ESSAYS IN ECONOMICS OF RENEWABLE RESOURCES

Erwin Bulte

Stellingen

1. Het verbieden van de handel in ivoor zal op langere termijn niet leiden tot grotere aantallen olifanten (*dit proefschrift*).

2. Het verbod op de commerciële walvisvaart kan met economische argumenten prima verdedigd worden (*dit proefschrift*). Het afschaffen van dit verbod ook (*Bulte and van Kooten (1997), Environmental valuation and declining marginal utility of preservation: the case of the minke whale in the Northeast Atlantic, University of British Columbia, Discussion paper*).

3. Omdat er nog veel onduidelijkheid bestaat over de "optimale hoeveelheid" tropisch regenbos, is de voortdurende klaagzang over het tempo waarmee dit bostype verdwijnt op zijn minst voorbarig.

4. Investeren in teakhout-plantages doet men met name voor de gemoedsrust; de portefeuille en tropische regenbossen hebben er minder baat bij (*Bulte en van Soest, ESB 4093, blz. 132-134*)

5. Als het hoofdstuk over de "No-Ponzi-game condition" tijdig in het Albanees was vertaald, dan had men veel narigheid kunnen voorkomen (*Blanchard and Fisher (1989), Lectures on Macroeconomics, Cambridge: MIT Press*)

6. Gegeven het feit dat per definitie de *steady state* van een niet-lineair systeem nooit bereikt wordt, is alle aandacht voor dit fenomeen in de tekstboeken wat overdreven.

7. Risico-aversie is aanmerkelijk ingewikkelder dan economen over het algemeen veronderstellen; dit kan afgeleid worden uit de observatie dat promovendi soms geen *back up* van het proefschrift maken, terwijl ze zich wel zorgen maken over eventuele gevolgen van de aantasting van milieu en natuurlijke hulpbronnen.

8. De Nederlandse overheid heeft met de overgang van een "rationeel" naar een "biologisch visstandsbeheer" in 1993 een kortzichtig beleid gevoerd, waarmee zowel vissers als natuurbeschermers een slechte dienst is bewezen.

9. Intelligentie is voor 50% het denkvermogen dat iemand heeft, en voor 50% dat wat men denkt dat die persoon heeft.

10. Het huidige AIO-systeem is er beter in geslaagd om intelligente dan om slimme studenten aan te trekken.

11. Veel economen bakken lucht, niet in de laatste plaats omdat de schoorsteen moet roken.

12. Ondanks de rake typering van Paul Theroux ("toeristen weten niet waar ze geweest zijn, reizigers weten niet waar ze heen gaan") is het verschil tussen toeristen en reizigers vele malen kleiner dan laatstgenoemden ooit willen toegeven.

13. De oude Patagonische geneeswijze van schapenschurft (als volgt beschreven door Bruce Chatwin "stop een suikerklontje in het schaap zijn bek en zuig hem onder zijn staart tot het zoet smaakt") zal in Nederland eerder op verzet van dierenartsen dan van tandartsen stuiten.

14. Niets is zo kleingeestig als het coûte que coûte vermijden van elke schijn van burgerlijkheid.

Stellingen behorende bij het proefschrift
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ESSAYS IN ECONOMICS OF RENEWABLE RESOURCES

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PREFACE

This thesis is an overview of my work during the past four years in the field of renewable resource management. Much of this work has been co-authored, and I would like to thank my co-authors (some of which have become good friends) in this preface first. In alphabetical order, I am grateful to (without implicating) Dick Brazee, Arlene Ells, Henk Folmer, Wim Heijman, Rick Horan, Kees van Kooten, Joost Pennings, Jim Shortle and (last but not least) Daan van Soest for their support and the insights they shared with me. Obviously, I owe special thanks to my promotor Henk Folmer, who, among many other things, convinced me that finishing the NAKE program was in my best interest, and my supervisor Wim Heijman. Considering everything, I think it is fair to conclude that we had a pleasant time, without appreciable crises along the way. Further, many thanks to my former colleagues at the Department of General Economics and my new colleagues at the Department of Development Economics for their good-fellowship and all sorts of (technical) assistance. I also want to thank the Foundation LEB fonds, Shell Reisfonds, NWO-SIR fonds, Canadian FRDA Contribution Agreement No. H6.0-14-001, Geelen Consultancy and USIA (United States Information Agency) for financial support. Spending some time abroad was both stimulating and instructive. Finally, I am more than grateful to my friends, family and especially Marèse for enduring and encouraging me along the way (although, of course, without some of these people this book would have been finished for quite some time).

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CHAPTER 1

GENERAL INTRODUCTION

1.1 Introduction

Renewable resource management has received ample attention in recent years, stimulated by growing awareness that many natural resources are wasted, unsustainably exploited or deteriorating because the environment in which they regenerate gets increasingly polluted. For example, it is estimated that tropical forests disappeared at a pace of approximately 15 million hectares per annum, corresponding to about 0.8% of the stock per year, during the eighties (FAO 1993). Many people believe this to be a cause of great concern for a variety of reasons, among which potential loss of biodiversity and (local) climate regulatory functions, and the threat it implies to the way of life and survival of those millions of people who traditionally depend on forests for food and shelter. While tropical deforestation is mainly a matter of agricultural conversion and much less of commercial logging, logging does have an important impact on loss of forest habitat in temperate zones. Harvesting big trees in old growth forests in, for example, Canada and the U.S., is hotly debated.

The national and international conflicts over dwindling fish stocks are another example of a highly-valued resource under pressure. The estimated herring stock in the North sea in 1996 amounts to approximately 500 thousand tons, which is way below optimum biological and economic stock levels (Bjorndal 1988), and may be in the vicinity of the minimum viable population (the minimum stock that is required for long-run population survival for biological reasons). The North sea herring stock serves as an illustrative example for many more species at many more locations. With ongoing depletion of marine resources the way of life and incomes of many people and communities that historically depend on fisheries will be affected in the near future (see, e.g. Weber 1995).

While all of this may seem dramatic, it should be recognized that refraining from consuming an overexploited renewable resource for a long enough period of time implies (under some conditions) that this stock or population is allowed to recover. History provides us with some examples of successful resource recuperation after extensive periods of overexploitation. These examples include some severely depleted whale populations, harvested for meat, blubber and tradition (see chapter 6) and the African elephant, hunted in the seventies and eighties mainly for ivory (see chapter 5). A moratorium on commercial whaling and a trade ban on elephant products were implemented in 1986 and 1990 respectively, and triggered off recovery of the resources. On the other hand, one should not count on the regenerative abilities of nature. Recovery may be impossible or extremely slow in a deteriorated environment or if the stock is depleted to a level below the minimum viable population, as noted above. An additional problem may arise when recovery does not constitute mere biomass increment (as in commercial fisheries) but restoration of the original ecosystem and all of its complexities. For the latter reason, it is sometimes assumed that harvesting primary tropical forests and old growth forests in temperate regions basically boils down to exploiting a non-renewable resource: even though new trees will soon take the place of those harvested, it may take a long time before nature has shaped the ecosystem to its original form.

In this thesis I will discuss some economic aspects of renewable resource management, among which the rationale behind overexploitation and what economic wisdom suggests to overcome these problems. In general, economics is concerned with the question how to allocate scarce resources among competing uses, where such uses may be identical activities temporally separated. Traditionally, economists have regarded natural resources as factors of production with some characteristics (e.g. they must be "produced" and yield productive services over time), which make them rather similar to capital. More recently, it is recognized that *in situ* stocks of natural resources may also be sources of non-use values, or use values not directly related to exploitation, such as recreation. This has made economic models more complex, but richer. Before I turn to a discussion of some of these models in chapter 2, I will introduce in this chapter the concepts of efficiency, sustainability and social discounting. First, however, a brief introduction to the classical theory of exhaustible resource management will be provided in the next section. The reasons for this digression are twofold. First, conceptually, exhaustible resource exploitation provides a natural starting point for the economic analysis of renewable resource management. Second, many of the relevant conclusions and insights derived from studying non-renewable resource utilization spill over to renewable resource management.

1.2 Hotelling style models

Natural resources are commonly divided into resources capable of regenerating themselves, such as fish populations and trees as referred to above, and those that are not, such as oil and coal. The distinguishing feature here is the time required for reproduction of the resource. Non-renewable (or exhaustible) resources may be produced by geological processes measured at an evolutionary time scale, whereas renewable resources reproduce and grow at a rate that enables sustainable exploitation by humans. The distinction between renewable and exhaustible resources is blurred because the first category is easily over-exploited and depleted, such that actual renewal is no longer possible.

In addition to these two types of natural resources, there are resources such as solar radiation and the ability of the environment to absorb non-persistent pollution, that may be assumed constant flow resources. The main feature is that use of the resource at one particular point in time will not affect the amount that can be used in the future (that is, as long as possible critical loads are not exceeded). In contrast, with exhaustible resources, current exploitation implies foregoing benefits of exploitation at any future time, and with renewable resources, current exploitation of a unit of biomass rules out future harvesting of this unit and possible offspring or growth. This issue is at the core of the field of natural resource economics, where natural resources are considered capital assets. Allowing a renewable resource stock to grow is investing in future harvesting possibilities, whereas decreasing the stock size is interpreted as disinvesting. Rational decision making with respect to resource management requires an intertemporal comparison of marginal costs and benefits of exploitation. For the case of non-renewable resources, the economic approach will be discussed in this section.

In his seminal paper, Hotelling (1931) mathematically solved the question of how to allocate a stock of a non-renewable resource over time, such that social welfare from exploitation is maximized (see also Heijman 1991). After decades of negligence, renewed attention in the 1970s and 1980s, that was partially provoked by the oil crises and the linkages between resource extraction and environmental problems, further developed the depletion theme. This has resulted in more general models with less stringent assumptions.

As mentioned above, the implication of extracting and consuming a unit of a limited stock of a non-renewable resource today implies that less of the resource is available for future consumption. This means that current extraction involves an opportunity cost; that is the value that might have been obtained at some future date. Efficient use of exhaustible resources implies that this cost, which is usually referred to as *rent*, is taken into account. This implies that to maximize welfare, society should consume its stock of non-renewable resources in a way that is more conservative than consumption of ordinary goods, where marginal utility should equal marginal production costs.

The central issue of intertemporally efficient allocation of a stock of a non-renewable resource concerns the development of rent over time. Under the standard Hotelling assumptions, notably (i) a given and known stock X, (ii) extraction costs independent of the stock size, and (iii) marginal extraction costs independent of the extraction rate, the optimal control problem for society as a whole can be written as:

$$Max_{y(t)} \int_0^\infty \left[U(y(t)) - C\right] e^{-rt} dt \qquad (1)$$

subject to:

$$\dot{X}(t) = F(X(t), y(t), t) = -y(t) \tag{2}$$

and

$$y(t) \geq 0, \; X(t) \geq 0. \tag{3}$$

where $U(\cdot)$ is the instantaneous utility of consumption of the resource;[1] $y(t)$ represents exploitation (and consumption) at t; C is (fixed) exploitation costs; r is the discount rate; and $X(t)$, as mentioned above, indicates the size of the resource stock at time t. The dot over any variable (here over X) indicates a change in time, hence $F(\cdot)$ is a function that describes the change of the state variable over time. The current model specification without "growth" of the resource or new discoveries implies that the change of the stock is due to exploitation only. Finally, it is obvious that neither extraction rates nor the total stock *in situ* can take negative values.

