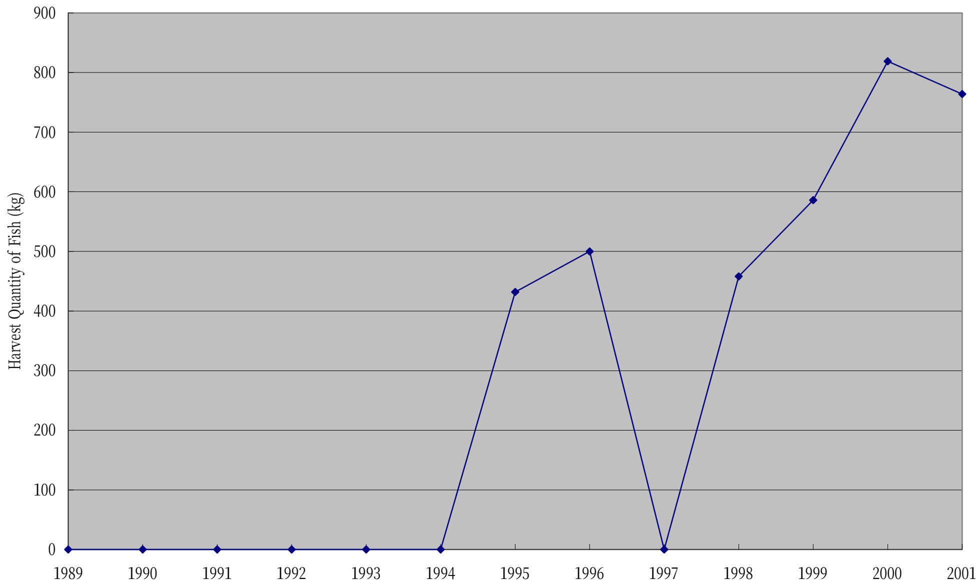


Figure 9.3 Harvest quantity of fish of DNYKNP. Source: SMCDS (1994-1999; 2001), Cheng-Sheng Gau (pers. comm.) and personal calculation.

Figure 9.4 describes the development trend of the physical capital stock of DNYKNP. These statistics are calculated by the author, based on the annual investment in physical capital provided by SMCDS and on the assumption that the depreciation rate of physical capital stock equals 0.1 in ordinary years, and equals 0.5 in the year when the typhoon caused great capital loss. In general, the physical capital stock increased steadily since 1993, except that there was a significant decrease in 1996 caused by the typhoon Herb. Figure 9.5 shows the trend of the annual investment in physical capital. The investment reached its top in 1994 when DNYKNP was prepared for opening for the public. Investment in 1997 increased significantly in response to the great loss of capital stock occurred in 1996. The significant increase in investment in 1999 reflects the fact that management capacity must be improved in response to the dramatic increase in the number of tourists since 1999 (see figure 9.7). Thereafter, investment decreased gradually again. In sum, the long-term trend of investment seems to be declining while the total capital stock increased. Hence, the development trends of physical capital stock and investment seems to be consistent with those drawn from our theoretical models, that investment and management capital stock correlate negatively to each other. But a definite answer to this question still needs long-term observation in the future.

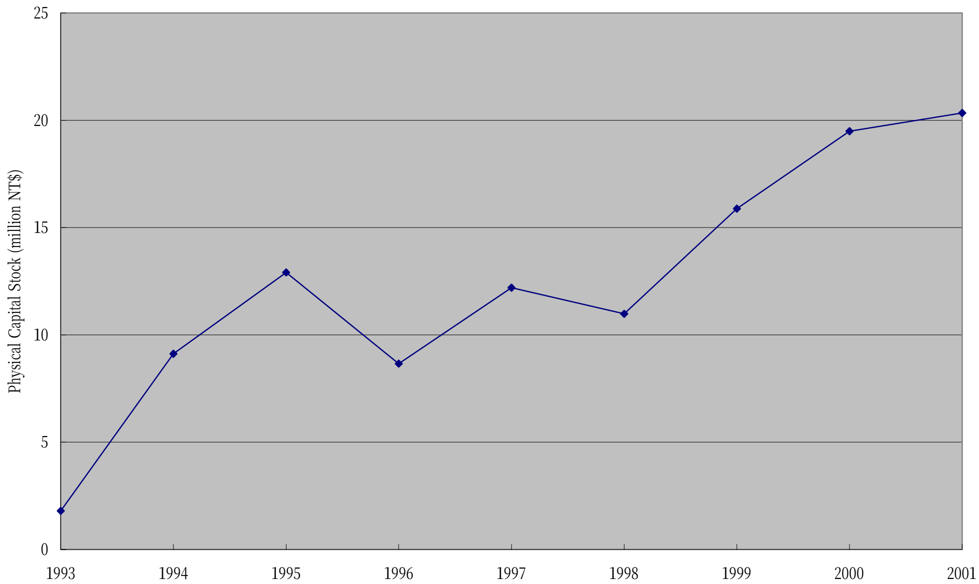


Figure 9.4 Physical capital stock of DNYKNP. Source: SMCDS (1994-1999; 2001), Cheng-Sheng Gau (pers. comm.) and personal calculation.

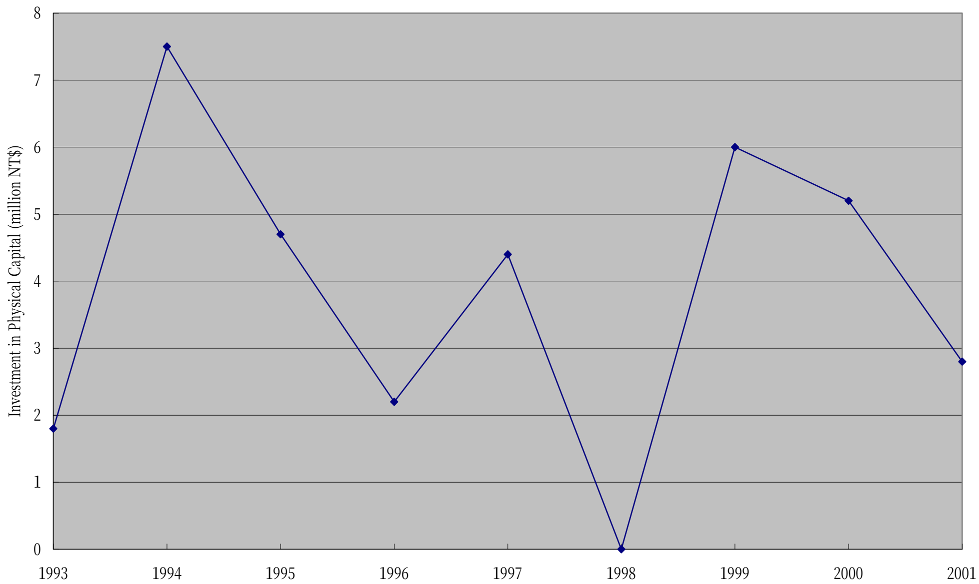


Figure 9.5 Investment in physical capital of DNYKNP. Source: SMCDS (1994-1999; 2001) and Cheng-Sheng Gau (pers. comm.).

As discussed in section 7.2 about management capital, it is extremely difficult to measure the human and institutional components of management capital stock. What we can do is simply calculating the manpower input as an indicator for the investment in human and institutional capital. Figure 9.6 demonstrates an increasing trend of the manpower input of the conservation project that formally paid by SMCDS. However, these statistics do not reflect the fact that many voluntary manpower were devoted to the conservation project in the beginning stage before 1994, because even rough statistics about voluntary manpower input are not available.[57] In fact, since the opening of DNYKNP in 1994, all manpower inputs in DNYKNP were paid by SMCDS, and thereby voluntary works were virtually almost totally transformed into formal works. It follows that it is difficult to judge the long-term trend of the investment in human and institutional capital without credible data resources.

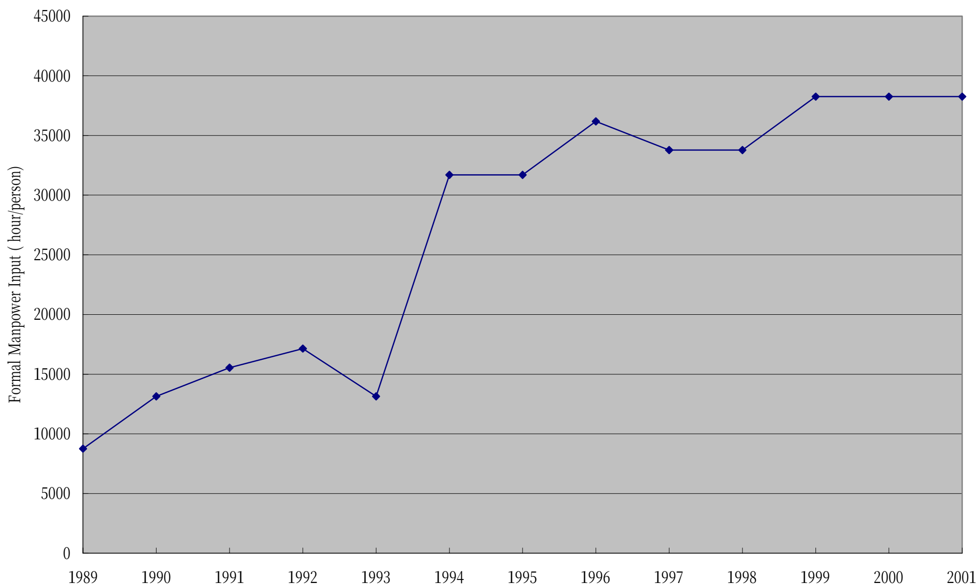


Figure 9.6 Formal manpower input of DNYKNP. Source: Cheung-Mei Yang (pers. comm.) and personal calculation.

The reputation of DNYKNP as a successful conservation project and the famous, spectacular fish population attracted numerous tourists in recent

[57]For example, villagers remember that there were often more than ten persons that voluntarily patrolled DNYKNP at one night in the beginning stage. Meanwhile, intensive voluntary manpower was devoted to improving infrastructure, and villagers spent much time in meeting and other organizational issues. Currently, no more voluntary workers are devoted to patrolling, and those voluntary manpower devoted to infrastructure improvement and organizational issues have significantly decreased.

years. Figure 9.7 shows that the tourist number dramatically increased since 1999. This also brought DNYKNP significant increase in revenues since 1999, as shown by figure 9.8.[58]

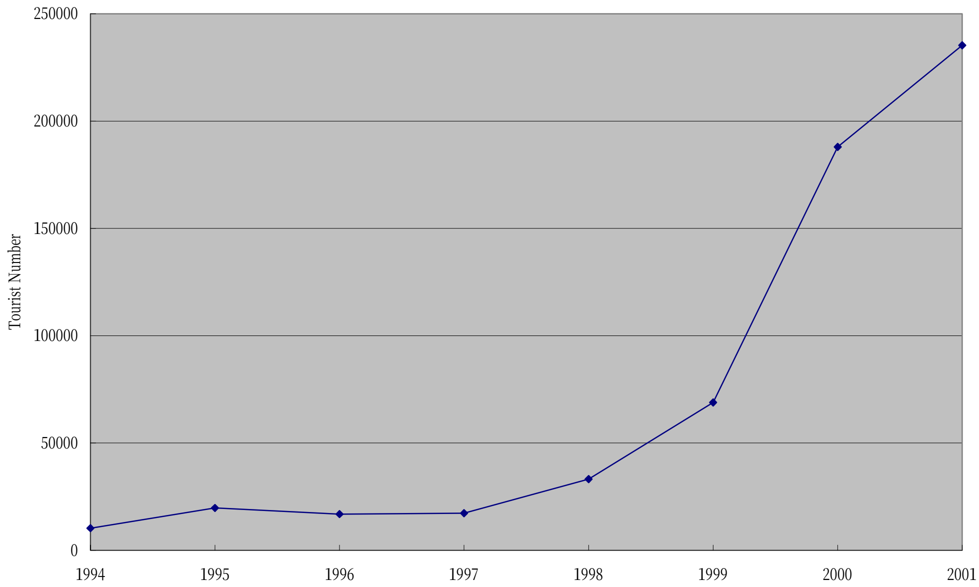


Figure 9.7 Tourist number of DNYKNP. Source: SMCDS (1994-1999; 2001) and Cheng-Sheng Gau (pers. comm.).