The current value Lagrangian is defined as (see Chiang 1992; Kamien and Schwartz 1994):

$$L_c = U(y(t)) - C - \lambda(t)y(t) + \mu_1(t)X(t) + \mu_2(t)y(t). \tag{4}$$

This current value Lagrangian includes the inequality constraints and is an augmented version of the current value Hamiltonian [$H_c = U(y(t))-C-\lambda(t)y(t)$] that will mainly be employed in successive chapters. The necessary conditions for a maximum solution are as follows:

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[1]It is quite well possible to question conventional, mainstream economics for its naive assumption of maximizing simple utility functions or discounted social welfare functions. Taylor (1996) and many others have pointed out that there is no such thing as societal consensus on either environmental or economic questions - the degree of class and distributional conflict in any real economy rules out such accord: "The state itself may be autonomous, a creature of conflict-ridden civil society, or something in between. In none of these cases is the "objective" of state policy likely to resemble a neoclassical social welfare function" (Taylor, 1996). This matter is further developed in chapter 8 of this thesis.

$$U'(y(t)) - \lambda(t) + \mu_2(t) = 0 \tag{5}$$

$$\dot{\lambda}(t) = r\lambda(t) - L_{X(t)} = r\lambda(t) - \mu_1(t) \tag{6}$$

$$\mu_1(t)X(t) = 0, \ \mu_2(t)y(t) = 0, \tag{7}$$

and the equation of motion [i.e., eq. (2)] again. The subscript y or X indicates a partial derivative with respect to that variable. First, equation (7) is the Kuhn-Tucker conditions. For an interior solution [i.e., $y(t)$ and $x(t)$ positive], it follows from (7) that the Kuhn-Tucker multipliers μ_1 and μ_2 should be zero (Kamien and Schwartz 1994). In what follows in these introductory chapters it is assumed that an optimal interior solution exists, hence these multipliers are neglected.

Equation (5), the first condition of optimal depletion, is that the rent (or opportunity cost) $\lambda(t)$ equals marginal utility of current consumption plus a Kuhn-Tucker multiplier. $\lambda(t)$ is the co-state variable that reflects a shadow price; the change in the optimal value of the objective function corresponding to a small change in the constraint. The interpretation of this condition is that the Lagrangian is maximized if the rate of extraction is such that the net current gain in utility from extraction is just balanced by the discounted future losses.

The second optimality condition, equation (6), describes the efficient rent path over time. For an interior solution ($\mu_1 = 0$), it states that the shadow price should grow at the rate of interest, or equivalently that the present value of a unit extracted must be the same in all periods. This makes economic sense. Resources in the ground can be considered capital assets, and efficient use of assets implies that there can be no gain in shifting from one asset to another. Hence, the return must be the same for all. Therefore, rent should rise at the rate of interest in order for producers to be indifferent to the timing of extraction and in order for *in situ* reserves to be competitive asset holdings (Solow 1974).

The Hotelling rule thus implies that the shadow price increases exponentially over time. Hence, if demand is stable, exploitation declines monotonically over time and, ultimately, declines to zero.[2] Differentiating (5) yields $\dot{\lambda}=U''\dot{y}$. Combining this result with the Hotelling rule gives:[3]

$$\frac{\dot{y}(t)}{y(t)} = -\frac{r}{\eta} \tag{8}$$

where η is the elasticity of marginal utility of consumption (indicating the curvature of the utility function), which equals $-(yU''/U') > 0$. Hence, along an optimal depletion path satisfying the optimal conditions for optimal depletion, the rate at which consumption y falls over time is equal to the ratio of the discount rate and the elasticity of marginal utility of consumption (Heal 1993). The higher the discount rate and/or the lower the elasticity of marginal utility, the more unequal the distribution of the given stock over different generations.[4]

To check whether the maximum principle conditions are both necessary and sufficient for a maximum solution, the Arrow sufficiency theorem can be applied [see Chiang (1992) for a discussion, and Kamien and Schwartz (1994) for a proof]. This comes down to checking whether the maximized Hamiltonian [substitute the optimal values of the control variable y in the

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[2]But only asymptotically if the demand curve does not intersect the price axis at a finite price. Then exhaustion does not occur in finite time.

[3]Assume that $U(y)$ is strictly concave.

[4]The relationship between the interest rate and extraction may be not as simple as assumed in this model, because other than pure asset management considerations may play a role. In these models capital costs, capacity investment and adjustment costs are not explicitly included. As argued by Toman and Walls (1995) this is an unfortunate omission since resource industries are highly capital intensive. Lassere (1985) demonstrated that relaxing the assumption of fully malleable and costlessly, instantaneously adjustable capital may considerably alter the predictions of the standard model (Toman and Walls 1995). In particular, output may grow and rents may decline when extraction capacity is being built up. Capital costs can also affect marginal extraction costs. If capital is a necessary input in the exploitation process, high discount rates can well be expected to increase extraction costs and hence reduce extraction rates (Sweeney, 1993). Finally, Farzin (1984) developed a resource-substitute model that explicitly allows for the need of capital in both extraction of the resource and production of a substitute. He showed that the allocative effect of a change in the discount rate depends on capital requirements for the development and production of the substitute, on capital costs in resource extraction, and on the size of the resource stock.

(current value) Hamiltonian: $H^*(x,\lambda,t)$] is concave in x for any given λ (i.e., the second derivative of the maximized Hamiltonian with respect to x should be non-negative). When the objective function $U(\cdot)$ and the right hand side of the equation of motion $F(\cdot)$ are both concave in (x,y) and $\lambda>0$, then the current value Hamiltonian $U+\lambda F$ is also concave in (x,y), and the conditions of the maximum principle are sufficient for a global maximum. However, the (current value) Hamiltonian can be concave in x even if U and F are not concave in (x,y).

Explicitly solving for optimal depletion paths implies that starting and end-point conditions should be specified. For this purpose transversality conditions are employed (see Chiang 1992 for an overview of different conditions reflecting different assumptions with respect to the problem at hand). In this thesis and elsewhere in the literature on renewable resource it is often assumed that the terminal time can be fixed at infinity. This eliminates the transversality condition and allows the solution to reach a steady state (Hanley *et al.* 1997).

Hotelling demonstrated that the optimal rate of exploitation for firms with perfect foresight in a competitive industry is the same as the optimal rate of exploitation for a benevolent social planner. The reason is that the marginal utility curve applied by the planner is the demand curve that the industry faces. The market forces of supply and demand ensure that Hotelling's rule will apply. If, for instance, current supply were excessive, prices would go down and resource owners would be reluctant to supply until future periods, thereby restoring equilibrium (for details and intuition, see, for instance, Hotelling 1931; Solow 1974; Fisher 1981; or Hartwick and Olewiler 1986). Yet, there are at least three reasons to suspect why in practice optimal depletion may deviate for firms and society. First, the discount rate applied by firms and society as a whole may differ; second, extraction of resources may involve external effects; and third, property rights may be ill defined. I will return to these issues later.

The assumptions of the basic Hotelling model (constant marginal extraction costs, a fixed resource stock etc.) are very restrictive, and the implications of relaxing them have been explored by many researchers in recent decades. It is beyond the purpose of this section to discuss these extensions in detail. Instead, I will mention some major findings in a non-technical manner.

A first and obvious elaboration is incorporating the effect of cumulative extraction, such that extraction costs are a function of the *in situ* stock: $C=C(X)$ (e.g. Schulze 1974; Weinstein and Zeckhauser 1975; Levhari and Liviatan 1977; Devarajan and Fisher 1981; Sweeney 1993). In theory, C_X could have a positive sign (this could happen when resource owners "learn by doing", and gradually lower their marginal extraction costs as the stock is depleted), but most often it is assumed that C_X is negative. Hence, the smaller the *in situ* stock, the higher extraction costs, for example because costs rise when the mine gets deeper.

When a reduction in the stock has the effect of increasing future extraction costs, then holding the resource in the ground and not reducing the stock provides a dividend in the form of cost savings in the future.[5] This dividend equals the discounted sum of these increments to costs in all future periods, and is reflected in the necessary conditions for an optimum solution as an additional term in (5): $\dot{\lambda} = r\lambda + C_X$. Since $C_X < 0$, the optimum rent path is more gently sloped (but see Farzin 1992, 1993).

A second extension involves market power. For a variety of reasons, exploitation of natural resources often occurs under conditions resembling a monopoly (or oligopoly). The monopolist faces a downward-sloping demand function and will take the depressing effect of additional supply on prices into consideration. This implies that marginal revenue (*MR*) is not equal to price, and *MR* should grow at the rate of interest instead (Fisher 1981; Hartwick and Olewiler 1986). Consider a downward-sloping linear demand curve. The competitive industry moves up the demand curve and the monopolist moves up the *MR* curve, both satisfying their version of the Hotelling rule. Since the marginal revenue curve is steeper than the demand curve, the extraction profile of the monopolist should be more gently sloped than the extraction profile of a competitive industry. As a consequence, prices will rise more

[5]Hence even if the resource will never be fully depleted, for instance because environmental constraints curtail exploitation before exhaustion occurs such that current use does not impede future use and the scarcity component of rent is equal to zero, λ is still non-zero.

slowly in a monopoly than in a competitive industry. However, since the monopolist initially extracts less, the monopolistic price path starts at a higher level. If cumulative extraction is the same for the monopolist and the competitive industry, the price paths will cross after some time, and from that time onward the monopolist extracts more than the competitive industry and monopolistic prices will be lower.

Yet, it is not a general result that monopolistic extraction and resource conservation coincide, as proved by Stiglitz (1976), Dasgupta and Heal (1979) and Lewis *et al.* (1979). Since $MR(y)=P(y)(1+1/\epsilon)$, where ϵ is the elasticity of demand, it is obvious that the answer to the question in what way monopolistic exploitation deviates from the social optimum is determined by the nature of demand. It depends critically on the development of ϵ over time. When demand changes over time, becoming less elastic, the monopolist will take advantage of this shift, and will restrict supply in future periods. If restricting future supply results in increased supply in early periods (as is obviously the case in a two-period model with complete exhaustion, considered by Stiglitz 1976, but see Tullock 1979), the monopolist will act as an anti-conservationist.[6] The monopolist may also accelerate depletion if the elasticity of demand ϵ increases with the rate of production, for instance because sufficiently low prices attract bulk users at the margin to switch from substitutes. Then as y varies over time, ϵ will vary accordingly, even though ϵ itself is not a direct function of time. For a constant elasticity of demand curve, Stiglitz (1976) demonstrated that extraction for a competitive industry and monopoly is identical.

Third, Hotelling already recognized the importance of exploration, and discussed the implications of exploration in the context of uncertainty and market failure.[7] Society is not endowed with reserves, but must develop them

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[6]Devarajan and Fisher (1981) remark that the assumption of decreasing elasticity over time is unlikely. As time passes, new substitutes are developed and the elasticity of demand will probably increase, instead of decrease.

[7]He concluded that, if whoever finds a deposit can file a claim, which will result in excessive levels of exploratory activity. Moreover, he asserted that spillovers of geological information (i.e. firms benefit from exploratory activities of their neighbours because it helps

through the process of exploration instead. By exploration the reserve base is increased, hence in some periods production may gradually increase, rather than steadily decrease as in the Hotelling model.

Consider a simple model where in addition to extraction costs an exploration cost function $EC(z(t))$ is included, where $z(t)$ represents the finds in period t. A heroic assumption could be that exploration is a deterministic process. Hence, society has two variables to control: $y(t)$ and $z(t)$. In addition to decisions about exploitation, it must decide at a target level of new finds $z(t)$. With stock effects in extraction costs, $C(X)$ with $C_X < 0$, the new finds hold exploitation costs down through their influence on $X(t)$. On the other hand, they bear a cost in themselves. Optimally, exploratory activity is chosen to build the reserve base up to a level that reduces extraction costs and then is adjusted over time so as to trade off cost savings from postponed exploration with savings from lower extraction costs and revenue gains from greater total production (Pindyck 1978).[8] If production costs were independent of reserves, society would postpone much of its exploratory effort (thereby discounting its costs) and maintain no reserves.