[58]The statistics demonstrated by figure 9.7, figure 9.8 and table 9.1 are calculated on the basis of the fiscal year. For example, the fiscal year 1999 began in July, 1998 and ended in June, 1999. However, in response to the modified accounting system, the fiscal year 2000 included 18 months, namely began in July, 1999 and ended in December, 2000. Since 2000, the fiscal year begins in January and ends in December.

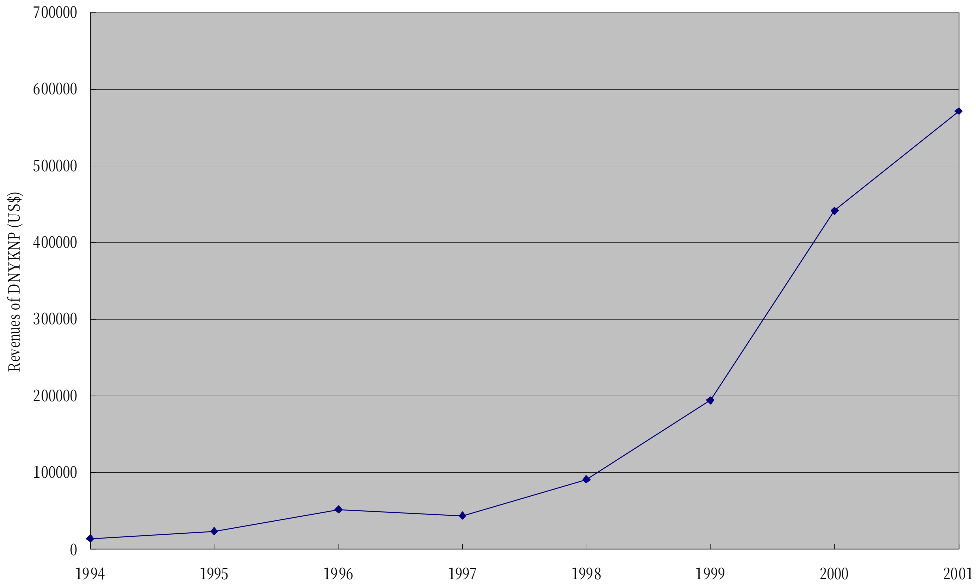


Figure 9.8 Revenues of DNYKNP. Source: SMCDS (1994-1999; 2001), Cheng-Sheng Gau (pers. comm.) and personal calculation.

Table 9.1 analyses the financial performance of SMCDS in fiscal year 2000. The main sources of revenues were entrance and parking fee of tourists, which made up of 90.86% of the total revenues of DNYKNP. In fact, part of the item ´other revenues´ were contributed by fish viewing tourism. Hence, non-consumptive value undoubtedly played a much more important role than consumptive value which contributed only 2.67% to the total revenues of DNYKNP. Governmental donations used primarily for physical capital investment had played an important role in the beginning stage. Nonetheless, as a result of the dramatic increase in tourism revenues, the proportion of self-financing to donor funding has significantly increased in recent years, so that donor funding made up of only 13.92% of the total revenues of SMCDS in fiscal year 2000. These revenues were primarily (about 71% of total expenditures) spent on staff salaries, administration costs and physical capital investment, but also lots of money (about 29% of total expenditures) were spent for supporting social welfare and cultural and communal development. In sum, there was a slight deficit of US$25,229 in fiscal year 2000. If the expenditures on social welfare and cultural and communal development that are not necessary for maintaining conservation project, and the governmental donations were not taken into account, the conservation project itself would virtually generate a net surplus of US$59,258. According to the rough estimates of the fiscal year 2001, with a total revenue of more than

US$572,000 from DNYKNP and a decreased physical capital investment of US$80,000, the financial surplus of the conservation project itself will undoubtedly approach to the level of about US$286,000. Given the fact that physical capital investment is decreasing after infrastructure has been greatly improved, and staff salaries remains relative stable, it follows that, as long as the tourist number remains stable at the level of the year 2000, or even slightly less than that of the year 2000, the financial self-sustainability of the conservation project will be guaranteed. In fact, to maintain financial self-sustainability, SMCDS even doesn´t need to attract so many tourists like the year 2000, because the expenditures on salaries, administration and capital investment will also, though not proportionally, decline when tourist number decreases. In any case, the financial self-sustainability of SMCDS can almost totally be attributed to the high non-consumptive value of kooye minnow population.

Since 1989, the intensive patrolling has effectively brought poaching under control. Outside poachers were scarcely detected (Cheung-Mei Yang, pers. comm.), but inside poaching, as anonymous interviewer reported, still exists. The quantity of the illegally harvested fish biomass is unknown.

Table 9.1 Financial performance of SMCDS in fiscal year 2000

	Net revenues/expenditure in US$
REVENUES OF DNYKNP	441,637
Entrance fee	375,907
Sport fishing fee	11,782
Parking fee	25,348
Other revenues	28,600
GOVERNMENTAL DONATIONS	71,429
EXPENDITURES	−389,724
Staff salaries	−127,034
Administration costs	−106,773
Social welfare	−55,205
Traditional dance team	−54,410
Other subsidies	−46,302
PHYSICAL CAPITAL INVESTMENT	−148,571
SURPLUS/DEFICIT	−25,229

Source: SMCDS (2001) and personal calculation.

9.5 Ecological, economic and social benefits

9.5.1 Ecological benefits

Since 1989, the whole ecosystem of Danayiku has benefited from the strict protection measures against poaching and other illegal activities. First, about 200 ha. private farms and bamboo plantation lands in Danayiku were returned to nature. The Danayiku river also returned to life. Although the protected target species is kooye minnow, no special measures are taken to discriminate other species. All endemic fresh water fish species of Danayiku have virtually shared the benefits of protection measures, especially *Acrossocheilus paradoxus Gunther* and *Zacco barbata*. The populations of these two species have significantly increased in recent years. Similarly, the national forest covering Danayiku valley are more effectively protected. Illegal logging that usually occurred in the last several decades ceased at last. The once disappeared wildlife gradually returns to Danayiku valley from the nearby Yushan National Park. Currently, presence of many big mammalian species are often reported, including Formosan white-faced flying squirrel, Formosan giant flying squirrel, Formosan macaque, Formosan black bear, Formosan wild boar, Formosan Reeve´s muntjac, Formosan sambar, Formosan serow and Chinese leopard cat (Cheng-Sheng Gau, pers. comm.).

9.5.2 Economic benefits

In the following we will briefly describe the economic benefits of DNYKNP in terms of income creation, full time and part time job creation, and dividends to the community. The success of DNYKNP brings Shan-Mei huge amount of income. For example, together with the governmental donations, the whole conservation program created a cash inflow of approximately US$524,286 for SMCDS in the fiscal year 2000. About half of the US$524,286 were directly transformed into the income of villagers through the ways of staff salaries, social welfare expenditures, pay for the traditional dance team and other subsidies. Part of the other half of the cash inflow which were expended for administration costs and capital investment also raised the income level when villagers contracted, or are employed to participate in these works. The revenues of SMCDS directly created some full time and part time job, currently including 12 full time employees, 4 full time rangers, 4 part time rangers, 30 part time dancers and some temporary workers. It is worth noting that, except some jobs at manager level, most of the employees and rangers of SMCDS have terms of office, usually two years. Villagers take turns at working for SMCDS. This is an equitable solution for those who don´t have any job.

For most of the villagers, what more important are the added values brought by tourism. As a result of the remote location of Shan-Mei, most tourists have to find accommodation or at least have a meal during stay in Shan-Mei. They usually also buy souvenir and need certain kinds of services. According to the experiences of villagers, an average added value of at least one dollar will be brought by per one dollar revenue of DNYKNP. It implies that, with the total revenues of more than US$571,429 in 2001, DNYKNP generated an added value of at least US$571,429 for the whole village in one year. Though precise figures are not available, the total financial benefit of DNYKNP is undoubtedly huge for the small village with less than 600 population, especially from the perspective of indigenous community with a relative low standard of living.

To date, two hotels in Shan-Mei operate at their full capacity with total 130 beds.[59] They directly create about 10 full time jobs and some part time jobs, depending on business, and indirectly create some job opportunities for farmers who supply agricultural products to them. More than 50 villagers work in snack bars, restaurants and souvenir shops. Most of them work part time, namely only on weekends and holidays when tourists pour in. But even these part time jobs earn them a higher income than any jobs they can find elsewhere, according to personal survey of the author. 5 tour guides operate interpretation tour, charging a fee of US$22.9-28.6 per tour. Some skilled hunters recently began to operate sporting hunting for tourists in the area around Danayiku valley, aiming at the hunting of Formosan wild boar, Formosan white-faced flying squirrel and Formosan giant flying squirrel. In general, except a few teachers and government officers, almost all villagers directly or indirectly depend on DNYKNP for their living (Chia-Ping Chuang; Cheng-Sheng Gau, pers. comm.). The most significant sign of the huge economic benefits is that population of Shan-Mei, especially the younger generation, begins to return to and resettle in the village. This is a phenomenon that hardly can be found in another indigenous communities throughout Taiwan.[60]

According to the resolution of SMCDS, some revenues of SMCDS will be made available for communities, used primarily for social welfare and communal development. In this way, the revenues arising from conservation

[59] As a result of the high occupancy rate of these two hotels on holidays, four hotels outside Shan-Mei with total 800 beds currently depend primarily on tourists visiting Danayiku for doing their business.

[60] Even many Chinese people wanted to do business in Shan-Mei, but were refused by SMCDS (Cheung-Mei Yang; Cheng-Sheng Gau, pers. comm.).

project can therefore be more equally distributed amongst villagers. This, together with other economic benefits, have contributed to some of the social benefits that have been obtained because of the conservation project.

9.5.3 Social benefits

The social components of the conservation project can be judged according to the following criteria, including health, education, institution, infrastructure, excessive drinking, family problems, social security, and culture preservation and development. The higher income level may have positive influence on health conditions and formal education level of villagers, but the effect is presumably negligible in the short run. Currently, no credible data sources are available. One interesting phenomenon is that some young people organized a small group to educate themselves through reading and discussion. From the perspective of institution, the conservation project provided chances for villagers to participate in decision-making process of public affairs. The community is developing management, marketing and business skills, and thereby improving its management capacity. Through steady investment, the infrastructure of the community are significantly improved. Unemployment dramatically declined, and this led to the decrease in excessive drinking and in other family problems (Cheung-Mei Yang, pers. comm.). The financial support of SMCDS enhanced an initial social security system. It also promoted preservation and development of the Tsou culture, and reconstructed the self-confidence of the Tsou people of Shan-Mei in their own cultural tradition (Wen, 2000).

9.6 Negative impacts

Except positive influences, the conservation program also has negative impacts on ecological and human system. From the ecological point of view, excessive tourism, especially on holidays, constitutes the most serious threat that gradually degrades the original ecosystem of DNYKNP. All interviewers agree that the current tourist number on holidays has far exceeded the carrying capacity of DNYKNP, and it keeps growing. This will reduce the tourism potential, and injure the reputation of DNYKNP as a conservation model. Furthermore, in pursuit of economic benefits brought by tourists, some land owners in Danayiku began to do their business outside the zone where is designated as legal business area. This will further cause more damage to ecosystem and tourism potential. However, no adequate measures are taken to stop the trend. Fortunately, all these negative impacts are restricted to the area of about 20 ha. that is opened for tourism. Therefore, rather than ecological health, what people really concerned about is the damage to tourism potential, because, once tourism potential was significantly reduced, the financial foundation of SMCDS, and in turn the whole conservation project would be threatened with collapse.

Another issue with reference to ecological health is the problem of fish feeding. To attract tourists, fish feeding was allowed at three specific sites since the opening of DNYKNP. Some ecologists were seriously concerned about this problem. Recently, considering the possible negative effects, all fish feeding was ceased.

With reference to the negative economic and social impacts, what people most concerned about is the problem of wealth distribution. The introduction of tourism and market mechanism led to a greater gap of wealth between rich and poor families than a decade ago. Those who are quickly accustomed to free market and learned modern business skill got a greater proportion of the economic benefits created by the conservation program. Furthermore, people began to complain that, in pursuit of economic benefits, villagers gradually became selfish and utilitarian. Today, most villagers is unwilling to work voluntarily for the community without any pay (Chia-Ping Chuang, pers. comm.). In the long run, whether these phenomena will influence the identity of villagers, and in turn influence the performance of the conservation program, remains to be seen.

9.7 Comparison of different community-based conservation projects in the A-Li-Shan area: an assessment procedure

9.7.1 The assessment procedure

The success of DNYKNP at Shan-Mei has inspired several indigenous communities in the A-Li-Shan area, including Hsing-Mei, Li-Chia, Da-Ban and Cha-Shan (see figure 9.1), to initiate similar conservation projects in 1990s aimed at the sustainable use of fresh water fish resources. In addition, the Shan-Ming Township located in the Nan-Zi-Shen river basin (see figure 9.1) independently initiated its fresh water fish conservation program in 1990. To date, only DNYKNP at Shan-Mei can be ranked as a successful conservation program. Hsing-Mei and Li-Chia are advancing in protecting their fresh water fish resources, but still have to struggle for the maintenance of their conservation programs whose fate remains to be seen. The conservation programs in Da-Ban, Cha-Shan and Shan-Ming have apparently failed. Based on these facts, some questions emerge: Which factors affect the feasibility of the sustainable use strategy as a conservation instrument? Is there a general framework that can explain, partly at least, why some community-based conservation programs succeeded while some other failed?

Prescott-Allen and Prescott-Allen (1996) suggested an systematic assessment procedure that evaluates the sustainability of uses of wild species. However, this procedure can be used only after the use program of wild species has been brought into practice. It is therefore an *ex post* assessment procedure. Instead of an *ex post* assessment procedure, an *ex ante* assessment framework will be developed in this section to assess, or even forecast the feasibility of a sustainable use strategy before it is brought into practice. In the following discussion we will demonstrate how this systematic procedure works.

As discussed in chapter 8, we have identified eight factors that influence the equilibrium resource stock of a sustainable use program, including the non-consumptive value coefficient (β), the efficiency coefficient of management capital (γ), the intrinsic growth rate of utilized species (ρ), the gross profit coefficient of harvest (τ), the discount rate (r), the depreciation rate of management capital (δ), the poaching coefficient (α) and the cost coefficient of investment (σ). We have also demonstrated that, other things being equal, the equilibrium resource stock will increase, when β, γ, ρ and τ are increased, and when r, δ, α and σ are decreased. In general, the more the equilibrium resource stock closes to the carrying capacity, the better it would be for ecosystem health, and, from the ecological point of view, the use program would be more likely to be sustainable. Accordingly, we can use the equilibrium resource stock, and in turn the eight factors as indicators for judging the feasibility of the sustainable use strategy as a conservation instrument.

In reality, it is impossible for almost all practical cases to precisely estimate the size of the eight parameters without extensive scientific research. We also don´t know the precise functional forms of relevant functions and the precise comparative static effects of the eight parameters on equilibrium resource stock. Since information is never complete, therefore, instead of executing a computer simulation, what we practically can do may be giving them a rough ranking based on the findings of field study. An equal weight is then given to the eight parameters by assuming that they have equal influence on equilibrium resource stock.[61] The formal procedure is as follows. The eight parameters are first classified into five different ranking according to their size, including ´very high´, ´high´, ´middle´, ´low´ and ´very low´. For the parameters β, γ, ρ and τ, we define ´very high´ as ´highly positive´

[61] In fact, some parameters may play a more important role than others do. For example, according to historical experiences, the intrinsic growth rate of species may have greater influence on equilibrium resource sock than some of the other factors.

in the sense that a very high β, γ, ρ or τ has a highly positive influence on the equilibrium resource stock, and thereby can highly contribute to sustainable use of resources. Similarly, ´very low´ is defined as ´highly negative´ in the sense that a very low β, γ, ρ or τ has a highly negative influence on the equilibrium resource stock, and thereby can seriously threaten the sustainable use of resources. The ranking ´high´, ´middle´ and ´low´ are defined as ´positive´, ´neutral´ and ´negative´, respectively. A numerical ranking 1, 2, 3, 4 and 5 is then given to the evaluations ´highly positive´, ´positive´, ´neutral´, ´negative´ and ´highly negative´, respectively. The smaller the ranking of a parameter is, the more positive influence it would have for equilibrium resource stock and sustainable use of resources. Contrarily, for the parameters r, δ, α and σ, we define ´very high´ as ´highly negative´ in the sense that a very high r, δ, α or σ has a highly negative influence on the equilibrium resource stock, and thereby can seriously threaten the sustainable use of resources. The ranking ´high´, ´middle´, ´low´ and ´very low´ are hence defined as ´negative´, ´neutral´, ´positive´ and ´highly positive´, respectively. The numerical ranking from 1 to 5 can be then given to the evaluations from ´highly positive´ to ´highly negative´.

After all ranking about the eight parameters of a specific case are given, an arithmetical average of these ranking can be obtained. The final score represents the overall status of a sustainable use project. The smaller the score is, the greater the equilibrium resource stock would be, and the more feasible a sustainable use strategy would be as a conservation instrument. In the following we will apply this assessment procedure for assessing the feasibility of different sustainable use programs in the A-Li-Shan area, Taiwan, as a conservation strategy.

9.7.2 A comparison of different community-based conservation projects in the A-Li-Shan area

Table 9.2 demonstrates the results of the assessment procedure with regard to the six community-based conservation programs, including Shan-Mei, Hsing-Mei, Li-Chia, Da-Ban, Cha-Shan and Shan-Ming. The ranking are based on the findings of the field study executed during 1999 and 2002. Now let´s explain the reasons underlying these ranking.

The target species of all the six conservation programs are fresh water fishes, especially kooye minnow. As the experience of Shan-Mei showed, kooye minnow has a very high intrinsic growth rate (ρ). Therefore, all the ranking of the six conservation programs are 1 (highly positive) in the column ´ρ´. Next, as a result of the extremely high market prices of kooye minnow, the gross profit coefficient of harvest (τ) of all cases are extremely high. According to our survey, there is no significant difference in the expensive sport fishing fee charged by these villages. It follows that all the ranking of the six conservation programs are also 1 (highly positive) in the column ´τ´. In addition, all six conservation programs are threatened with highly active poaching. Most of poachers are insiders of villages and, without continuous protection actions against poaching, the fresh water fish resources would be rapidly depleted to the level of local extinction (Cheng-Sheng Gau, pers. comm.). A ranking of 5 (highly negative) is hence given to all six villages in the column ´α´. Finally, the six villages have similar economic conditions. Compared to ordinary Chinese communities in Taiwan, their income level and standard of living are relatively low. Getting into debt is a prevailing phenomenon among them. The villagers have therefore relatively strong incentive to cash in natural resources. But most of them are not really so poor that they would deplete natural resources irrespective of the possible negative impacts. Based on these observations, we believe that a ranking of 4 (negative) for the discount rate (r) will be reasonable. With reference to the depreciation rate of management capital (δ), no information or facts can be used to assess its status. Hence, it is here excluded from the assessment procedure.

Table 9.2 Results of the assessment procedure

	ρ	τ	α	r	δ	β	γ	σ	Average ranking
Shan-Mei	1	1	5	4	*	1	1	2	2.14
Hsing-Mei	1	1	5	4	*	5	4	2	3.14
Li-Chia	1	1	5	4	*	5	4	2	3.14
Da-Ban	1	1	5	4	*	5	4	3	3.29
Cha-Shan	1	1	5	4	*	5	4	4	3.43
Shan-Ming	1	1	5	4	*	5	5	4	3.57

1: highly positive. 2: positive. 3: neutral. 4: negative
5: highly negative.
*unknown

There are significant difference between the non-consumptive value coefficient (β) of different villages. Among these villages, Danayiku of Shan-Mei originally owns the most abundant fish population (Cheng-Sheng Gau, pers. comm.). The beautiful scenery of Danayiku makes it an attractive tourist destination and significantly promotes the non-consumptive value of kooye minnow. Shan-Mei is therefore given a ranking of 1 (highly positive) in this item. Hsing-Mei, Da-Ban, Cha-Shan and Shan-Ming don´t have the spectacular fish population and outstanding scenery like Shan-Mei does. These lead to the very low non-consumptive value of kooye minnow. Li-Chia has nearly similar natural conditions like Shan-Mei, but its remote location offsets its natural advantages.[62] The five villages are therefore given a ranking of 5 (highly negative) in this item.

Natural topography and human road system lead to significant difference between the efficiency coefficient of management capital (γ) of different conservation programs. In the case of Shan-Mei, there is only one road to the entrance of DNYKNP and nowhere in DNYKNP is accessible by car to poachers. Once the only entrance is controlled, the whole DNYKNP is in principle safeguarded. In other words, the efficiency coefficient of management capital of Shan-Mei is very high and should be ranked as 1 (highly positive). In Hsing-Mei, Li-Chia, Da-Ban and Cha-Shan, the river is at many sites easy accessible by car or by walking as a result of the relative complex road system. This leads to a relative low efficiency coefficient of management capital. The four villages can be ranked as 4 (negative) in this item. The worst situation occurs in Shan-Ming. A highway was constructed along the river, and virtually everywhere is easy accessible by car. This is why it is ranked as 5 (highly negative) in the column ´γ´.

About the cost coefficient of investment (σ), there is no significant difference in the investment cost of physical and human capital between different villages. The major difference exists in the investment cost of institutional capital as a result of the different combinations of tribes. For example, almost all the villagers of Shan-Mei, Hsing-Mei and Li-Chia are the Tsou people. The homogeneity of villagers greatly reduces the communication costs of building management institution. The low investment cost in institution capital leads to the ranking of 2 (positive). Contrarily, Cha-Shan and Shan-Ming primarily consist of Tsou, Bunun[63] and Chinese people (Li and Tang, 1999). The complex population combinations make it very difficult to reach consensus when discussing public affairs (Li and Tang, 1999), and significantly raise the communication costs of building management institution. We therefore rank them as 4 (negative) in this item. The village Da-Ban primarily consists of the Tsou people that stem from two different communities that are historically often hostile to each other (Li and Tang, 1999). Its communication costs are probably higher than Shan-Mei, but lower than Cha-Shan and Shan-Ming. It is classified into the rank 3 (neutral).

[62] Shan-Mei is about one car hour distant from Chia-Yi, the most important city in this area. Compared to Shan-Mei, tourists have to drive another two hours to reach Li-Chia. Almost no one will do that to see what they can see in Shan-Mei.

[63] Bunun is also one of the ten officially recognized indigenous tribes in Taiwan.

Based on previous discussions, an average ranking of each village can be calculated. In sum, Shan-Mei owns many positive, but only two negative exogenous conditions. This helps it obtain the lowest average ranking of 2.14 (approximately positive) among all villages, which implies that the biological, economic and social conditions are generally adequate to the application of the sustainable use strategy. Compared to Shan-Mei, Hsing-Mei and Li-Chia have lower non-consumptive value of resources and lower efficiency of management capital. They both get an average ranking of 3.14 (approximately neutral), which is not particularly in favor of or against a sustainable use strategy. Compared to Hsing-Mei and Li-Chia, Da-Ban and Cha-Shan have a higher cost coefficient of investment. They obtain an average ranking of 3.29 and 3.43, respectively, that are probably somewhat disadvantageous to sustainable resource use. Finally, under the existing conditions of low efficiency of management capital and high cost coefficient of investment, Shan-Ming obtain the highest average ranking of 3.57.

The results of the assessment procedure imply that, without intensive outside (governmental or non-governmental) support that can offset some disadvantageous exogenous conditions, the use strategy would probably fail in the sense that it cannot maintain a relatively high resource stock level, and would not be feasible as a conservation instrument in the cases of Da-Ban, Cha-Shan and Shan-Ming. For Hsing-Mei and Li-Chia, the exogenous conditions are not especially good or bad. Only Shan-Mei owns the biological, economic and social conditions that are generally adequate to the application of the sustainable use strategy. These conclusions are surprisingly consistent with what really happened in the past years. Only Shan-Mei has succeeded in maintaining a high resource stock level that approaches the carrying capacity. Hsing-Mei and Li-Chia are advancing in protecting their fish resources, but the results are not especially good. Their fish population levels are not high enough so that sport fishing is allowed only in one week each year. As a result of the scarcity of revenues caused by the relatively low population level, they still have to struggle for the maintenance of their conservation programs whose fate remains to be seen. The conservation programs of Da-Ban, Cha-Shan and Shan-Ming did not last for a long time and have soon failed.