Solving a model with exploration costs yields as a result that the rent of the resource λ is equal to the cost of finding another unit to replace it. This implies that the marginal exploration costs may be used as an estimator of the rent, which is fortunate since direct data on resource rents are often hard to find (Fisher 1981). A nice result of this model is that in theory it can explain why prices do not rise exponentially, as in the basic Hotelling model. The change in price depends on the changes in extraction cost and rent. When new finds exceed extraction in a certain period, the total stock increases, which, under the assumption of an aggregate cost function, implies that extraction costs may decrease. This effect will mitigate the effect of the rise in rent, and

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them to locate their own deposits) result in excess profits. Peterson (1975) demonstrated that in the latter situation sub-optimally low levels of exploration may be conducted because every firms hopes to be a free rider.

[8]To me, this is not very satisfying, because it is not clear what the effect is of new discoveries in one part of the world on extraction costs in another. After all, the total stock is divided over several small resource deposits, and the characteristics of those separate deposits determine the extraction cost function. (Also, if a large stock reduces extraction costs for some physical reason, it is unclear why it would not have this effect when it is not known.)

possibly exceed it, so that price will actually fall. Hence, if the initial reserve endowment is small (provoking a lot of exploration), the price profile of the resource can be U-shaped, rather than steadily increasing as in the Hotelling model.[9] See Slade (1982) for a similar result and empirical results with respect to ongoing technological progress.

Lastly, a lot of effort has been put into incorporate uncertainty into the standard model. Uncertainty can relate to, for example, future demand or technology (e.g. Weinstein and Zeckhauser 1975; Lewis 1977), property rights (e.g. Long 1975), or the size of the stock (e.g. Gilbert 1977; Loury 1978; Pindyck 1980; and Arrow and Chang 1982). Some sources of uncertainty (for example, with respect to future tenure rights) imply that a rational resource owner (risk averse or not) adds a risk premium to the discount rate and accelerate depletion.

However, this is not a general result. Uncertain demand in the form of a shifting demand curve may have no impact on risk-neutral resource owners. For risk-averse owners, on the other hand, depletion may be accelerated or postponed, depending on the characteristics of uncertainty. If uncertainty is positively related to the distance in time (such as with prices following a random walk), the resource owner is less certain about prices further in the future, and will therefore tilt exploitation to the present. If uncertainty is related to the amount supplied (revenues are defined as price times quantity, so that revenues will be more uncertain when quantities supplied are big), the risk-averse owner will typically try to spread his supply more evenly over time. Hence, from early periods, when supply is relatively large according to Hotelling's model, to future periods where supply and variations in returns are relatively small.

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[9]Of course the analysis is much more complicated with uncertain exploration (see also below). Arrow and Chang (1982) write: "the stock of the resource is very far from known. Hence, new discoveries yielding upward revisions of estimated reserves change the basis of calculation for the Hotelling rents. The predicted rise at the rate of interest is offset by repeated downward revision of the initial price in response to changing supply conditions." New discoveries will reduce the future price of already known deposits. This will provoke an increase in exploitation, causing a sudden price fall.

Pindyck (1980) demonstrated that risk-neutral and risk-averse resource owners alike should increase exploitation when exploration is modelled as a stochastic process. With (non-linear) stock effects on extraction costs, equally big upward and downward deviations from the mean value in a stochastic model do not outweigh each other, which is a consequence of Jensen's inequality. In Pindyck's model, future marginal extraction costs were higher, which resulted in faster depletion than along the certainty equivalent path.

Efforts to empirically test the theory of efficient natural resource exploitation have been few. Researchers have attempted to work with prices[10] (e.g. Barnett and Morse 1963; Smith 1978; Heal and Barrow 1980; Slade 1982), or exploration cost (Fisher 1981; Pesaran 1990) or price minus marginal cost, (Stollery 1983; Farrow 1985; Halvorsen and Smith 1991) as an indicator of rent. In general, it can be concluded that empirical support is inconclusive, with some apparent refutations of the theory (e.g. Farrow 1985; and Halvorsen and Smith 1991) and some consistent results (e.g. Stollery 1983).

1.3 Sustainability and efficiency

As mentioned in the previous section, efficient exploitation maximizes the present value of welfare for society, but this does not necessarily lead to equitable solutions.[11] Efficient exploitation may be Pareto efficient (i.e., such that it is impossible to improve welfare of one generation without harming welfare of other generations), but it is by no means the only Pareto efficient allocation (see Page 1977) and it has been criticized by many as ethically questionable because of the large weight it attaches to the welfare of current generations. Given a positive discount rate, the interests of distant future

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[10]It should be recognized that not all of these studies aimed to test the theory of efficient exploitation, as some were investigations of increasing scarcity.

[11]As noted in 1.2, in a simple model consumption should fall at a rate equal to the ratio of the discount rate and the elasticity of marginal utility of consumption.

generations hardly affect current decision making. An important concept in natural resource economics, in addition to efficiency, is sustainability. This may be interpreted as an intertemporal equity principle.[12] Non-sustainable exploitation is often referred to as living off capital rather than income, implying that future consumption opportunities are affected.

The major part of modern mainstream economics in the field of natural resources is firmly rooted in the efficiency concept, and so is most of the content of this thesis. Unfortunately this implies to a certain extent digression from, without in any way minimizing, the concept of sustainability [although it could be argued that preferences with respect to "sustainability" are easily incorporated in mainstream economic models, either in the objective function or as (an) additional constraint(s)]. In the remainder of this section I will provide an introductory review of sustainable resource management, and will also discuss the relation between efficiency and sustainability. The extent to which these key concepts should be considered complementary or conflicting depends primarily on the subjective interpretation of what the concept of sustainability actually means.

There is little certainty about what sustainable solutions to natural resource models exactly consist of. Economists, ecologists and philosophers have interpreted the concept differently, and even within a certain profession opinions may be divided greatly. A well-known definition of sustainable development that is applicable to resource modelling is provided by the Brundtland report (1987): "*Sustainable development is development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs*". While it is certainly not the aim of this thesis to deal with details and differences with other definitions [see Pezzy (1989) for an overview, and also Victor (1991) who attempts to construct different "schools of thought"], it is of interest to note that it makes a big difference whether "meeting future needs" is in terms of (consumption based) utility on the one hand, or in terms of a broader set of environmental functions provided by natural resources (e.g. a variety of regulatory functions and

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[12]Turner *et al.* (1994) mention that some definitions of sustainability imply both intergenerational- and intragenerational equity concerns.

existence values) on the other. The first interpretation, combined with the perception that the natural environment can be regarded as one of many fungible assets, implies that a deteriorated environment and depleted stocks of some resources can possibly be compensated for by intergenerational transfers of capital and technology. This interpretation of sustainability is often referred to as weak sustainability. The latter interpretation, especially combined with concern for ecological integrity, will usually lead to the conclusion that use of resources must be curtailed and investments in creation of substitutes should be enhanced [see Daly (1990) for some operational principles]. The view that (some) elements of the natural capital stock can not be substituted for by man-made capital (implying that these stocks should not be allowed to deteriorate), is often called strong sustainability.

Two central themes, sketched above and intimately linked, in the concept of sustainability are the nature of the current generation's responsibility to future generations and the degree of substitutability between natural capital and forms of social capital, such as physical capital and knowledge. Both issues have proven to be interesting subjects of debate, of which lucid summaries are provided by Toman (1994) and Toman *et al.* (1995).

With respect to the first issue, i.e., the nature of the current generation's responsibility to future generations, opinions are divided. Howarth and Norgaard (1990) have argued that intergenerational equity can be achieved when a fair allocation of property rights over current and future generations is realized. This would provide an alternative for the efficiency criterion that would be very similar to the Rawlsian "maximin" criterion.[13] Modified versions of this criterion that allow for some trade-off between equity and welfare maximization have been proposed, but others have been quick to point out that assigning rights to potential future individuals leads to many difficulties, both in theory and in practice. Some economists would like to stray away further from efficiency considerations, and claim that individ-

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[13]The maximin approach consists of an objective function plus a rigid side constraint of justice. Alternative specifications have attempted to incorporate sustainability concerns into the objective function itself, for instance, by adding a positive but finite weight to future, satisfying a sustainability condition (Toman *et al.* 1995). Basically this implies an approach very similar to conventional efficiency analysis with extra arguments in the objective function, allowing trade-offs between intergenerational concerns and other social goods.

ualistic preferences need not be the basis for society's decision making with respect to natural capital at all. The reason is the existence of overriding social values and societal concerns embodied in varying degrees in all members of society in their "roles" as citizens, rather than consumers. According to this organicist or stewardship perspective, sustainability involves an obligation to not just future individuals, but to the species as a whole and the ecological systems that surround it.[14]

In addition to considerations concerning the obligation of current generations towards future ones, the degree of substitutability between social and natural capital determines to what extent current exploitation can be justified. Here too, opinions are divided. Under the assumption that natural resources are fungible, sustainability in the sense of non-declining consumption is easily satisfied. According to this view, depletion of resources, biodiversity loss and climate change may be perfectly compatible with sustainable development. Provided that the elasticity of substitution between natural and social capital is high enough (or if technical development is sufficient to offset the reduction of environmental services), the consumption of future generations need not decline when resource stocks run down. Many analyses of economic growth in the face of limited resources are based on the Cobb-Douglas production function, but unfortunately this does not allow an analysis of the limits of substitution among inputs (e.g. labour, capital and natural resources) because the elasticity of substitution with this specification is by definition equal to 1. For example, Solow (1974), Dasgupta and Heal (1974) and Stiglitz (1974) have explored the possibilities of constant or growing consumption with depletable resources as a factor of production and with technology exhibiting constant elasticity of substitution.[15] Daly (1990),

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[14]However, as pointed out by Toman (1994), some of humankind's "most cherished economic, political and other social institutions derive fundamentally from giving high respect to individual rights".

[15]However, the realism of some of the conditions obtained, for example, that the economy can run on an extremely small and diminishing quantity of energy, has been questioned in light of physical laws. Also, some authors have argued that the potential for capital to substitute for resources so that output is maintained indefinitely despite the declining stock of resources is undermined because capital is not a gift from nature but rather made from resources (e.g., Victor 1991).

among others, is very sceptical about substitution possibilities between natural and man-made capital. He thinks that natural and man-made capital are in the first place complements in production, with the latter form of capital as the "agent of transformation" and the first as "that being transformed" (Daly 1990).

The conventional neo-classical framework also emphasizes the role of technological progress. Sometimes a backstop technology (providing unlimited services at a constant cost) is simply assumed to be out there (Victor 1991). Given sufficient opportunities for technical progress and substitution [in the optimistic words of Hartwick (1996): "*the bounty of past technological progress seems to assure us of future additional large dividends*"], Hartwick (1977, 1996) demonstrated that sustainable consumption can be achieved when resource rents from exhaustible resources are invested elsewhere in the economy, such that depleting a stock of natural resources is compensated for by accumulation of a stock of physical capital.[16] However, for some functions, especially the so-called "life support functions" as maintenance of hydrologic and nutrient cycles, it is not clear whether compensatory investments are feasible or ethically defensible. Then, a so-called "strong sustainability" approach may be justified, where net damages to (aspects of) the stock of natural capital should be non-positive (Pearce and Atkinson 1995).[17]

It can be concluded that the choice between different interpretations of the concept of sustainability is a matter of believing in (i) what kind of world we should live in, (ii) technological progress and (iii) possibilities of substituting man-made capital for natural capital in production. It should be realized

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[16]For this condition to hold, resource rents should not be measured with their market prices (which could be too low with excessive depletion and corresponding low prices) but with shadow prices reflecting the sustainability constraint.

[17]Other reasons provided by Pearce and Atkinson (1995) why the principle of strong sustainability may be more appropriate than the weak equivalent (or a combination of the principles is required) are (i) that the total value of ecosystems is more than the sum of individual functions, (ii) that uncertainty exists about aspects of natural capital, and (iii) "loss aversion" with respect to natural capital, so that individuals are interested in maintaining the status quo (it has been argued that loss aversion could account for the differences in willingness to pay and willingness to accept for similar environmental assets, as measured in various studies).

that in reality trade-off decisions with respect to production and conservation are taken in the context of uncertainty and irreversibilities, which further complicates matters.

Depending on the interpretation of the concept of sustainability, efficient solutions may be considered more or less compatible with sustainable development. For example, maximizing profits in a fishery may result in a steady-state fish stock and associated harvest level that can be supported infinitely. (In theory, infinitely exploiting a stock of fish is possible at all stock levels ranging from the minimum viable population to the stock size where the carrying capacity of the environment for the stock in question is reached.) However, whether all of these stock levels would qualify as "sustainable" or not is ambiguous and depends in part on the interpretation of the concept as discussed above. When efficiency and sustainability are considered incompatible, Randall (1986) and Toman (1994) have suggested to introduce a kind of extended safe minimum standard approach to determine which concept should prevail: small scale, easily reversible impacts may be addressed by conventional efficiency theorems, whereas large scale, irreversible impacts should be subject to moral imperatives for resource and ecosystem protection.

1.4 The social discount rate and government intervention

While interpretation of the concept of "efficiency" is in itself not controversial (in contrast to interpretation of the meaning of sustainability, as discussed above), there has been a lot of debate in the literature about the correct discount rate that should be applied, i.e., the rate at which society should be willing to substitute present consumption for future consumption at the margin. Broadly speaking, there is a school of thought that advocates use of the social opportunity cost (SOC) of capital which is based on the productivity of capital,[18] and a second school advocating the social rate of time

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[18]Due to corporate taxation, the social discount rate according to this school should be higher than the market rate of interest. Baumol (1968) is not particularly worried about adverse implications for investments in the future: Since per capita income is growing, he regards

preference (SRTP). The latter discount rate is likely to reflect a measure of impatience and a social judgement of the effect that future generations are probably richer, such that the marginal benefit of additional revenues is lower.[19] In a world without taxes, risk or uncertainty and with perfect capital markets, the two approaches coincide (Pearce and Turner 1990; Clark 1990).

When the government is considered a trustee for unborn generations, a more or less "ethical approach" with (very) low discount rates could be validated. Some authors have argued that impatience should have no implications for public policies, especially since impatience may be inconsistent with lifetime welfare maximization. Others have questioned the argument of diminishing marginal utility of consumption by stating that the wealth of future generations will (at least in part) depend on the state of the environment, so that real consumption (and rates of return on investments) may not grow or even fall when we degrade the environment. For an overview of more criticisms, see Solow (1974), Heijman (1991) and Hanley and Spash (1993). These arguments suggest that the social discount rate should be lower than the SOC or SRTP, but it is not immediately clear how much lower. Pearce and Turner (1990) argue that "adjusting" discount rates to achieve intertemporal equity is inefficient and clumsy, for example, because the relation between discount rates and resource degradation is complex (see section 1.2). Serageldin (1993) adds that it is not clear why society should give more priority (by applying separate, lower discount rates) to environmental protection than to health, education or family planning projects. An alternative way to circumvent some of the concerns mentioned above is to integrate a sustainability requirement into the economic model, more or less along the lines discussed in the previous section. Depending on the view of the policymaker, the aim could be to avoid depletion of the overall capital stock (Hartwick 1996) or that the more narrow subclass of natural capital should be maintained intact (Daly 1990).

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redistribution of income from present to future generations a "Robin Hood activity, standing on its head".

[19] This effect can be incorporated by adding a term where the elasticity of the marginal utility of consumption is multiplied by the rate of growth of real consumption per capita.

Ultimately, the specification of the social discount rate appears to be a political or ethical choice, partly depending on the government's value judgement on the importance of preferences for policy-making. Howe (1979), among others, is sceptical about a benevolent role of the government and remarks that "*it is not clear that the government's effective time horizon is any longer or any more responsive to future conditions than that of private enterprises, large amounts of rhetoric notwithstanding. (..) The next election is always around the corner*".

In reality, actual exploitation of many natural resources is carried out through private firms, probably applying discount rates that exceed the desired social discount rate (Heijman 1991). This implies that to maximize welfare for society, the government should intervene and, most probably, slow down exploitation. Government intervention is also appropriate when well-defined property rights for resources are lacking, or if exploitation and consumption of a certain resource brings about external effects, such as pollution. Intervention can take many forms, for instance (i) taxation, which can shift depletion to both the future and the present, depending on the tax base chosen;[20] (ii) price regulations which have similar effects (see Kalt and Otten 1985), and (iii) direct intervention through joint ventures, et cetera.

Typically, a host of objectives may be served through intervention, of which correcting for market failure is only one. Examples include generation of local employment, macro-economic issues, distribution of benefits (transfers of rents), and security of supply. An important aspect of government intervention is the private firm's subjective assessment of its permanence. When intervention is aimed at postponing depletion by implementing high severance taxes, firms may decide to temporarily cease exploitation altogether to circumvent tax payments, when they believe that stringent regulation will be relaxed in the (near) future. Alternatively, subsidy schemes that are

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[20]For instance, severance taxes (a fixed charge per ton of output) will shift depletion to the future as discounting will reduce the present value of the fixed nominal charge, whereas property taxes (a tax on reserves in situ) will shift extraction to the present for obvious reasons. In theory, profit taxes and resource rent tax will not affect depletion patterns, although the latter may affect decisions with respect to exploration and development of resource bases. For more information, the reader is referred to, for example, Conrad and Hool (1981) and Heaps (1985).

considered temporary may provoke enhanced exploitation. These issues have been discussed extensively in economic literature.

1.5 Organization

In chapter 2, I describe the basics of renewable resource management, based on a brief literature review. Key issues are (i) optimum stock size (i.e., since harvesting a renewable resource represents disinvesting in the stock and refraining from harvesting implies investing, how large should the stock be to maximize utility for society?), (ii) approach dynamics (how fast should society try to reach the steady state?), (iii) extinction (should society try to sustain a positive stock level for the renewable resource in question, or is utility maximized when the stock is exhausted?), (iv) property rights (what different regimes of property rights exist, and what do they imply for resource management?), and (v) tropical deforestation (what are the characteristics of the deforestation process? Does it pose a problem? What can be done to combat it?). These issues return in more detail in subsequent chapters. Chapters 3 through 11 are (slightly revised) versions of papers I have co-authored during the past 3 years.

Chapter 3 focuses on tropical deforestation in more detail. As will be explained in section 2.6, the distinction between primary, undisturbed forests and secondary, or selectively logged, forests is crucial for understanding deforestation. The reason is that there are significant differences in accessibility for shifting cultivators. With a model that explicitly recognizes the transformation of primary forests into secondary forests, the validity of two widely held presumptions is theoretically examined. First, we examine the claim that encroachment is necessarily detrimental for nature conservation. Second, we investigate whether high discount rates accelerate deforestation, as conventional wisdom implies. Chapter 4 deals with deforestation too. In this chapter the usefulness of providing international transfers to developing countries to promote conservation of tropical forests is examined. This chapter is an extension of a model by Barbier and Rauscher (1994), but relaxes the restrictive assumptions of risk neutrality and deterministic prices.

In chapter 5 we construct a fairly standard renewable resource model to study the impact of the trade ban on ivory on the optimum elephant population as perceived by the government of African countries. The trade ban has halted the massive slaughter of elephants that occurred during the seventies and eighties, so it receives wide support from environmentalists and the general public in Western countries alike. However, the rationale of the ban has been disputed, especially by government officials in countries with large herds of elephants. Since a trade ban implies altering the investment and disinvestment considerations made with respect to elephants as an asset, there is scope for economic analysis. Questions that are addressed are the following: (i) will the optimum elephant population (as for example defined in section 2.2) increase or decrease as a result from the trade ban? (ii) under what conditions may banning trade be detrimental for nature conservation, and (iii) will the apparent success of the trade ban in halting (illegal) killing of elephants sustain in the future?

Chapter 6 follows naturally from the previous chapter, where the *performance* of a trade ban is explored. In chapter 6, the central issue is whether, from an economic point of view, trade bans should have been *implemented* in the first place. The case examined relates to commercial (minke) whaling, which has been subject to a commercial moratorium since the mid-1980s. We analyze whether this moratorium on whaling is economically efficient by comparing the intertemporal marginal costs and benefits of strict conservation. We also assess whether the current stock of minke whales is too small or too great, and compare the outcomes of a dynamic model with the findings of a simple static approximation, as recently proposed by some authors.

Chapters 7, 8 and 9 are about fisheries policies. Chapter 7 goes back to the property rights problems that will be discussed in section 2.4. In the mid-1970s, many countries declared sovereign rights within a 200-mile zone. This implied that within this exclusive (fishing) zone, a national government or supra-national organization as the European Union, was judicially able to formulate fisheries policy. Prior to the declaration of sovereign rights, fishing occurred under conditions of open access. This implies that every individual fisher had an incentive to harvest at full force [or, in the words of Neher (1990), every fisher had an incentive "*to rape, ruin and run*"]. As mentioned before, refraining from harvesting is an act of investment, and under conditions of open access, nobody is willing to undertake this investment because surely somebody else will reap the benefits. In chapter 7 we empirically try to address the question whether declaration of the exclusive zones and formulation and implementation of subsequent policies has had an impact on resource use.

Chapter 8, on the other hand, is a more formal theoretical model of fishing policies. We relax the restrictive and unrealistic assumption that governments maximize welfare for society (see Howe's quote in the previous section), and explore the consequences of a political preference function approach, based on interest group lobbying (think of pressure by fishers, consumers and labourers in the sector) for optimum stock size. Hence, it is recognized that formal declaration of sovereign rights need not result in socially optimum fishing policies because powerful and well-organized groups in society may have different interests. Exclusive zones are therefore a *necessary*, though certainly not a *sufficient*, condition for optimum management.

In chapter 9 we discuss an instrument that could be used to facilitate implementation of quota cuts and other management instruments that have the potential to meet resistance from the sector. We demonstrate how a futures market of fishing rights reduces the risk for fishermen, which increases utility of risk-averse fishermen. This implies that there is scope for cutting individual quotas, without making the fishermen worse off (that is, compared to their initial utility level).

Chapters 10 and 11 are based on forestry models that incorporate uncertainty. In chapter 10 we discuss and apply fuzzy logic to deal with a land use problem in British Columbia. Allocation of land to various purposes may require processing of data that are far less than perfect. This could relate to prices (for instance, there are no clear prices for many environmental functions) or technical information, which is sometimes sparse and unreliable. Also, preferences of policymakers may be vague (i.e., "*expand wilderness protection*", "*revenues from timber harvesting should be high*", or "*forest employment should preferably be maintained*"). We compare fuzzy and classical decision models, and evaluate the outcomes in light of the political decision-making process in British Columbia, which relies on consensus seeking interest groups.