Some critical conclusions can be drawn from the findings of the case study. First, sustainable use of wild species is not omnipotent recipe for resolving conservation problems (and development problems). As our case study shows, even under the same conditions of intrinsic growth rate of species, consumptive value of species, discount rate and poaching pressure, use strategy worked well in few cases. It doesn´t work in other cases , because a variety of exogenous factors also can influence the outcomes of conservation programs. It follows that it is highly dangerous to judge the feasibility of a sustainable use program by using only one or only few indicators, including the intrinsic growth rate of species, the discount rate and the price/cost ratio of harvest, as some people were used to do in the past.[64] To evaluate the feasibility of the sustainable use of wild species as a conservation strategy, it always needs an overall assessment about relevant biological, economic and social conditions at the site concerned.

Secondly, in the previous cases at least, the results of our assessment procedure are in principle consistent with what happened in the reality. We therefore argue that the assessment procedure can be applied for sustainable use programs before they are brought into practice. For those cases with low or very low average ranking, sustainable use strategy can be encouraged. For those cases with high average ranking, more caution should be taken, or they simply should be stopped. This may help reduce the probability of resource overexploitation and environmental degradation caused by the failure of sustainable use projects.

Thirdly, at the level of economic theories, both the Clark model and the Swanson model demonstrated the important roles played by the three factors when discussing resource use problems and relevant conservation issues, including the intrinsic growth rate of species, the discount rate and the price/cost ratio of harvest (the gross profit coefficient of harvest in our terminology). However, our model and the case study show that, rather than the previous three often discussed factors, they are non-consumptive value of resources, efficiency of management capital and investment cost in management capital that explain the difference of the performance of various use programs in this case. Hence, these newly introduced factors also play an important role, because they sometimes can determine the fate of the sustainable use project. We conclude that it is worth while extending bioeconomic models to investigate problems of management capital accumulation, non-consumptive value of wild species and illegal harvest. Such modifications help us have more insight into the complex resource use problems and relevant conservation issues.

Finally, our case study shows that, under prevailing natural, economic and social conditions in Taiwan, the probability of success of a sustainable

[64] For example, Caughley argued that the consumptive use of African elephant is not a feasible conservation strategy because of the low intrinsic growth rate (Caughley, 1993).

use program initiated by indigenous communities is relatively low, if villages in the A-Li-Shan area can represent the ordinary indigenous communities in Taiwan. According to author´s personal observation, there are only few indigenous communities that have general conditions that can obtain the ranking like Shan-Mei has. At the policy level, this implies that some popular thoughts about resource use, conservation and indigenous people should be reconsidered. In recent years, more and more people argue that, from both perspectives of human rights and conservation, the property rights of natural resources in some protected areas should be returned to indigenous communities, and decentralized, community-based conservation programs would work better than traditional protected areas managed by the central government. From the perspective of human rights, whether the property rights of natural resources should be returned to indigenous communities is a question of value judge and beyond the scope of our discussion. But from the perspective of conservation, we assert that the performance of community-based conservation programs is in general not so good as it is supposed to be. Compared to the National Park system and other nature reserves managed by the central government, the success probability of community-based conservation programs is relatively low so that they can protect only small fragments of the whole ecosystem. Based on the case study in chapter 4 about the National Park system of Taiwan, it is clear that, under current circumstances, only the central government has the capacity to support and manage a large, systematic protected area network in Taiwan. It is unreasonable to suppose that indigenous communities can deal with the problems like poaching, institution building and financial deficit more effectively than the central government.

9.8 Some challenges to DNYKNP at Shan-Mei

Some problems still threaten the long-term sustainability of DNYKNP. First, the majority of DNYKNP is national forest land owned de jure by the central government, although villagers of Shan-Mei have de facto the use rights of some natural resources, such as fresh water fish, since a long time. The ambiguity of property rights of natural resources has troubled Shan-Mei, since all charges in DNYKNP is virtually illegal, and, once the friendly attitude of the government toward DNYKNP changed some day, the project would probably fail if charges were not allowed. Secondly, the current mass tourism would significantly reduce the tourism potential of DNYKNP, if no adequate measures are taken to control the number of tourists. Finally, as a result of the huge economic benefits brought by tourism, SMCDS seemed to have gradually lost the control of individual business behavior of villagers in DNYKNP. Most villagers began to worry about that excessive business activities will degrade the natural environment of DNYKNP, and in turn damage its tourism potential and its good reputation as a conservation model. Whether Shan-Mei can overcome these challenges in the future remains to be seen.

Chapter 10
Conclusions, policy implications and limits in applicability of the theoretic model

In recent years, the sustainable use of wild species in protected areas or in buffer zones of protected areas is usually promoted as an alternative conservation strategy. This dissertation has focused on the relevant resource harvest and management issues. What this research especially concerned about is the question, whether and under which biological and socio-economic conditions the sustainable use of wild species is an adequate strategy for biodiversity conservation. To provide a general theoretical framework for answering this question, several models were developed which investigate the dynamic interaction between the harvest of wild species, management of protected areas, population levels of the utilized species and illegal harvest. The case study involving the community-based conservation projects in the A-Li-Shan area of Taiwan was addressed to test the applicability of the theoretic models. This final chapter offers study conclusions, policy implications and some comments involving the limits in applicability of the theoretic model and recommendations for future research.

10.1 Study conclusions

10.1.1 Conclusions of the theoretic models

At the theoretic level, our models enriches the analytical framework of the traditional harvest model of renewable resources by developing a bioeconomic model with two state and two control variables. Compared to the Clark model which considers only the harvest problem, we additionally consider the problems of the resource management and illegal harvest. On the other hand, both the Swanson model and our models take the factor of the evolution of the management capacity into account. However, rather than also considering issues of land use competition, as Swanson did, we confine the models to addressing the related harvest and management issues of the sustainable use strategy applied in given protected areas, while the Swanson model did not handle the problem of illegal harvest. Furthermore, we provide a more deliberate modeling for relevant issues than the Swanson model by introducing a new state variable, namely the management capital, and thereby regarding the evolution of management capacity as a process of capital accumulation. This is the characteristic that differentiates our models from almost all other bioeconomic models which either treated the management factor as a flow variable, as Swanson did, or addressed the issue of accumulation of the capital utilized in harvesting renewable resources. In the following the results of the theoretic models are briefly summarized.

The uniqueness of the steady state solution of the simple model in chapter 6 and of the extended model in chapter 7 can be confirmed. Under specific assumptions, it can also be verified that the steady state solution of the simple model and of the extended model is saddle point stable, while in chapter 8 only the existence of the steady solution of the general model can be proved. A special case of the simple model which did not consider the non-consumptive value of resources possesses the properties of uniqueness and saddle point stability without using similar assumptions in the simple model.

The phase diagrams of the simple model can be derived, while those of the extended model, as generally recognized, cannot be directly obtained via analytical method as a result of the complex interaction between multiple state and control variables. By the application of the computer simulation, the phase diagrams of the general model were derived. The outcomes showed that, on the optimal dynamic path, the resource stock and the harvest rate will increase over time, if the initial resource stock is lower than the steady state resource stock level. On the other hand, if the initial resource stock is higher than the steady state resource stock level, the resource stock and the harvest rate will simultaneously decrease over time. In addition, on the optimal dynamic path, the management capital stock will increase while the investment rate will decrease over time, if the initial management capital stock is lower than the steady state management capital stock level. On the contrary, if the initial management capital stock is higher than the steady state stock level, the management capital stock will decrease while the investment rate will increase over time.

The poaching (illegally harvested quantity of resources) depends on both the resource stock and management capital stock level. Thus, unlike the clear interaction between the resource stock and harvest rate or between the management capital and investment rate, the development trend of the poaching in general cannot be determined when the resource and management capital stock simultaneously vary. It depends mainly on the initial conditions of the resource and management capital stock. Different scenarios that leads to different results were demonstrated in section 8.3.

Under the specific assumptions used to prove the saddle point stability of the steady state solution of the simple model and of the extended model, some critical comparative static effects of exogenous parameters on the equilibrium resource stock can be derived. By using the computer simulation, the results from the comparative static analysis of the general model confirmed the comparative static effects found in the simple model and the extended model. To sum up, the lower the discount rate, the poaching coefficient, the cost coefficient of investment and the depreciation rate of management capital, and the higher the intrinsic growth rate of species, the non-consumptive value coefficient, the gross profit coefficient of harvest and the efficiency coefficient of management capital are, the higher the equilibrium resource stock level will be.

Of these comparative static results, what especially worth while noting is the impact of an variation of the gross profit coefficient of harvest on the equilibrium resource stock. Contrary to the conclusion of the Clark model and to the popular belief, the general model demonstrated that, other things being equal, the higher the gross profit coefficient of harvest (termed as price/harvest cost ratio in the context of the Clark model) is, the higher the equilibrium resource stock level will be. This conclusion is consistent with that drawn by the Swanson model. The reason underlying this difference in model results is, that both Swanson´s and our model take the factor of the evolution of management capacity into account, while the Clark model did not. In the context of the Clark model, an increase in the price/harvest cost ratio will enhance the incentive to harvest resources, without a concomitant increase in the resource stock resulting from the devotion of a higher level of management capital. However, in the context of our model, a higher gross profit coefficient of harvest will induce more capital devoted to the management of resources, and an improved management capacity will finally result in a higher equilibrium resource stock level.

Although both the Swanson model and our general model agree with the comparative static effect of an variation of the gross profit coefficient of harvest on the equilibrium resource stock, there are some differences between the results of the two models. First, the general model clearly demonstrated that, on the optimal time path, the harvest rate and the equilibrium resource stock vary in the same direction, and the investment rate and the equilibrium management capital stock vary in the opposite direction, while the Swanson model did not depict the interactions between these variables. Next, the Swanson model cannot study the comparative static effects of some other parameters such as the poaching coefficient, the cost coefficient of investment, the depreciation rate of management capital, the non-consumptive value coefficient of species and the efficiency coefficient of management capital, because it treated management as a flow variable and therefore cannot handle the relevant problems of capital accumulation, and it did not take the problems of poaching, anti-poaching and non-consumptive value of species into account.

As a result of the usually observed correlation between the gross profit coefficient of harvest and the poaching coefficient in the real world when certain conservation policies are brought into practice, the impact of a simultaneous variation of these two parameters on the equilibrium resource stock has also been addressed. Although the net effect is in many cases ambiguous, such discussion help investigate the origin of the controversy involving the conservation and the consumptive use of certain wild species, such as in the case of the African elephant.

The size of the equilibrium resource stock depends on a variety of factors, as previously discussed. In a special case of the simple model which did not consider the non-consumptive value of resources, the equilibrium resource stock level is always lower than the stock level which can afford the maximum sustainable yield. Once the factors of the non-consumptive value and poaching were taken into account, as the simple, the extended and the general model did, the equilibrium resource stock level will not be necessarily lower or higher than the maximum sustainable yield stock level.

10.1.2 Conclusions of the case studies

The case study involving the national park system of Taiwan showed that the national park system is successful, at least at its beginning stage between the year 1984 and 2000, in the sense that it has in principle effectively safeguarded biodiversity within park boundaries. Its success can be attributed to the two pivotal factors: the political and financial support of the central government and the strict protection policy. But nowadays the strict protection policy, as a critical factor contributing to safeguarding biodiversity, ironically hindered the planned enlargement of the national park system, because it also intensified the conflicts between park authorities and local communities. Whether the national park system of Taiwan can overcome this problem by means of some innovative approaches, such as the co-management, or by modifying the strict preservation policy, remains to be seen.

The case study involving the Danayiku Nature Park at Shan-Mei, Taiwan was conducted to explore the performance of a typical community-based conservation project, an important form of the sustainable resource use strategy.