Chapter 11 deals with another source of uncertainty, namely stochastic prices with a known distribution. With aid of a dynamic programming model we determine reservation prices for thinning and harvesting in Dutch forestry. We compare profitability and timing of exploitation for the strategic forest owner who tries to exploit upswings in prices to increase profitability, and the forest owner who focuses on expected values and simply adopts Faustmann's criterion with expected prices instead (see section 2.6). Chapter 11 stands out from the other chapters because dynamic programming is employed instead of optimal control, and because it represents a so-called closed loop problem as opposed to an open loop problem. In the latter class of problems a resource owner decides on an optimal exploitation scheme once and for all and is committed to this scheme in all future periods, whereas in the first class of problems (also called feedback problems) the resource owner will continue to update expectations and act accordingly. Lastly, in chapter 12 conclusions are given.

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CHAPTER 2

RENEWABLE RESOURCE MANAGEMENT

2.1 Introduction

In this chapter I present and discuss some of the basic concepts of the economics of renewable resource management. Although there are many parallels with the theory of exhaustible resource exploitation, as discussed briefly in the previous chapter, there are some notable distinctions. The regenerative capacity of the resource and the associated possibilities of steady-state harvesting are obviously the most striking differences. The basic theory of renewable resources is outlined in sections 2.2 to 2.4. Although the specification of growth and production functions may be (quite) different for different resources, the general insights spill over to fishery, wildlife management and forestry models alike. In section 2.5, the basic theory will be applied to typical fisheries issues, and in 2.6 the focus will be on forestry problems.

Fishery economics is different from the economics of forestry, because of, for instance, the prominent role of property rights (although this issue may also play a role in some deforestation problems in developing countries) and search costs: fishermen have to go out and find fish. Since search costs are possibly influenced by the size of the wild stock, there may be an incentive to maintain fish populations at relatively high levels. Alternatively, typical forestry problems are the optimum rotation of a single-aged stand (where it is recognized that the Beverton-Holt approach in fisheries management is rather similar conceptually) and the difficulties associated with multiple use values and multiple products. An additional difference between fisheries problems and the classical forestry issue is that fishery models typically attempt to determine steady-state stock levels where harvesting equals increment, whereas harvesting in traditional forestry models involves both increment and the standing stock.

Basic understanding of the biology of (single) renewable resource stocks is necessary to analyze management of these resources (although it could equally validly be argued that the biology of ecosystems and food webs needs to be better understood for proper management of natural resources). Two simplified models[1] that describe fertility, mortality and growth characteristics of a population are the logistic growth function and the Gompertz function.[2] The first, suppressing time notation, is given as:

$$G(X) = \gamma X\left(1 - \frac{X}{X_{CC}}\right) \tag{1}$$

and the latter:

$$G(X) = \gamma X \ln \frac{X_{CC}}{X} \tag{2}$$

where γ is called the intrinsic growth rate of the resource and X_{CC} is the carrying capacity of the ecosystem for the population in question. The growth curve of most growth functions has the shape of a bell or parabola (see Figure 2.1 for an illustration of a logistic growth curve). When the stock is small, growth will also be modest, even under the most favourable conditions. In terms of fisheries, the reason is that there are few female fish to produce offspring. Growth of the stock will also be small when the population is close to its maximum size, i.e., when it has filled its niche in the ecosystem. When

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[1]With respect to fisheries, these biological models both fit in the so-called Schaefer approach. In contrast, the Beverton-Holt approach mentioned above deals with separate cohorts or year classes that are followed through time. Harvesting separate cohorts requires a (currently almost not available) knife edge selectivity in harvesting techniques. Since modelling the economics of multi-cohort fishing becomes complicated (Clark 1990) the Beverton-Holt model has not been very successful as a foundation for economic models (Munro and Scott 1985), but see Flaaten and Kolsvik (1996) for a nice application.

[2]Working with the discrete time analogues of these equations (i.e., working with difference equations) may produce chaos, depending on the value of the parameters used. Duarte (1994) provides a discussion of the complex interaction in unstable dynamic systems between policy measures and harvesting on the one hand and stock behaviour on the other: stationary steady-state was only one of many different possible outcomes of bioeconomic equilibrium.

the stock has expanded to X_{CC} it has reached its carrying capacity, and the ecosystem is not able to support further growth. Possible reasons are food scarcity or spreading of disease because of high densities.

A more complete illustration of the growth function would include the concept of *minimum viable population* (MVP). This corresponds to the population level below which, without intervention, the population would decrease and eventually approach zero. The reason may be genetic deterioration, but also insufficient defense against predators. Minimum viable population requirements could be incorporated (but it is not, for consistency with the formal analysis presented below) in Figure 2.1 by allowing for a non-concave interval with negative values for $G(X)$ at low stock sizes, such that the growth function intersects the horizontal axis at X_{MVP}, see Clark 1990).[3] Note that X_{CC} is a stable equilibrium while X_{MVP}, the minimum viable population size, is unstable (i.e., when for some reason the actual population deviates from the equilibrium population, the equilibrium will not be restored without intervention).

Besides the growth function, the yield or production function $y(t)$ is needed to describe the equation of motion of the optimal control model. A common and general production function (especially in fisheries and wildlife modelling; incorporation of a biological stock term X in forestry models is unusual, but this distinction is harmless for the purposes of this section) is as follows:

$$y(t) = qE(t)^{\alpha} X(t)^{\beta} \tag{3}$$

where q is a (possibly species dependent) catchability constant, and E is effort, a measure for the inputs devoted to the yield process.[4] Now, the

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[3]The growth function is called *purely compensatory* if $X_{MVP} = 0$ and $G(X)$ is strictly concave, as in Figure 2.1. It is called *depensatory* if $G(X)$ is initially convex and later concave (note that this does not correspond to the functions described above). Finally, the function is said to exhibit *critical depensation* if $X_{MVP} > 0$, and $G(X)$ initially convex and later concave (Conrad and Clark 1987).

[4]An alternative specification sometimes applied in fisheries economics is the exponential- or Spence production process: $y(t) = X(t)(1 - e^{-qE(t)})$ (see Amundsen *et al.* 1995).

equation of motion for the renewable resource problem, while suppressing time notation, is given as:

$$\dot{X} = G(X) - y. \tag{4}$$

Sustained yield or harvesting is defined as harvesting with X, y and E constant over time, i.e., $y = G(X)$. In the Schaefer fisheries model (1957), α and β of the production function (3) are set at one, and the growth of the stock is described by the logistic function. Then in equilibrium:

$$y = qEX = \gamma X\left(1 - \frac{X}{X_{CC}}\right) \tag{5}$$

solving for X gives:

$$X = X_{CC}\left(1 - \frac{qE}{\gamma}\right) \tag{6}$$

which implies:

$$y = qEX = qEX_{CC}\left(1 - \frac{qE}{\gamma}\right) \tag{7}$$

This sustained yield-effort function is parabolic. If $qE > \gamma$, the resource stock is asymptotically depleted and eventually y and X approach 0 (Conrad and Clark 1987).

As is readily observed from Figure 2.1, rapid growth of the stock is possible with moderate stock sizes. Growth is at its maximum when $G'(X) = 0$. The harvest associated with this phenomenon is called maximum sustained yield (MSY), and the stock size corresponding to maximum sustained yield is depicted in Figure 2.1 as X_{MSY}. For the logistic growth function, $G'(X) = 0$ at $X_{MSY} = X_{CC}/2$, and for the Gompertz function at $X_{MSY} = rK/4$. Harvesting $G(X_{MSY})$ at stock levels of X_{MSY} has been promoted for many years by fishery and forestry ecologists alike; it was considered good policy to search for the largest harvest that can be sustained forever. However, for a long time economists have argued that X_{MSY} is seriously flawed as a management concept. Neher (1990) puts forward three different arguments tailored to a fishery but applicable to forestry as well. First, although perhaps least important in this respect, the steady-state analysis does not give any insight into the dynamics of approaching the optimal stock size. A trade-off arises between adjustment costs and bridging time that is overlooked in the ecological model.

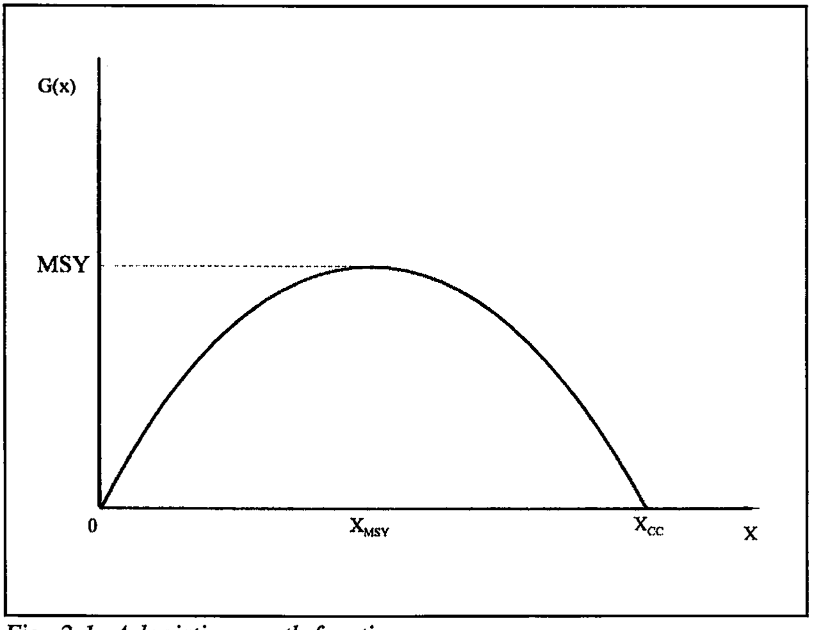

Fig. 2.1: A logistic growth function.

Second, prices and costs are disregarded in the X_{MSY} concept. As mentioned above, harvesting costs are likely to be influenced by stock size because it is easier to find fish as there are more of them around (see 2.5.1 for a more elaborate discussion). Since excessive harvesting costs are a waste of productive resources, it may be worthwhile to thicken the stock by accepting smaller harvests, and arrive at a steady-state stock size to the right of X_{MSY}.[5]

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[5]Straightforward multiplying by the price of fish means that $G(X)$ of Figure 2.1 can be transformed to a revenues curve. Next, assume that marginal harvesting costs can be represented by a declining function of stock size in the same figure. Then the optimal stock size will be the stock size where the slope of the revenue curve is equal to the slope of the cost curve (in that

Third, discounting is ignored in the maximum sustained yield concept. By the harvesting process, *in situ* renewable resources can be transformed into money in the bank. Since both money in the bank and, for example, fish in the sea are productive [the first grows at the interest rate r, the latter produces offspring with an average rate of return of $G(X)/X$ and a marginal rate of return $G'(X)$], they can be compared as assets. Disregarding harvesting costs for the time being and simply assuming that y is a control variable, the problem can be represented in a simple model as follows:

$$Max_{y(t)} \int_{o}^{\infty} P(t)y(t)e^{-rt}dt \tag{8}$$

subject to:

$$\dot{X}(t) = G(X(t)) - y(t) \tag{9}$$

With the necessary first order conditions for an optimum solution (while suppressing time notation):

$$P - \lambda = 0 \tag{10}$$

$$\dot{\lambda} = (r - G'(X))\lambda \tag{11}$$

Combining (10) and (11) yields:

$$\frac{\dot{P}}{P} + G'(X) = r \tag{12}$$

Hence rising prices ($\dot{P} > 0$), for instance due to changing preferences, encourage conservation as it is a motivation to build up future stocks. The intuition is as follows. Along an optimal path, (12) must hold. Given a certain (exogenous) value for r, the higher $\dot{P}/P$ the lower the required value for $G'(X)$ to balance the LHS and RHS of (12). Since $G(X)$ is concave, lower values of

---

case, marginal harvesting costs are equal to marginal benefits). This point of tangency is to the right of X_{MSY} (see for instance Pearce and Turner 1990; Tietenberg 1994).