Another five similar conservation projects were also addressed so that a comparison between different cases can be done. An assessment procedure based on the findings of the general model was developed. The assessment showed that only Shan-Mei possesses the biological, economic and social conditions that are generally adequate to the application of the sustainable use strategy. This explains to a great extent why sustainable use strategy scored a success in Shan-Mei, while similar projects in vicinal communities failed or did not work so well like Shan-Mei did.

Based on the findings of the Clark model and the Swanson model, three fundamental factors, including the intrinsic growth rate of species, the discount rate and the price/cost ratio of harvest (the gross profit coefficient of harvest in our terminology), were usually discussed when resource use problems and relevant conservation issues were concerned about. However, the case study showed that, rather than the three often discussed factors, they are non-consumptive value of resources, efficiency of management capital and investment cost of management capital that explained the difference of the performance of various use programs in this case. These findings confirmed that it is worth while extending traditional bioeconomic models to study problems of management capital accumulation, non-consumptive value of wild species and illegal harvest. Such modifications help us have more insight into the complex resource use problems and relevant conservation issues.

10.2 Policy implications

The theoretic models and case studies offered some critical policy implications. First, as defined in section 4.3, the use of renewable resources is sustainable, if the equilibeium populations of utilized species will not be reduced to such levels that they are vulnerable to local extinction, that their ecological roles in the ecosystem is impaired, and that they lose their significance as useful resources to human users. Following these criteria, our models showed that the impact of the sustainable use approach on conservation is double-edged, in the sense that the sustainable use approach will not necessarily result in a better conservation status of renewable resources in a given protected area, because, depending on a variety of biological, economic and social conditions, the equilibrium resource stock could be higher or lower than the initial stock level, and, in the case when the equilibrium resource stock is lower than the initial stock level, it can sometimes be reduced to such a low level that the previous criteria cannot be satified. This reflects the fact that, in the complicated real world there is no single, omnipotent approach that can solve all the conservation problems in different cases throughout the world. The sustainable use approach could, or could not be a feasible conservation strategy. To evaluate the feasibility of the sustainable use of wild species as a conservation strategy and to reduce the risk of overexploitation, it always needs an overall assessment about relevant biological, economic and social conditions at the site concerned. It is highly dangerous to judge the feasibility of a sustainable use program by using only one or only few indicators, for example by using only the intrinsic growth rate and/or the discount rate.

As section 8.3 concluded, the variation of the poaching rate is not an appropriate indicator for evaluating the success of the sustainable use strategy. Under the general premise that the more the equilibrium resource stock closes to the carrying capacity, the better it would be for the whole ecosystem, we may use the equilibrium resource stock as an indicator for judging the feasibility of the use approach as a conservation strategy. Accordingly, the eight parameters affecting the equilibrium resource stock may be viewed as indicators for evaluating the success probability of a sustainable use project before or when it is brought into practice. The sustainable use strategy may potentially be more appropriate in sites with more positive indicators, namely high intrinsic growth rate of species, non-consumptive value coefficient, gross profit coefficient of harvest and efficiency coefficient of management capital, and low discount rate, poaching coefficient, cost coefficient of investment and depreciation rate of management capital, than those sites with less positive indicators. Based on this conclusion, some policy implications can be drawn.

At the individual species level, some long-lived and slow-reproducing species, such as primates, elephants, whales and sharks, have generally low intrinsic growth rates and are particularly vulnerable to harvest. At the ecosystem level, compared to open grasslands or habitats with a mosaic of forest and grassland, forest ecosystems, especially tropical forests, as a whole are particularly vulnerable to overharvesting of plant and mammalian communities. Hence, a specially cautious attitude toward their harvest problems should be taken, although this does not necessarily imply that, from the perspective of conservation, they should not be utilized in any case. Sometimes the positive effects of some other feasible conditions on the resource stock may compensate for the risk resulted from the low intrinsic growth rate.

At the national level, compared to the developed countries, the developing countries are generally characterized by high discount rate, high poaching pressure and low non-consumptive value of wild species. Given these conditions, the success probability of the sustainable use approach might be relatively low in developing countries. This might imply that a more conservative attitude toward the application of the sustainable use approach should be taken in developing countries, especially in the tropical rain forests where the overall reproductive rate of faunal and floral communities is low. Certainly, one thing we should bear in mind is that the previous general conclusion neglects the considerable differences in socio-economic and biological conditions between various countries, regions and habitat types. Whether the use strategy is appropriate, depends always on the site- and species-specific conditions. Nonetheless, the general conclusions provide a fundamental direction for the rethinking of the current conservation policies.

Based on the discussion in section 8.4 about the usual correlation between the gross profit coefficient of harvest and the poaching coefficient, we may conclude that sporting hunting as a conservation strategy might be a feasible policy option, for example in the cases of the conservation of the African elephant and the wildlife of Taiwan, because it could enhance the gross profit coefficient of harvest while the poaching coefficient would not be affected.

Finally, the case study in the A-Li-Shan area of Taiwan showed that, under prevailing natural, economic and social conditions in Taiwan, the success probability of community-based conservation programs initiated by indigenous communities is relatively low, and, from the viewpoint of conservation, the performance of community-based conservation programs is in general not so good as it is supposed to be, if those villages studied can represent the ordinary indigenous communities of Taiwan. Compared to the national park system managed by the central government, community-based conservation regime can hardly create systems of a scale sufficient to preserve large portions of ecosystems. It is clear that, under current circumstances, only the central government has the capacity to support and manage a large, systematic protected area network in Taiwan. It is unreasonable to suppose that indigenous communities (and other local communities) can overcome the problems like poaching, institution building and financial deficit more effectively than the central government. The argument that decentralized, community-based conservation programs managed by indigenous communities would work better than traditional protected areas managed by the central government should be questioned. If the community-based conservation model is promoted considering the reasons of rural community development and/or social justice, intensive outside (governmental or non-governmental) supports will be needed to offset some disadvantageous exogenous conditions prevailing in most local communities of Taiwan, such as active poaching, high discount rate, high investment cost of management capital and low non-consumptive value of renewable resources.

10.3 Limits in applicability of the theoretic model and recommendations for further research

Mainly based on the general model, the conclusions and policy implications presented in section 10.1 and 10.2 were drawn. However, one should also be careful to realize that some features of the general model could limit it´s applicability.

First, the comparative static and the phase diagram analysis of the general model were conducted by using a set of specific functional forms for relevant functions and by applying the method of computer simulation. Otherwise, no unambiguous results can be yielded via analytical method in the comparative static analysis, and, as generally recognized, it is not possible to depict the relevant phase diagrams as a result of the complex interaction between multiple state and control variables. It is not certain, whether some another sets of specific functional forms will fundamentally change the model results.

Secondly, the phase diagram analysis of the general model demonstrated that the optimal paths are globally monotonic. However, in models involving multiple state and control variables, it usually happens that some initial conditions for the state variables can be found so that the optimal paths are non-monotonic. In this case, it must be recognized that we are not sure whether the optimal paths of the general model are always monotonic.

Thirdly, the influence of certain exogenously given policies, for example the lifting of the hunting ban, on the gross profit coefficient of harvest and the poaching coefficient was addressed in section 8.4 and 8.5. Nonetheless, the two coefficients are also exogenous variables in the general model. A modified model would be better, if it can endogenize the two coefficients and study what would happen, when certain exogenously given policies are implemented.

Fourthly, we confine the theoretic models to addressing the related harvest and management issues when the sustainable use strategy is applied in given protected areas. However, the influence of the sustainable use strategy on land use decisions outside existing protected areas is an another critical dimension that is worthy of being investigated, from the viewpoint of both the economic theory and conservation. The general model might be modified in the future to simultaneously consider the interaction between resource use, land use competition, evolution of management capacity and illegal harvest.

Fifthly, our model is a typical one-species model which considers only the harvested species. The impacts of harvesting the target species on all the other species living within the same ecosystem are neglected in such a model. The impacts would be especially enormous when the target species in question is a keystone species. From the perspective of biology, most of such impacts cannot even be modeled by the economic theory. Maybe, the general model could be modified in order to address the cases when two species have competitive or predatory relationships, and they are simultaneously harvested or only one species is harvested.

Sixthly, apart from the intrinsic growth rate of species, there are still many special biological factors that can influence the outcomes of the resource use. For example, tropical rain forest species are easily threatened by local extinctions, because they are generally characterized by high diversity and low densities. Species whose behavior allows easy harvest, that do not have the ability to recolonize hunted area, or that are intrinsically rare are highly vulnerable to harvest. In addition, our model is a deterministic model. However, it has long been recognized that uncertainty which emerges from the inherent stochasticity of ecosystems and from human ignorance about biological and ecological knowledge usually leads to overexploitation and significantly raises the risk of extinction. It is therefore necessary to develop a stochastic model and thereby to study the influence of the uncertainty on the optimal use of renewable resources and relevant management issues.

Finally, some directions are worthy of being studied in further research. In this dissertation, the resource harvest and management problems were investigated under the premise of the sole ownership. In fact, some other ownership regimes, for example the co-management which means that local communities and park authorities share the management and use rights of resources, might be as important as the sole ownership regime, and it is therefore worthy of being addressed, whether different forms of ownership regime will lead to different outcomes of resource harvest and management. Furthermore, we may attempt to obtain detailed data involving resource stock, harvest rate, management capital stock and investment, thereby conduct a econometric analysis, test the results of the general model and quantify the interactions between these variables. This may help offer a more precise assessment foundation for the field work and reduce the risk of overexploitation, before or when a resource use project is brought into practice.

References

[1] Adams, J. S. and T. O. McShane (1996) The Myth of Wild Africa. University of California Press.

[2] Adams, W. and D. Hulme (2001) Conservation and Community: Changing Narratives, Policies & Practices in African Conservation. In D. Hulme and M. Murphree (Eds.), African Wildlife & Livelihoods: The Promise & Performance of community Conservation. James Currey Ltd., pp. 9-23.

[3] Arrow, K. J. and A. C. Fisher (1974) Environmental preservation, uncertainty, and irreversibility. Quarterly Journal of Economics, 88: 312-319.

[4] Aylward, B. (1992) Appropriating the value of wildlife and wildlands. In T. M. Swanson and E. B. Barbier (Eds.), Economics for the wilds. Earthscan, London, pp. 34-64.

[5] Barbier, E. B. (1992) Economics for the wilds. In T. M. Swanson and E. B. Barbier (Eds.), Economics for the wilds. Earthscan, London, pp. 15-33.

[6] Barnard, P., C. J. Brown, A. M. Jarvis, A. Robertson and L. V. Rooyen (1998) Extending the Namibian protected area network to safeguard hotspots of endemism and diversity. Biodiversity and Conservation, 7: 531-547.

[7] Barrow, E. and M. Murphree (2001) Community Conservation: From Concept to Practice. In D. Hulme and M. Murphree (Eds.), African Wildlife & Livelihoods: The Promise & Performance of community Conservation. James Currey Ltd., pp. 24-37.

[8] Bennett, E. L. and J. G. Robinson (2000a) Hunting for the snark. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 1-9.

[9] Bennett, E. L. and J. G. Robinson (2000b) Hunting for sustainability: The start of a synthesis. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 499-519.

[10] Bishop, R.C. (1978) Endangered species and uncertainty: The economics of a Safe Minimum Standard. American Journal of Agricultural Economics, 60: 10-18.

[11] Bishop, R.C. (1982) Option value: An exposition and extention. Land Economics, 58(1): 1-15.

[12] Bodmer, R. E. (1995a) Susceptibility of mammals to overhunting in Amazonia. In J. Bissonette and P. Krausmanm (Eds.), Integrating People and Wildlife for a Sustainable Future, pp.292-5. Bethesda, MD: The Wildlife Society.

[13] Bodmer, R. E. (1995b) Managing Amazonian wildlife: Biological correlates of game choice by detribalized hunters. Ecological Applications 5: 872-877.

[14] Bodmer, R. E., J. F. Eisenberg, and K. H. Redford (1997a) Hunting and the likelihood of extinction of Amazonian mammals. Conservation Biology, 11: 460-6.

[15] Bodmer, R. E., J. W. Penn, P. Puertas, L. Moya I., and T. G. Fang (1997b) Linking conservation and local people through sustainable use of natural resources: Community-based management in the Peruvian Amazon. In C. H. Freese (ed.), Harvesting Wild Species: Implications for Biodiversity Conservation. The Johns Hopkins University Press, Baltimore and London, pp. 315-358.

[16] Bodmer, R. E. and P. E. Puertas (2000) Community-based comanagement of wildlife in the Peruvian Amazon. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 395-409.

[17] Boyce, J. R. (1995) Optimal capital accumulation in a fishery: A nonlinear irreversible investment model. Journal of Environmental Economics and Management, 28: 324-339.

[18] Brandon, K. (1997) Policy and practical considerations in land-use strategies for biodiversity conservation. In R. Kramer, C. P. van Schaik and J. Johnson (Eds.) Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, pp. 90-114.

[19] Brandon, K., K. H. Redford, and S. E. Sanderson (Eds.) (1998) Parks in Peril: People, Politics, and Protected Areas. Island Press, Washington, D. C..

[20] Brandon, K. and M. Wells (1992) Planning for people and parks: design dilemmas. World Development, 20: 557-570.

[21] Bromley, D. W. (1994) Economic dimensions of community-based conservation. In D. Western and R. M. Wright (Eds.), Natural Connections: Perspectives in Community-based Conservation. Island Press, Washington, D. C., pp. 428-447.

[22] Brown, K. (1993) Biodiversity. In D. Pearce (ed.), Blueprint 3: Measuring sustainable development. Earthscan, London, pp. 98-114.

[23] Caro, T. M. and G. O´Doherty (1999) On the use of surrogate species in conservation biology. Conservation Biology, 13(4): 805-814.

[24] Caughley, G. (1993) Elephants and Economics. Conservation Biology, 7: 943-945.

[25] Chang, L.-S. (2001) On indigenous hunting tradition and the modification of the National Park Act. The Nature, 70: 98-101. (In Chinese)

[26] Child, B. (2000) Application of the Southern African Experience to Wildlife Utilization and Conservation in Kenya and Tanzania. In H. H. T. Prins et al. (Eds.) Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, pp. 459-467.

[27] Ciriacy-Wantrup, S. V. (1952) Resource Conservation. Berkeley: University of California Press.

[28] Clark, C. W. (1973) Profit maximization and the extinction of animal species. Journal of Political Economy, 81(2): 950-961.

[29] Clark, C. W. (1976) Mathematical Bioeconomics: The Optimal Management of Renewable Resources. John Wiley & Sons, Inc..

[30] Clark, C. W. and G. R. Munro (1975) The economics of fishing and modern capital theory: A simplified approach. Journal of Environmental Economics and Management, 2: 163-180.

[31] Clark, C. W., F. H. Clarke, and G. R. Munro (1979) The optimal exploitation of renewable resource stocks: Problems of irreversible investment. Econometrica, 47: 25-47.

[32] Clayton, L. and E. J. Milner-Gulland (2000) The trade in wildlife in north Sulawesi, Indonesia. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 473-496.

[33] Council of Agriculture and Department of National Parks, Construction and Planning Administration, Ministry of Interior, Republic of China (COA and DNP) (1992) Island of Diversity-Nature Conservation in Taiwan. COA and DNP, Taiwan, ROC.

[34] Council of Agriculture, Republic of China (COA) (1997) The Nature Reserves in Taiwan. COA, Taiwan, ROC.

[35] Construction and Planning Administration, Ministry of Interior, Republic of China (CPA) (2000) Yearly Report of Construction and Planning Affair Indicator, Taiwan and Fuchien Area, Republic of China. CPA, Taiwan, ROC.

[36] Dai, C.-F., K.-M. Kuo, Y.-T. Chen, and C.-H. Chuang (1998) Changes of coral communities in Nanwan Bay, Kenting National Park: 1987-1997. Journal of National Park, 8(2): 79-99. (In Chinese)

[37] Dai, C.-F., K.-M. Kuo, Y.-T. Chen, and C.-H. Chuang (1999) Changes of coral communities on the east and west coast of the Kenting National Park. Journal of National Park, 9(2): 131-143. (In Chinese)

[38] Daily, G. C. and P. R. Ehrlich (1995) Population Extinction and the Biodiversity Crisis. In C. A. Perrings, K.-G. Mäler, C. Folke, et al. (Eds.), Biodiversity Conservation. Kluwer Academic Publishers, The Netherlands, pp. 45-56.

[39] Dinerstein, K. and E. D. Wikramanayake (1996) Beyond ´Hotspots´: How to Prioritize Investments to Conserve Biodiversity in the Indo-Pacific Region. In F. B. Samson and F. C. Knopf (Eds.), Ecosystem Management. Springer Verlag, New York, pp. 32-45.

[40] Dixon, J. A. and P. B. Sherman (1991) Economics of Protected Areas. Earthscan, London.

[41] Department of National Parks, Construction and Planning Administration, Ministry of Interior, Taiwan, Republic of China (DNP) (1999) National Parks of Taiwan. DNP, Taiwan, ROC.

[42] Dockner, E. (1985) Local satbility analysis in optimal control problems with two state variables. In G. Feichtinger (ed.), Optimal Control Theory and Economic Analysis, 2: 89-103.

[43] Dublin, H. T., T. Milliken and R. F. W. Barnes (1995) Four Years After the CITES Ban: Illegal killing of Elephants, Ivory Trade and

Stockpiles. Gland, Switzerland: IUCN/SSC African Elephant Specialist Group.

[44] Duffy, R. (2000) Killing for Conservation: Wildlife Policy in Zimbabwe. Indiana University Press, Bloomington & Indianapolis.

[45] Ehrlich, P. and A. Ehrlich (1981) Extinction. Random House: New York.

[46] Ehrlich, P. R. and G. C. Daily (1993) Population extinction and saving biodiversity. AMBIO, 22: 64-68.

[47] Ehrlich, P. R. and E. O. Wilson (1991) Biodiversity Studies: Science and Policy. Science, 253: 758-762.

[48] Eidsvik, H. K. (1992) Strengthening Protected Areas Through Philosophy, Science and Management: A Global Perspective. In J. H. M. Willison, C. Drysdale, T. B. Herman, et al. (Eds.), Science and Management of Protected Areas. Elsevier, Amsterdam, pp. 9-18.

[49] Eisenberg, J. F. (1980) The density and biomass of tropical mammals. In M. E. Soulé and B. A. Wilcox (eds.), Conservation Biology: An Evolutionary Ecological Perspective. Sunderland, MA: Sinauer Associates. pp. 34-55.

[50] Eiswerth, M. E. and J. C. Haney (1992) Allocating conservation expenditures: accounting for inter-species genetic distinctiveness. Ecological Economics, 5(1): 235-250.

[51] Erwin, T. (1988) The tropical forest canopy: The heart of biotic diversity. In E. O. Wilson (Ed.), Biodiversity. National Academy Press, Washington, pp. 123-129.

[52] Executive Yuan, Republic of China (EYROC) (1981) Budget of the Central Government, Budget Year 1982, Republic of China, p. 198. (In Chinese)

[53] Executive Yuan, Republic of China (EYROC) (1982) Budget of the Central Government, Budget Year 1983, Republic of China, p. 213. (In Chinese)

[54] Executive Yuan, Republic of China (EYROC) (1983) Budget of the Central Government, Budget Year 1984, Republic of China, p. 204. (In Chinese)

[55] Executive Yuan, Republic of China (EYROC) (1984) Budget of the Central Government, Budget Year 1985, Republic of China, p. 225. (In Chinese)

[56] Executive Yuan, Republic of China (EYROC) (1985) Budget of the Central Government, Budget Year 1986, Republic of China, pp. 234-235. (In Chinese)

[57] Executive Yuan, Republic of China (EYROC) (1986) Budget of the Central Government, Budget Year 1987, Republic of China, pp. 230-233. (In Chinese)

[58] Executive Yuan, Republic of China (EYROC) (1987) Budget of the Central Government, Budget Year 1988, Republic of China, pp. 271-277. (In Chinese)

[59] Executive Yuan, Republic of China (EYROC) (1988) Budget of the Central Government, Budget Year 1989, Republic of China, pp. 290-296. (In Chinese)

[60] Executive Yuan, Republic of China (EYROC) (1989) Budget of the Central Government, Budget Year 1990, Republic of China, pp. 310-315. (In Chinese)

[61] Executive Yuan, Republic of China (EYROC) (1990) Budget of the Central Government, Budget Year 1991, Republic of China, pp. 309-314. (In Chinese)

[62] Executive Yuan, Republic of China (EYROC) (1991) Budget of the Central Government, Budget Year 1992, Republic of China, pp. 342-347. (In Chinese)

[63] Executive Yuan, Republic of China (EYROC) (1992) Budget of the Central Government, Budget Year 1993, Republic of China, pp. 447-457. (In Chinese)

[64] Executive Yuan, Republic of China (EYROC) (1993) Budget of the Central Government, Budget Year 1994, Republic of China, pp. 652-666. (In Chinese)

[65] Executive Yuan, Republic of China (EYROC) (1994) Budget of the Central Government, Budget Year 1995, Republic of China, pp. 585-600. (In Chinese)

[66] Executive Yuan, Republic of China (EYROC) (1995) Budget of the Central Government, Budget Year 1996, Republic of China, pp. 605-619. (In Chinese)

[67] Executive Yuan, Republic of China (EYROC) (1996) Budget of the Central Government, Budget Year 1997, Republic of China, pp. 612-628. (In Chinese)

[68] Executive Yuan, Republic of China (EYROC) (1997) Budget of the Central Government, Budget Year 1998, Republic of China, pp. 520-532. (In Chinese)

[69] Executive Yuan, Republic of China (EYROC) (1998) Budget of the Central Government, Budget Year 1999, Republic of China, pp. 317-329. (In Chinese)

[70] Executive Yuan, Republic of China (EYROC) (1999) Budget of the Central Government, Budget Year 2000, Republic of China, pp. 273-255. (In Chinese)

[71] Fa, J. E., J. Juste, J. Perez del Val, and J. Castroviejo (1995) Impact of market hunting on mammal species in Equatorial Guinea. Conservation Biology 9: 1107-1115.

[72] Fisher, A. C. and J. V. Krutilla (1985) Economics of Nature Preservation. In V. K. Allen and J. L. Sweeney (Eds.), Handbook of Natural Resource and Energy Economics, Volume I, Elsevier, pp. 165-189.

[73] Fisher, A. C. and W. M. Hanemann (1987) Quasi-option value: Some misconceptions dispelled. Journal of Environmental Economics and Management, 14: 183-190.

[74] Fitzgibbon, C. D., J. Mogaka, and J. H. Fanshawe (1995) Subsistence hunting in Arabuko-Sokoke Forest, Kenya, and its effects on mammal populations. Conservation Biology 9: 1116-1126.

[75] Freeman, A. M. (1985) Supply uncertainty, option price and option value. Land Economics, 61: 176-181.

[76] Freese, C. H. (Ed.) (1997) Harvesting Wild Species: Implications for Biodiversity Conservation. The Johns Hopkins University Press, Baltimore.

[77] Freese, C. H. (1998) Wild Species as Commodities: Managing Markets and Ecosystems for Sustainability. Island Press, Washington, D. C..

[78] Gadgil, M. (1992) Conserving biodiversity as if people matter: A case study from India. Ambio, 21: 266-270.

[79] Ghimire, K. B. and M. P. Pimbert (1997) Social Change and Conservation: An Overview of Issues and Concepts. In K. B. Ghimire and M. P. Pimbert (Eds.) Social Change and Conservation. Environmental Politics and Impacts of National Parks and Protected Areas. Earthscan, London, pp. 1-45.

[80] Gordon, H. S. (1954) The economic theory of a common property resource: The fishery. Journal of Political Economy, 62: 124-142.

[81] Gould, J. R. (1972) Extinction of a fishery by commercial exploitation: A note. Journal of Political Economy, 80: 1031-1038.

[82] Green, M. J. B. and J. Paine (1999) State of the world´s protected areas at the end of the 20th century. In S. Stolton and N. Dudley (Eds.), Partnerships for Protection: New Strategies for Planning and Management for Protected Areas. Earthscan Publications Ltd., pp. 19-28.