$G'(X)$ correspond with greater stock sizes. Similarly, higher discount rates (for a given value of $\dot{P}/P$) imply that $G'(X)$ should increase, which can be achieved by reducing X. The relative strength of the countervailing effects of discounting on the hand and harvesting costs and increasing prices on the other hand, determines whether optimal stock size according to economists is bigger or smaller than maximum sustained yield levels.[6]

Other reasons why MSY is not to be trusted as a management objective are that (i) harvesting $G(X_{MSY})$ will not necessarily be sustainable in the long run due to natural fluctuations in the stock (although similar criticism applies to other rigid policy prescriptions); (ii) the fact that the relations between interdependent species are ignored; and finally (iii) that preservation values are not included (Conrad and Clark 1987). Especially with respect to large marine mammals (see chapter 6), the latter is a serious omission.

2.2 Optimal population size and dynamics

As argued above, X_{MSY} is in general not a correct indication of optimal stock size if society wants to maximize utility of exploitation and preserving a stock (or if firms with well-defined property rights want to maximize profits). In this section, optimal stock size is determined by finding a steady-state solution to the first order conditions of the optimal control problem for society. For the dynamics of the optimal management program it makes a difference whether the current value Hamiltonian (or Lagrangian) is linear in the control variable $y(t)$ or not (Conrad and Clark 1987; Chiang 1992). Non-linearity may be caused by a downward-sloping demand curve, or by a certain specification of the harvesting costs. We will return to this issue later. Currently, the aim of the analysis is to maximize utility for society, and a non-linear specification with downward-sloping demand is chosen.

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[6]For the case of the North sea herring fishery, Bjorndal (1988) demonstrated that the economically optimum population is smaller than the population that would support harvesting at MSY level (the latter is approximately 2 million tons of herring). For various discount rates, ranging from 0% to 20%, Bjorndal calculated the optimum economic stock size, which ranged from almost 1.6 million tons to 1 million tons respectively. Both the biological and economic optimum stocks are greater than 1996 stock levels of about 500 thousand tons.

Let the management problem be formulated as follows:

$$Max_{y(t)} \int_{0}^{\infty} U(y(t), X(t)) e^{-rt} dt \tag{13}$$

subject to:

$$\dot{X}(t) = G(X(t)) - y(t) \tag{14}$$

where U indicates utility derived from either exploiting the stock or preserving it. If society faces a downward-sloping inverse demand function $P(t) = D(y)$; utilities of harvesting and conservation $[W(X)]$ are assumed separable; and if the resource in question is exploited with a simple multiplicative yield function ($y = qXE$), this management problem (without time notation) can be rewritten as:

$$\begin{aligned} Max_E \int_{0}^{\infty} \left[\int_{0}^{y(t)} D(s) ds + W(X) - CE \right] e^{-rt} dt = \\ Max_y \int_{0}^{\infty} \left[\int_{0}^{y(t)} D(s) ds + W(X) - \frac{Cy}{qX} \right] e^{-rt} dt \end{aligned} \tag{15}$$

where C is the per unit cost of effort. The current value Hamiltonian reads as:

$$H_c = \int_{0}^{y} D(s) ds + W(X) - \frac{Cy}{qX} + \lambda (G(X) - y) \tag{16}$$

The first order conditions, assuming an interior solution (see e.g. Chiang 1992; Conrad and Clark 1987; Clark 1990):

$$\frac{\partial H}{\partial y} = D(y) - \frac{C}{qX} - \lambda = 0 \tag{17}$$

$$\dot{\lambda} = (r - G'(X))\lambda - W_X - \frac{Cy}{qX^2} \tag{18}$$

Equation (17) defines the rent of the resource, or the value of an additional unit *in situ*; price minus marginal harvesting costs. Condition (18) indicates under what condition society is indifferent to holding the stock in situ, when the alternative is to harvest and invest the revenues elsewhere in the economy. This condition is a non-arbitrage condition, similar to the Hotelling rule for non-renewable resources derived in section 2.2. With fish, trees or wildlife, society is compensated for delaying harvest, because the stock is self-reproducing. The marginal rate of growth must be subtracted from the discount rate to yield a "net rate of interest", r - $G'(X)$. The two additional terms on the RHS represent benefits for society of holding one extra unit of the stock *in situ*: it is a direct source of utility for conservationists, and with the current specification of the harvest function it reduces harvesting costs.

Differentiating (17) with respect to time gives:

$$\dot{\lambda} = D'(y)\dot{y} + \frac{C}{qX^2}\dot{X} \tag{19}$$

equating this expression to (18), and solving for $\dot{y}$, we find:

$$\dot{y} = \frac{1}{D'(y)}\left[(r - G'(X))(D(y) - \frac{C}{qX}) - W_X - \frac{C}{qX^2}G(X)\right] \tag{20}$$

The optimal stock size is found by setting all time derivatives equal at zero. The implication of $\dot{X} = 0$ is that harvest should be equal to the regenerated fraction of the stock: $y = G(X)$. The optimal size of the stock can be derived from (20). Rearranging terms gives:

$$r = G'(X) + \frac{W_X + CG(X)}{X[qXD(G(X)) - C]} \tag{21}$$

In equilibrium, the social rate of time preference must be equal to the rate of return on holding the marginal unit of stock *in situ*. The latter can be decomposed into two parts: the return from increased stock growth ($G'(X)$), and a complicated stock term. The numerator of this term reflects the combination of future marginal cost savings from leaving one unit of the resource in situ because less effort will be needed when the stock is thicker, and future marginal utility from this thicker stock. These savings and gains are "discounted" by the current marginal rent due to harvesting (the denominator). Since W_X and $CG(X)$ are both positive, the stock term is positive. The larger this term, the higher the optimal steady-state stock should be: given a certain value of r, a higher realization of the stock term corresponds with lower required values for $G'(X)$ to balance LHS and RHS of (21). Again, since $G''(X) < 0$ this implies that stock size should increase.

Hence, current harvesting reduces the stock, and therefore reduces future preservation value and increases future harvesting costs in all periods. - Taking this effect into account implies that current harvesting should be reduced at the expense of future exploitation. This is reflected in the more gently-sloped rent path [see (18)] and a larger steady-state stock [see (21)]. Expression (21) is an implicit solution for the optimal steady state $X = X^*$.[7]

Due to the assumption of downward-sloping demand above, the current value Hamiltonian is non-linear in the control variable y. Under this condition, the optimal approach path to the equilibrium solution is found with aid of (14) and (20). The differential equations $\dot{X} = \dot{y} = 0$ can be depicted in a phase-plane diagram where y is plotted against X. The combination of these equations is often called an autonomous system of differential equations, as the right-hand sides of these equations are not explicit functions of time t. The equilibrium is the intersection of the two *isoclines* (i.e., the curves along which $\dot{X} = \dot{y} = 0$). From (14), we know that the X-isocline is the curve $y = G(X)$. To determine the y-isocline from (20), we need to specify the demand curve and preservation utility. An example is worked out in chapter 5 for African elephant management, and the result is provided in Figure 2.2. Note that in the absence of stock-dependent harvest costs and preservation values, the $\dot{y}$ equation would reduce to the vertical line $X = X^*$, positioned at the stock level that equates $G'(X^*)$ and r.

---

[7]Conrad (1995) calls the summation on the RHS of equation (17) the internal rate of return of the resource. The interpretation is that this rate of return equals the rate of return on investments elsewhere in the economy (the discount rate).

The two isoclines divide the graph into four *isosectors*. Each isosector has a *directional* that indicates the movement of a point (y, X) over time. In Figure 2.2 a point would move to the right if $\dot{X} > 0$ (and to the left if the reverse was true) and upwards if $\dot{y} > 0$ (and downwards if the reverse was true). The signs of $\dot{X}$ and $\dot{y}$ come from the differential equations.

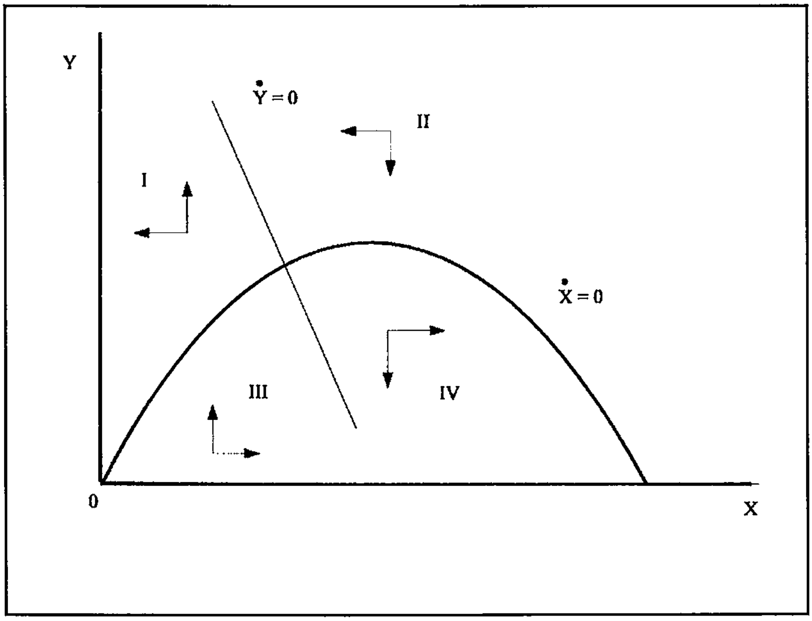

Fig. 2.2: Steady state dynamics

In Figure 2.2, from isosectors II and III, a point moves into the direction of the equilibrium. However, a point in isosector I and IV would move into the opposite direction. The equilibrium (y^*, X^*) would be classified as a saddle point.[8] In sectors II and III are trajectories that converge to (y^*, X^*). These trajectories are referred to as a separatrices, or stable branches. They define the optimal solution trajectories for the infinite horizon problem. Any other trajectory than the separatrix would be unsustainable. Initially, the other trajectories approach the saddle point, but later they veer off and

[8]This situation is typical of optimal control models with a single state variable. If the model has two or more state variables, one must fall back on numerical methods to determine the approach dynamics of equilibria (Clark 1990).

diverge from it (Clark 1990) so that eventually, either $y(t) = 0$ or $X(t) = 0$ as t goes to infinity. The separatrices can be computed by linearizing the dynamic system, and finding the eigenvalues and associated eigenvectors.[9] The eigenvectors determine the directions along which the separatrices approach the equilibrium (for more details, see Conrad and Clark 1987 and chapter 6). It is of interest to note that with a feedback control, the separatrices converge to the steady- state equilibrium (y^*, X^*), but the velocity of points nearer to the equilibrium gets increasingly smaller. The steady state is reached at t is infinity, in contrast to MRAP (or bang-bang) models where the equilibrium will be reached in finite time.

MRAP, or most rapid approach paths, are applied when the current value Hamiltonian is linear in the control variable y (Kamien and Schwartz 1994; Chiang 1992; Conrad and Clark 1987). The current value Hamiltonian in (16) will be linear in y when, for example, the objective function is reduced to: $\text{Max}_y \int [\bar{p}y + W(X) - Cy]e^{-rt}dt$, where $\bar{p}$ is a constant price per unit. This linearity condition is both necessary and sufficient for the optimality of MRAP solutions (Conrad and Clark 1987). Depending on the question whether the net revenue per unit of harvest (i.e. $P - C$) exceeds the shadow value of an additional unit of the stock λ or not, harvesting is conducted either at its maximum feasible level, or not at all. This corresponds with the following rule:

$$
\begin{aligned}
y(t) &= 0 \text{ , } \textit{whenever } X(t) < X^* \\
y(t) &= y_{MAX} \text{ , } \textit{whenever } X(t) > X^*
\end{aligned}
\qquad \textbf{(22)}
$$

In finite time, the stock will have developed or brought back to its optimal size.[10]

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[9]If both eigenvalues are positive, the equilibrium is an unstable node. If both are negative, the equilibrium is a stable node. Finally, and relevant for this kind of models with one state variable, if one eigenvalue is positive where the other is negative, the equilibrium is a saddle point.