[83] Grootenhuis, J. G. and H. H. T. Prins (2000) Wildlife utilisation: a justified option for sustainable land use in African savannas. In H. H. T. Prins, J. G. Grootenhuis and T. T. Dolan (Eds.) Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, pp. 469-482.

[84] Grove, N. (1988) Quietly Conserving Nature. National Geographic, 174(January): 818-844.

[85] Gullison, R. E. (1998) Chapter 6: Will Bigleaf Mahogany be conserved through sustainable use? In E. J. Milner-Gulland and R. Mace, Conservation of Biological Resources. Blackwell Science Ltd.

[86] Hackel, J. D. (1999) Community conservation and the future of Africa´s wildlife. Conservation Biology, 13(4): 726-734.

[87] Hart, J. A. (2000) Impact and sustainability of indigenous hunting in the Ituri Forest, Congo-Zaire: A comparison of unhunted and hunted duiker populations. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 106-153.

[88] Hawksworth, D. L. (1991) The fungal dimension of biodiversity: Magnitude, significance, and conservation. Mycological Research, 95: 641-655.

[89] Hearne, J. and M. Mckenzie (2000) Compelling reasons for game ranching in Maputaland. In H. H. T. Prins, J. G. Grootenhuis and T. T. Dolan (Eds.) Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, pp. 417-438.

[90] Henry, C. (1974) Investment decisions under uncertainty: The ´irreversibility effect´. American Economic Review, 64(6): 1006-1012.

[91] Huang, Y.-W. (1999) Aboriginal reserves policy in Taiwan´s national parks- an institutional and spatial perspective. Journal of National Park, 9(2): 182-198. (In Chinese)

[92] Hurt, R. and P. Ravn (2000) Hunting and its benefits: an overview of hunting in Africa with special reference to Tanzania. In H. H. T. Prins, J. G. Grootenhuis and T. T. Dolan (Eds.) Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, pp. 295-313.

[93] Ivory Trade Review Group (ITRG) (1989) The Ivory Trade and the Future of the African Elephant. Report to the Conference of the Parties to CITES, Lausanne.

[94] IUCN (1994) Guidelines for Protected Area Management Categories, CNPPA with the assistance of WCMC, IUCN, Gland and Cambridge.

[95] IUCN (1998) 1997 United Nations List of Protected Areas, prepared by WCMC and WCPA, IUCN, Gland and Cambridge.

[96] IUCN/UNEP/WWF (1980) World Conservation Strategy. Living resource conservation for sustainable development. IUCN, Gland, Switzerland.

[97] IUCN/UNEP/WWF (1991) Caring for the Earth: a strategy for sustainable living. IUCN, Gland, Switzerland.

[98] Janzen, D. H. (1994) Wildland biodiversity management in the tropics: Where are we now and where are we going?. Vida Silvestre Neotropical, 3: 3-15.

[99] Johansson, P.-O. (1988) On the properties of supply-side option value. Land Economics, 64: 86-97.

[100] Katz, E. G. (2000) Social capital and natural capital: A comparative analysis of land tenure and natural resource management in Guatemala. Land Economics, 76(1): 114-132.

[101] Kock, M. D. (1996) Zimbabwe: a model for the sustainable use of wildlife and the development of innovative wildlife management pratices. In V. J. Taylor and N. Dunstone (Eds.) The Exploitation of Mammal Populations. Chapman & Hall, London, pp. 229-249.

[102] Kramer, R. A. and C. P. van Schaik (1997) Preservation Paradigms and Tropical Rain Forests. In R. Kramer, C. P. van Schaik and J. Johnson (Eds.) Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, pp. 3-14.

[103] Kramer, R., C. P. van Schaik and J. Johnson (Eds.) (1997) Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press.

[104] Krautkraemer, J. A. (1995) Incentives, Development and Population: A Growth-Theoretic Perspective. In T. M. Swanson (Ed.), The Economics and Ecology of Biodiversity Decline: The Forces Driving Global Change. Cambridge University Press, pp. 13-24.

[105] Krutilla, J. V. (1967) Conservation reconsidered. American Economic Review, 57: 777-786.

[106] Lant, C. L. (1994) The role of property rights in economic research on U.S. wetlands policy. Ecological Economics, 11: 27-33.

[107] Laurance, W. F., H. L. Vasconcelos and T. E. Lovejoy (2000) Forest loss and fragmentation in the Amazon: implications for wildlife conservation. Oryx, 34(1): 39-45.

[108] Lavigne, D. M., C. J. Callaghan and R. J. Smith (1996) Sustainable utilization: the lessons of history. In V. J. Taylor and D. Dunstone (Eds.), The Exploitation of Mammal Populations, Chapman & Hall, London, pp. 250-265.

[109] Lee, R. J. (2000) Impact of subsistence hunting in north Sulawesi, Indonesia, and conservation options. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 455-472.

[110] Lewis, D. M. and A. Phiri (1998) Wildlife snaring-an indicator of community response to a community-based conservation project. Oryx, 32(2): 111-121.

[111] Li, C.-Z. and K.-G. Löfgren (1998) A dynamic model of biodiversity preservation. Environment and Development Economics, 3: 157-172.

[112] Li, T.-M. and H.-C. Tang (1999) Forever Danayiku: retrospect in Shan-Mei community development. Unpublished research report of Chia-Yi County and the Political Institute of National Chung-Cheng University, Taiwan. (In Chinese)

[113] Lin, L. (2000) Nature reserves and national parks. Quarterly Journal of Construction, 9(4): 13-30. (In Chinese)

[114] Lin, Y,-S. (2000) The treaty of biodiversity (5): protected areas. The Nature, 69: 110-115. (In Chinese)

[115] Liu, J.-S. (2000) Promoting indigenous economy through sustainable use of natural resources. The Nature, 67: 34-41. (In Chinese)

[116] Lovejoy, T. E. (1980) A Projection of Species Extinction. In G. Barney (Ed.), The Global 2000 Report to the President. Council on Environmental Quality: Washington, D. C..

[117] Lovejoy, T. E., R. O. Bierregaard Jr., A. B. Rylands, J. R. Malcolm, C. E. Quintela, L. H. Harper, K. S. Brown, Jr., A. H. Powell, G. V. N. Powell, H. O. R. Schubart, and M. B. Hays (1986) Edge and other effects of isolation on Amazon forest fragments. In M. E. Soulé (Ed.) Conservation Biology: The Science of Scarcity and Diversity. Sunderland, Mass.: Sinauer, pp. 257-285.

[118] Lu, D.-J. (1999) Participation, Institutions and Protected Area Management- A Qualitative Analysis of the Wildlife Refuges in Taiwan. Unpublished dissertation of University of Wales, Aberystwyth.

[119] Ludwig, D., R. Hilborn and C. Walters (1993) Uncertainty, resource exploitation and conservation: lessons from history. Science, 260(17): 36.

[120] Lugo, A. E., J. A. Parrotta and S. Brown (1993) Loss of species caused by tropical deforestation and their recovery through management. AMBIO, 22(2-3): 106-109.

[121] MacArthur, R. M. and E. O. Wilson (1967) The Theory of Island Biogeography. Monographs in Population Biology. Princeton University Press, Princeton, New Jersey.

[122] Machlis, G. E. and D. L. Tichnell (1987) Economic development and threats to National Parks: a preliminary analysis. Environmental Conservation, 14(2): 151-156.

[123] MacKinnon, K. (1997) The ecological foundations of biodiversity protection. In R. Kramer, C. V. Schaik and J. Johnson (Eds.) Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, pp. 36-63.

[124] Madhusudan, M. D. and K. U. Karanth (2000) Hunting for an answer: Is local hunting compatible with large mammal conservation in India? In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 339-355.

[125] Mangel, M., L. M. Talbot, G. K. Meffe, M. T. Agardy, D. L. Alverson, J. Barlow, D. B. Botkin, G. Budowski, T. Clark, J. Cooke, R. H. Crozier, P. K. Dayton, D. L. Elder, C. W. Fowler, S. Funtowicz, J. Giske, R. J. Hofman, S. J. Holt, S. R. Kellert, L. A. Kimball, D. Ludwig, K. Magnusson, B. S. Malayang III, C. Mann, E. A. Norse, S. P. Northridge, W. F. Perrin, C. Perrings, R. M. Peterman, G. B. Rabb, H. A. Regier, J. E. Reynolds III, K. Sherman, M. P. Sissenwine, T. D. Smith, A. Starfield, R. J. Taylor, M. F. Tillman, C. Toft, J. R. Twiss, Jr., J. Wilen, and T. P. Young (1996) Principles for the conservation of wild living resources. Ecological Applications, 6: 338-362.

[126] McCullough, D. R. (1984) Lessons from the George Reserve, Michigan. In L. K. Halls (ed.), White-tailed deer: Ecology and management. Harrisburg, Pa.: Stackpole.

[127] McNeely, J. A. (1988) Economics and biological diversity: Developing and using economic incentives to conserve biological resources. Gland, Switzerland: IUCN.

[128] McNeely, J. A., J. R. Miller, W. V. Reid, et al. (1990) Conserving the World´s Biological Diversity. IUCN, Gland, Switzerland.

[129] Medellín, R. A. (1999) Sustainable Harvest for Conservation. Conservation Biology, 13(2): 225.

[130] Meffe, G. K. and C. R. Carroll (1994) Principles of Conservation Biology. Sinauer Associates, Inc., Sunderland, Massachusetts.

[131] Milner-Gulland, E. J. and N. Leader-Williams (1992) Illegal exploitation of wildlife. In T. M. Swanson and E. B. Barbier (Eds.), Economics for the Wilds. Earthscan, London.

[132] Mitchell, J. G. (1994) Our National Parks. National Geographic, 186(4): 2-55.

[133] Mittermeier, R. A. and T. B. Werner (1990) Wealth of plants and animals unites ´megadiversity´countries. Tropicus, 4: 1, 4-5.

[134] Murphree, M. W. (1994) The role of institutions in community-based conservation. In D. Western and R. M. Wright (Eds.), Natural Connections: Perspectives in Community-based Conservation. Island Press, Washington, D. C., pp. 403-427.

[135] Myers, N. (1988) Threatened biotas: ´hotspots´ in tropical forests. The Environmentalists, 8: 1-20.

[136] Myers, N. (1994) Global Biodiversity II: Losses. In G. K. Meffe and C. R. Carroll (Eds.), Principles of Conservation Biology. Sinauer Associates, Inc., Sunderland, Massachusetts, pp. 110-140.

[137] Noss, A. J. (1997) Challenges to nature conservation with community development in central African forests. Oryx, 31(3): 180-188.

[138] Noss, R. F. (1991) Sustainability and wilderness. Conservation Biology, 5:120-122.

[139] Oates, J. F. (1999) Myth and Reality in the Rain Forest: How Conservation Strategies Are Failing in West Africa. University of California Press.

[140] Orians, G. H. (1994) Global Biodiversity I: Patterns and Processes. In G. K. Meffe and C. R. Carroll (Eds.), Principles of Conservation Biology. Sinauer Associates, Inc., Sunderland, Massachusetts, pp. 78-109.

[141] Owen, D. F. (1992) Chapter 6: The abundance and biomass of forest animals. In F. B. Golley (Ed.), Ecosystems of the world 14A: Tropical rain forest ecosystems. Elsevier Scientific Publishing Company.

[142] Pearce, D. W. and R. K. Turner (1990) Economics of Natural Resources and the Environment. Harvester Wheatsheaf.

[143] Pei, K. J. C. (2001) Dancing with Wildlife. (In Chinese)

[144] Peres, C. A. (2000) Evaluating the impact and sustainability of subsistence hunting at multiple Amazonian Forest sites. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 31-56.

[145] Perrings, C. and D. Pearce (1994) Threshold effects and incentives for the conservation of biodiversity. Environmental and Resource Economics, 4: 13-28.

[146] Phillips, A. and J. Harrison (1999) The Framework for International Standards in Establishing National Parks and Other Protected Areas. In S. Stolton and N. Dudley (Eds.), Partnerships for Protection: New Strategies for Planning and Management for Protected Areas. Earthscan Publications Ltd., pp. 13-17.