[10]An exception arises if $E_{MAX} < E^*$. Then the optimal solution is blocked by the control constraint. Under this condition, when building up a stock, it is optimal to resume harvesting prior to $X(t) = X^*$ (for a discussion, see Clark 1990).

The discussions above, for both the non-linear and the linear model, are based on the convenient but unrealistic assumption that exploitation costs are directly related to harvesting effort. In a paper by Clark *et al.* (1979) it is explicitly recognized that non-malleable capital is pervasive in renewable resources. An important conclusion was that relaxing the assumption that man-made capital can be regarded as a flow variable has a profound influence on (short-run) optimal resource management. More specifically, under the assumption that capital investments (K) in vessels, gear and so on are completely irreversible (i.e., there is no market for excess capacity) but subject to depreciation, they found that the X-K phase plane is divided into three separate regions where it is optimal to (i) not harvest and not invest, (ii) harvest at full force and not invest, and (iii) invest at full force, i.e., increase K up to a value determined by a switching curve (see Clark *et al.* 1979; Clark 1985).

Although the optimal policy converges to the same steady-state stock as determined by the variable cost model, Clark *et al.* (1979) have demonstrated that it may be optimal to have an excessively high capital stock and a severely depleted fish stock in transitory phases. Boyce (1995) turned the linear model of Clark *et al.* (1979) into a non-linear model, such that the bang-bang solution described above was no longer optimal. However, Boyce also found that temporary phases with excess capacity and small stocks may be optimal in the short run. This may shed new light on the often heard claim that, for example, fish stocks are "over-fished" worldwide: (temporarily) reducing (fish) stocks below their optimum stock size may be economically efficient when investments are irreversible. On the other hand, with irreversibilities in exploitation, as could occur with harvesting near MVP and with tropical deforestation, the finding of rational temporary overharvesting may lose its validity.

2.3 Extinction

It has been argued that extinction of plant and animal species occurs at an increasing rate as a result of expansion of human activities. Extinction as an economic problem has received some attention (e.g. Clark 1973; Cropper

1988). Extinction can be deliberate in the case of a renewable resource with well-defined and enforced property rights, and can occur inadvertently under open access management. The latter option will be discussed briefly in section 2.4.[11]

In case of a renewable resource with well-defined property rights, deliberate extinction may occur if the return yielded by maintaining a resource stock is not competitive with investments in other assets (Neher 1990),[12] hence if:

$$G'(0) + \frac{\dot{P}(0)}{P(0)} \leq r \tag{23}$$

where $P(y)$ represents the inverse demand function for the resource. This condition is necessary, but not sufficient for extinction. If the cost of transforming the resource stock into another asset is prohibitively high [$P(0) < C(0)$], an equilibrium stock will exist. This implies that extinction is less likely when harvest costs are stock dependent (such that harvest costs increase when the stock is reduced, and $C(0)$ is likely to be high). Depending on the size of this equilibrium size relative to MVP requirements, extinction may or may not be the result.

In the absence of price effects, Cropper (1988) proved that extinction of a small stock of a renewable resource may be optimal, even if the intrinsic growth rate exceeds the discount rate, when the growth function of the resource exhibits critical depensation (i.e., when a minimum viable population for the species exists). The reason is that the net discounted benefits of refraining from harvest for a long time to build up the small stock until seizable proportions are smaller than the discounted benefits of a depletion strategy. Of course, a necessary condition for this result is that the marginal benefits from harvesting are positive at this small stock size.

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[11]We will find that when open access and rent dissipation correspond with a stock size below minimum viable population levels, extinction will be the result. Of course, when $P(0) > C(0)$, extinction will also occur when the growth function does not exhibit critical depensation.

[12]Here it is simply assumed that no social value is put on preservation of a species.

In this respect it is also important to recognize two additional issues. First, the opportunity costs of harvesting effort play an important role. If opportunity costs are low (for example, because alternative employment opportunities are scarce), exit from the unprofitable resource depletion sector will be slow (as long as revenues pay for the variable costs, harvesting may continue), and the probability of extinction is greater. Second, for fisheries, the "braking impact" of harvesting costs on effort may be obscured in a multi-species fishery. One possible reason for near extinction of blue whales (as reported by Spence 1973), is that whalers were not very selective in the choice of whales harvested (most of the products obtained from blue whales can also be obtained from most other whales, be it in smaller quantities). This implies that it is the cost of harvesting *any* type of whales that determines whether whaling will continue, not the cost of locating and killing a blue whale. See Clark (1990, p.313) for an example concerning blue and fin whales.

Finally, Swanson (1994) has pointed out that Clark's model should be adjusted for land-based resources. While marine resources may have low opportunity costs (there are few competing uses for the sea), land-based resources (such as elephants, see chapter 5) often do compete with alternative uses such as agriculture. This implies that policymakers no longer maximize the difference between revenues and costs of exploitation, but rather the difference between revenues and the sum of harvesting costs and opportunity costs. Obviously, this extension to the model increases the probability of non-sustainable management. In chapter 5 we will demonstrate that the policy implications of Swanson's model may run counter to the standard Clark model, where low prices and high harvesting costs tend to increase stock size.

2.4 Property rights

Property rights can be understood as characteristics that define the rights and duties for using a particular asset or resource. According to, for example, Bromley (1992) and Feeney *et al.* (1996), natural resources can be utilized under one of the following property rights regimes:[13]

1. Under a *state property* regime the ownership and utilization of the resource is being controlled by the state. Individuals may be allowed to use the resource but only according to the rules imposed by the state.
2. Under a *private property* regime, the right to utilize the resource, as well as buying and selling, is controlled by individuals.
3. *Common property* regime means that a group of owners can control the use of the resource and prevent others from using the resource. The members of the group have specified rights and duties.
4. Under an *open or free access* regime, each potential user of the resource has complete autonomy to utilize the resource since no-one has the legal right to keep any potential user out. One cannot speak of property since there are no property rights.

Property rights, or rather the absence of them, have played a central role in natural resource economics. Fish populations, for instance, are usually very mobile and it has been difficult to assign property rights to individuals or even nations. As a consequence, fish stocks have been harvested in many places in the world under conditions with strong open access features. It is only since the late 1970s that exclusive economic zones have been established, within which governments or intergovernmental organizations can implement fisheries policies (see chapter 7). This implies that within the exclusive zones, open access has been transformed into state property or common property such that the problems related to open access management, elaborated upon below, are perhaps less pressing than in the past.[14] With respect to forestry,

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[13]Where it is understood that in practice resources are often held in overlapping combinations of these regimes (Feeney *et al.* 1996), and that it is possible to shift from one (dominant) regime to another when conditions change. The possible switch from common property regimes and state property regimes to open access regimes as population grows has been well documented (for example, Murty 1994; Bromley 1992).

[14]Conrad (1995) writes that in a strict sense, there are few, if any, commercially valuable open access fisheries in the world today. But he adds that what might appear to be a managed fishery may in practice be one where open access conditions are approximated. De facto open access can arise if management regulations are ineffective.

lack of well-defined property rights is especially a problem in developing countries as tropical deforestation is primarily caused by advancing slash-and-burn agriculture (see 2.6.2).

As already mentioned in 1.2, the cost of exploiting a resource consists of two distinct components: the private extraction costs and the unobserved opportunity cost, or the value of the resource *in situ*. The intuition behind the latter is that harvesting one unit of the resource today implies that this unit and possible growth or offspring are not available for future consumption. The (future) value of uncaught fish and unharvested trees depends on many different factors such as the rate of interest, future markets for the resource, technological development, reproductive features and so on. A sole private owner who aims to maximize profits will maximize the discounted value of this rent, and treat the resource as an asset. Hence, the value of uncaught fish prevents a rational fisherman from overfishing his stock as long as he expects that he is the one to benefit from his current "investments" in the future. This may result in a conservative harvesting policy. Similarly, if sufficient trust and social pressure exist, common property regimes may be successful in managing resources. Cultural norms, ideology and value systems are important determinants of the actual extent of "free riding". See, for example, Bromley (1992).

A certain resource can be regarded as open access property if there is no possibility of excluding entrepreneurs attracted by excess profits. If there is unrestricted access to the resource system, for example a stock of living fish, by all who care to use it, not any fisherman can be sure of who will benefit from the value of uncaught fish. Thus, fishermen have no incentive whatsoever to take the price of fish in the sea into account when they make decisions about their individual harvesting levels. In an open access situation, not any individual fisherman has an economic incentive to husband the resource, and not any can efficiently conserve the resource by refraining from harvesting. Doing so will only enhance the harvesting opportunities for his competitors, which is the tragedy of open access harvesting. One might say that the individual cares not at all about escaped fish as someone else is likely to reap the benefit from it, and discounts future harvests at an infinite rate (Neher, 1990). New fishermen will be attracted to the fishery, or existing fishermen will expand their efforts as long as fishermen earn more than the opportunity cost of their effort CE. In the so-called *bionomic equilibrium*, all rent has dissipated, and total costs equal total revenues (instead of marginal costs equal to marginal benefits).[15] The situation where marginal costs exceed marginal benefits is usually referred to as *economic overfishing*.[16]

Gordon (1954), who was the first to explain why established fisheries were often characterized by a lot of old vessels making few or no profits, developed a simple static model of rent dissipation based on zero discounting, constant prices and the Schaefer yield-effort function. His model is illustrated in Figure 2.3. Rent dissipation occurs at an effort level of E_1. Then, $U(X,E) = (pqX - C)E = 0$, hence $X_1 = C/pq$. Clearly, this is excessive when compared to two benchmark situations: X_{MSY} (or E_{MSY}) and efficient harvesting where marginal benefits equal marginal costs (E^*). Note that the position of E^* to the left of E_{MSY} depends on the implicit assumption of zero discounting. As the discount rate increases, E^* moves to the right and eventually approaches E_1 as r goes to infinity.[17] As the effort is socially excessive, the stock is smaller than socially optimal. Effort levels beyond E_{MSY} are usually called *biological overfishing*.

Depending on the growth function of the stock (i.e., with or without critical depensation) and the specification of the production function, the stock may be driven to extinction under open access. With critical depensation,

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[15]Note that fixed costs of fishing are ignored in this line of reasoning. Fixed costs will affect both entry in and exit from the fishery. A consequence of considering fixed (sunk) costs is a "gap" between entry and exit, as fishermen will only enter when both variable and fixed costs can be paid, but will not leave the fishery before revenues fall short to cover just variable costs.

[16]In a recent article, Feeney *et al.* (1996) question the main underlying assumptions of the famous "tragedy of the commons" model (assumptions with respect to individual motivations and characteristics, the nature of institutional arrangements, interactions among users and their ability to create new arrangements, and the behaviour of regulatory authorities), and have found sufficient anecdotal evidence to refute theoretical predictions of an "automatically" dissipating rent.

[17]Note that CE is an expression of the opportunity cost of effort devoted to the fishery. Opportunity costs are the incomes that could be earned in another fishery (for instance salmon instead of halibut) or in a totally new occupation. As alternative employment opportunities in remote coastal areas may be few, these opportunity costs may be low. Clark (1990) reports that there is some evidence to support the prediction that especially the fishermen with lowest opportunity costs remain active in open access fisheries.

extinction may occur for positive stock levels as long as $X_1 < X_{MVP}$. In the Schaefer model with $y = qEX$, the catch per unit of effort y/E goes to zero as X approaches zero. With constant prices, rent dissipation will therefore occur before the stock is totally depleted.