[147] Pimbert, M. P. and J. N. Pretty (1997) Parks, People and Professionals: Putting ´Participation´ into Protected Area Management. In K. B. Ghimire and M. P. Pimbert (Eds.) Social Change and Conservation. Environmental Politics and Impacts of National Parks and Protected Areas. Earthscan, London, pp. 297-330.

[148] Powell, G. V. N., J. Barborak and M. Rodriguez S. (2000) Assessing representativeness of protected natural areas in Costa Rica for conserving biodiversity: a preliminary gap analysis. Biological Conservation, 93: 35-41.

[149] Prescott-Allen, R. and C. Prescott-Allen (1996) The good, the bad, and the neutral: assessing the sustainability of uses of wild species. In R. and C. Prescott-Allen (Eds.) Assessing the Sustainability of Uses of Wild Species. Occational Paper of the IUCN Species Survival Commission No. 12. Cambridge, UK, pp. 81-101.

[150] Primack, R. B. (1998) Essentials of Conservation Biology, Second Edition. Sinauer Associates, Inc., Massachusetts, U.S.A..

[151] Prins, H. H. T., J. G. Grootenhuis, and T. T. Dolan (Eds.) (2000) Wildlife Conservation by Sustainable Use. Kluwer Academic Publishers, Boston.

[152] Rasker, R., M. V. Martin ,and R. L. Johnson (2000) Economics: Theory versus Practice in Wildlife Management. In M. A. Michael (Ed.) Preserving Wildlife: An International Perspective. Humanity books, New York, pp. 239-262.

[153] Raven, P. H. (1988) Our Diminishing Tropical Forests. In E. O. Wilson (Ed.), Biodiversity. Washington, D. C., pp. 119-122.

[154] Redford, K. H. (1991) The Ecologically Noble Savage. Cultural Survival Quarterly, 15(1): 46-48.

[155] Reid, W. and K. Miller (1989) Keeping Options Alive, World Resources Institute: Washington, D. C..

[156] Reid, W. (1992) How Many Species Will There Be? In T. C. Whitmore and J. A. Sayer (Eds.) Tropical Deforestation and Species Extinction. Chapman and Hall, London.

[157] Reid, W. V., J. A. McNeely, D. B. Tunstall, D. A. Bryant and M. Winograd (1993) Biodiversity Indicators for Policy-Makers. World Resources Institute, Washington, D. C..

[158] Ricklefs, R. E. (1990) Ecology. W. H. Freeman and Company, New York.

[159] Robinson, J. G. (1993) The limits to caring: Sustainable living and the loss of biodiversity. Conservation Biology, 7; 20-28.

[160] Robinson, J. G. (2000) Appendix: Calculating maximum sustainable harvests and percentage offtakes. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 521-524.

[161] Robinson, J. G. and E. L. Bennett (2000) Carrying capacity limits to sustainable hunting in tropical forests. In J. G. Robinson and E. L. Bennett (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 13-30.

[162] Robinson, J. G. and K. H. Redford (1986) Intrinsic rate of natural increase in neotropical forest mammals: Relationship to phylogeny and diet. Oecologia, 68: 516-520.

[163] Robinson, J. G. and K. H. Redford (1991) Sustainable harvest of neotropical forest mammals. In J. G. Robinson and K. H. Redford (Eds.), Neotropical Wildlife Use and Conservation. Chicago: University of Chicago Press, pp. 415-429.

[164] Romer, P. M. (1990) Endogenous technological change. Journal of Political Economy, 98(5): S71-S102.

[165] Roth, H. H. (1997a) Section 2.3.8. Elephants (*Proboscidea*). In H. H. Roth and G. Merz (Eds.), Wildlife Resources: A Global Account of Economic Use. Springer Verlag, pp. 244-256.

[166] Roth, H. H. (1997b) Section 2.3.10. Suiform Ungulates (*Nonruminantia : Artiodactyla*). In H. H. Roth and G. Merz (Eds.), Wildlife Resources: A Global Account of Economic Use. Springer Verlag, pp. 264-271.

[167] Sattler, P. (1992) Planning Towards Consolidation of Queensland´s National Park Estate. In J. H. M. Willison, C. Drysdale, T. B. Herman, et al. (Eds.), Science and Management of Protected Areas. Elsevier, Amsterdam, p. 107-116.

[168] Sayer, J. A. and S. Stuart (1988) Biological diversity and tropical forests. Environmental Conservation, 15: 193-194.

[169] Schmalensee, R. (1972) Option demand and consumer´s surplus: Valuing price changes under uncertainty. American Economic Review, 62(1): 813-824.

[170] Scott, A. D. (1955) The fishery: The objectives of sole ownership. Journal of Political Economy, 63: 116-124.

[171] Scott, J. M., B. Csuti, J. D. Jacobi, et al. (1987) Species richness. BioScience, 37: 782-788.

[172] Shafer, C. L. (1999) History of selection and system planning for US natural area national parks and monuments: beauty and biology. Biodiversity and Conservation, 8: 189-204.

[173] Shah, A. (1995) The Economics of Third World National Parks. Edward Elgar, UK.

[174] Shaw, J. H. (1991) The outlook for sustainable harvests of wildlife in Latin America. In J. G. Robinson and K. H. Redford (Eds.), Neotropical Wildlife Use and Conservation. Chicago: University of Chicago Press, pp. 24-34.

[175] Sherman, P. B. (1989) Market Failure and the Underprovision of Parks and Protected Areas. Unpublished Ph. D. dissertation, University of Hawaii.

[176] Siegfried, W. R. (1989) Preservation of Species in Southern African Nature Reserves. In B. J. Huntley (Ed.), Biotic Diversity in Southern Africa. Oxford University Press, Cape Town.

[177] Simberloff, D. (1986) Are we on the Verge of an Mass Extinction in Tropical Rain Forests?. In D. Elliot (Ed.), Dynamics of Extinction, John Wiley: New York.

[178] Simberloff, D. (1998) Flagships, umbrellas, and keystones: is single-species management passé in the landscape era?. Biological Conservation, 83(3): 247-257.

[179] Skonhoft, A. and J. T. Solstad (1996) Wildlife management, illegal hunting and conflicts: A bioeconomic analysis. Environment and Development Economics 1: 165-181.

[180] SMCDS (1994) Financial Report of Shan-Mei Community Development Society 1994. (In chinese)

[181] SMCDS (1995) Financial Report of Shan-Mei Community Development Society 1995. (In chinese)

[182] SMCDS (1996) Financial Report of Shan-Mei Community Development Society 1996. (In chinese)

[183] SMCDS (1997) Financial Report of Shan-Mei Community Development Society 1997. (In chinese)

[184] SMCDS (1998) Financial Report of Shan-Mei Community Development Society 1998. (In chinese)

[185] SMCDS (1999) Financial Report of Shan-Mei Community Development Society 1999. (In chinese)

[186] SMCDS (2001) Financial Report of Shan-Mei Community Development Society 2000. (In chinese)

[187] Smith, V. L. (1968) Economics of production from natural resources. American Economic Review, 58: 409-431.

[188] Smith, V. L. (1969) On models of commercial fishing. Journal of Political Economy, 77: 181-198.

[189] Solow, A., S. Polasky and J. Broadus (1993) On the measurement of biological diversity. Journal of Environmental Economics and Management, 24: 60-68.

[190] Songorwa, A. N., T. Bührs, and K. F. D. Hughey (2000) Community-Based Wildlife Management in Africa: A Critical Assessment of the Literature. Natural Resources Journal, 40: 603-643.

[191] Spinage, C. (1998) Social change and conservation misrepresentation in Africa. Oryx, 32(4): 265-276.

[192] Stevens, S. (1997) The Legacy of Yellowstone. In S. Stevens (Ed.) Conservation Through Cultural Survival: Indigenous Peoples and Protected Areas. Island Press, pp. 13-32.

[193] Stiling, P. D. (1992) Introductory Ecology. Prentice Hall, Englewood Cliffs, NJ.

[194] Struhsaker, T. T. (1998) A biologist´s perspective on the role of sustainable harvest in conservation. Conservation Biology, 12(4): 930-932.

[195] Sung, B.-M. (1999) The strategy for the impacts on indigenous culture from national parks in Taiwan. Journal of National Park, 9(1): 65-80. (In Chinese)

[196] Swanson, T. M. (1994) The International Regulation of Extinction. Macmillan, London.

[197] Swallow, B. M. and D. W. Bromley (1995) Institutions, governence and incentives in common property regimes for african rangelands. Environmental and Resource Economics, 6: 99-118.

[198] Terborgh, J. and C. P. van Schaik (1997) Minimizing Species Loss: The Imperative of Protection. In R. Kramer, C. P. van Schaik and J. Johnson (Eds.) Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, pp. 15-35.

[199] The Nature (1991) Newsletter, 32, p.121. (In Chinese)

[200] The Nature (1993) Newsletter, 40, p.118. (In Chinese)

[201] The Nature (1994a) Newsletter, 42, p.120. (In Chinese)

[202] The Nature (1994b) Newsletter, 43, p.114. (In Chinese)

[203] The Nature (1995a) Newsletter, 47, p.120. (In Chinese)

[204] The Nature (1995b) Newsletter, 49, p.114. (In Chinese)

[205] The Nature (1996) Newsletter, 53, p.120. (In Chinese)

[206] The Nature (1999) Newsletter, 63, p.118. (In Chinese)

[207] The Nature (2000) An Interview with the Chief Director of the Construction and Planning Administration, 67: 104-111. (In Chinese)

[208] Turner, A. M., C. D. A. Rubec and E. B. Wiken (1992) Canadian Ecosystems: A Systems Approach to Their Conservation. In J. H. M. Willison, C. Drysdale, T. B. Herman, et al. (Eds.), Science and Management of Protected Areas. Elsevier, Amsterdam, p. 117-127.

[209] Udvardy, M. D. F. (1975) A Classification of the Biogeographical Provinces of the World, Occasional Paper 18, IUCN, Gland, Switzerland.

[210] Wacker, H. and J.E. Blank (1998) Ressourcenökonomik, Band I: Einführung in die Theorie regenerativer natürlicher Ressourcen. Oldenbourg Verlag, München.

[211] Wade, R. (1987) The management of common property resources: Collective action as an alternative to privatisation or state regulation. Cambridge Journal of Economics, 11(2): 95-106.

[212] Wood, D. (1995) Conserved to death: Are tropical forests being overprotected from people? Land Use Policy, 12(2): 115-135.

[213] World Conservation Monitoring Centre (WCMC) (1992) Global Biodiversity: Status of the Earth´s Living Resources. London: Chapman & Hall.

[214] Weisbrod, B. (1964) Collective-consumption services of individual-consumption goods. Quarterly Journal of Economics, 78: 471-477.

[215] Weitzman, M. L. (1992) On diversity. Quarterly Journal of Economics, 107: 363-406.

[216] Wen, I.-J. (2000) Conservation of Danayiku and the community development of Shan-Mei. In China Times Foundation (Ed.), Rivers and Community. China Times Press, Taipei, Taiwan. pp. 171-184.

[217] Western, D. and R. M. Wright (1994) The background to community-based conservation. In D. Western and R. M. Wright (Eds.), Natural Connections: Perspectives in Community-based Conservation. Island Press, Washington, D. C., pp. 1-12.

[218] Whittaker, R. H. (1975) Communities and Ecosystems, 2d ed.. Macmillan, New York.

[219] Wildlife Conservation Law (WCL) (1994) The Central Government of the Republic of China.

[220] Wilson, E. O. (1988) The current state of biological diversity. In E. O. Wilson (Ed.), Biodiversity. National Academy Press, Washington, pp. 3-27.

[221] Wilson, E. O. (1992) The Diversity of Life. The Belknap Press of Harvard University Press, Cambridge, MA.

[222] Woodruff, D. S. (1989) The Problems of Conserving Genes and Species. In D. Western and M. C. Pearl (Eds.), Conservation for the Twenty-First Century, Oxford University Press, New York, pp. 76-88.

[223] World Resources Institute (WRI) (1994) World Resources 1994-95. Oxford University Press, Oxford, p.192.