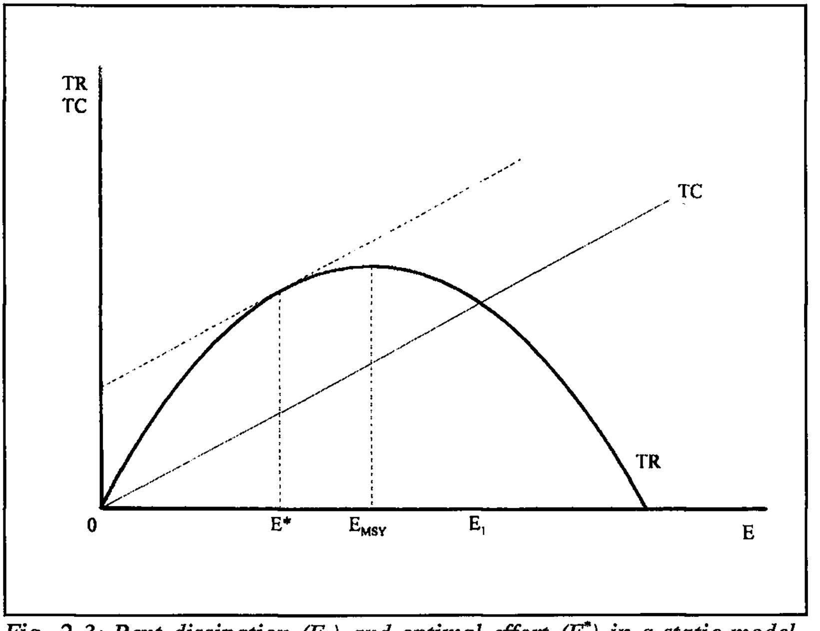

Fig. 2.3: Rent dissipation (E_1) and optimal effort (E^*) in a static model

With aid of a simple model the *dynamics* of open access can be demonstrated. Under the assumption that new entrants in the fishery are attracted with a time lag, and the response of effort leaving the fishery is also subject to a lag, a simple entry-exit function (while suppressing time notation) can be formulated:

$$\dot{E} = v\frac{(Py - CE - \alpha)}{E} \tag{24}$$

where v is an industry response parameter that describes the speed with which effort responses to profitability in the fishery, and α is a cut-off rate of return for the industry, equal to the opportunity cost of capital multiplied by the number of vessels (rE) (Hartwick and Olewiler 1986; Conrad 1995). If the stock has a quadratic growth function with parameters a and b, and we use the production function of the Schaefer model, then in the steady state the following must hold:

$$\dot{X} = aX - bX^2 - qEX = 0 \tag{25}$$

which implies that either (i) the stock has been depleted ($X = 0$), or (ii) no harvesting takes place ($E = 0$ and $X = a/b$), or (iii) an interior solution exists: $X = (a - qE)/b$. Since (23) can be rewritten as:

$$\dot{E} = v(pqX - c - r) \tag{26}$$

We know that in equilibrium exit and entry must be in balance, hence $\dot{E} = 0$ so (25) can be solved for X:

$$X = \frac{c + r}{pq} \tag{27}$$

Combining the two isoclines in a phase plane yields the steady-state equilibrium. (This steady-state equilibrium for the open access fishery is obviously not the same as the steady-state equilibrium for the optimally managed fishery as derived in 2.2, but the same technique can be applied to study it). Depending on the various parameter values and starting condition [$E(0)$, $X(0)$], the isoclines may intersect in the interior of the phase plane. Even if an interior solution exists, there is the possibility that excessive entry and delayed exit will deplete the resource before the equilibrium is reached, as is illustrated with path 2 in Figure 2.4.

The dynamics of open access resources have been analyzed by a number of researchers. Especially the situation where different countries are exploiting the same stock has received attention. For the case of the North Pacific fur seal in the late 1800s, exploited by a number of countries during the migratory phase in their reproductive cycle, Wilen (1976) calculated values for the parameters. He predicted that a positive steady state would be reached, and that stock extinction was unlikely to result. The observations on effort and stock levels were consistent with the oscillating path of Figure 2.4.

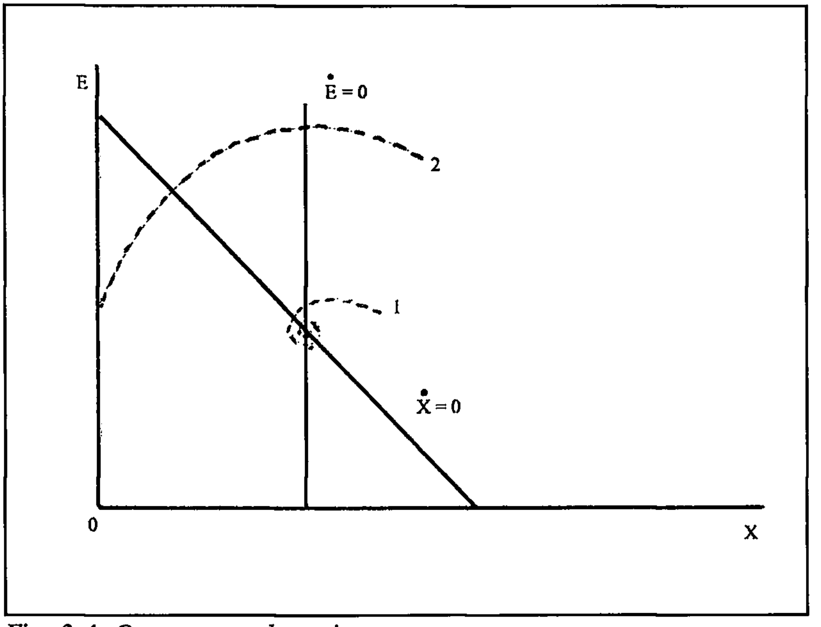

Fig. 2.4: Open access dynamics

A similar conclusion has been reached by Amundsen *et al.* (1995) for the case of open access minke whale exploitation in the North Atlantic. In contrast, Bjorndal and Conrad (1987) found that North sea herring was probably saved from extinction when, after decades of *de facto* open access management, the authorities decided to cease harvesting for a couple of years. Their model predicted that, without banning, extinction might have occurred in 1983.[18] Under the moratorium, which lasted in some sectors until 1981 and in others until 1984, the stock recovered, setting the stage for new periods of harvesting.

[18]In order to find a steady-state solution, Bjorndal and Conrad worked with a constant price-cost ratio to describe effort dynamics.

2.5 Fisheries

The issue of property rights as discussed in the previous section implied a move to the field of fisheries economics. In this section, some specific fishery problems will be touched upon. People are predators of fish, and occupy a top niche of a complicated, and for a great deal unknown, marine ecosystem. The literature on economics of fisheries, that arose since the mid-1950s, aims to provide insights into the optimal utilization of marine resources, and explains why actual exploitation may often deviate from socially optimal practice. Usually, analyses are simplified by the single species assumption, without further notice of underlying characteristics of the ecosystem and predator-prey relationships. One of the interesting areas of future research is optimal management of ecosystems instead of single fish species biomass (for an application, see, for example, Pearce and Moran 1996). In this chapter, however, I present a fairly standard description of the economics of fisheries.

2.5.1 Schooling and searching fisheries

Fisheries are commonly divided into "schooling fisheries" and "searching fisheries", the distinguishing feature being whether the species in question has the propensity to school in large numbers or not. Neher (1990) mentions as possible reasons for schooling, migration courses and defense against predators. Well-known examples of schooling species are salmon, herring, anchovies and tuna. It has been found that the size of schools does not decrease much if the overall biomass of the stock decreases. Of course, this implies that there are fewer schools.

Since large schools of fish are relatively easy to find, it is expected that searching (and catching) costs are not greatly influenced by the size of the stock in the sea. This implies that for schooling fisheries there is less reason for thickening the stock in order to reduce harvesting costs than for searching fisheries. This, in turn, means that their optimal stock size (see 3.3) will typically be smaller. Moreover, if property rights are not well established or enforced, the risk of extinction is higher, as it pays fishermen to keep fishing and reduce the stock, even if the stock is near the X_{MVP} level.

The difference between schooling and searching fisheries is formally as follows. For searching fisheries, harvesting $y(t)$ is a function of effort devoted to the catch (E) and the size of the stock (X), hence $y(E,X)$, with $\partial y/\partial E > 0$ and $\partial y/\partial X > 0$. For schooling fisheries the latter term is of less importance, and therefore often ignored in mathematical analyses: $y(E)$.

2.5.2 Fisheries regulation

As described in the previous section, property rights were not well defined in marine fisheries for a long time, which resulted in excessive harvesting. (For many years, coastal nations only controlled fisheries within a small zone ranging from an outer limit of 3 to 12 miles from the shore). Widespread depletion of near-shore and pelagic fish stocks resulted in discussions about management policies, and after the extension of territorial waters to 200 miles, a provision of the United Nations Convention on the Law of the Sea signed in 1982, there was a call for the government to step in.[19] As formulated by Anderson (1977), extended jurisdiction is an important first step, but it will be meaningless unless proper management is instituted. Otherwise the open access nature of the resource would remain, be it at a different level; fishermen of one nation would compete for the resource as opposed to the previous situation where there was national and international competition among fishers.

According to Hartwick and Olewiler (1986), economic policies to regulate fisheries should (1) find methods of rationing the amount of effort in the fishery, (2) find methods of regulating the harvest to maintain efficient stocks of fish, and (3) recognize that any policy implemented may affect the distribution of income through the reduction of effort and generating rents. They add that much existing regulation in fisheries is designed to sustain the fishery and increase the incomes of fishermen, not to reach a socially optimal

[19] Fishing jurisdiction for many valuable stocks was not easily assigned to adjacent coastal fishing nations. Rettig (1995) describes the problems associated with transboundary stocks (stocks partly outside the exclusive zone in the open seas; stocks straddling the fishing zones of adjacent countries; and highly migratory shared stocks), and the management tools that could be implemented to deal with them.

equilibrium.[20] Despite this, the literature on fisheries regulation mostly focuses on reaching optimal harvest and effort levels, and the optimal steady-state stock of a particular fishery (but see chapter 8). A number of methods to achieve this will be reviewed briefly in this section. Common assumptions are that the growth function of the fish stock is known, and that the behaviour of fishermen in the sector can be predicted and controlled. Both assumptions are often violated in practice. If policies tend to overlook the economic forces behind overexploitation, this leads to regulation measures which may temporarily reduce the level of overharvesting but which still include the market signals that cause overexploitation. This may result in serious enforcement problems and high costs. Enforcement costs are often ignored in economic analyses.[21]

Most of public regulation alternatives can be classified as follows (Tahvonen and Kuuluvainen 1995; Hartwick and Olewiler 1986; Conrad 1995):

1. *Closed seasons* limit harvesting during crucial periods when the fish population is reproducing.
2. *Gear restrictions* limit the use of 'too effective' catching devices or try to preserve the habitat of the harvested population.
3. By *limited entry* the authorities restrict the number of fishing vessels. The authorities may first require a vessel to have a license to operate and then proceed with limiting the number of licenses.
4. *Aggregate catch quotas* try to shorten the fishing period. The authorities may have a monitoring system for ceasing fishing when the cumulative harvest level equals the aggregate quota.
5. *Taxes* can be imposed on the catch or on some harvesting input.
6. *Individual transferable quotas* (ITQ) limit the level of harvest for each individual fisherman per fishing period.

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[20]Anderson (1989) argues that for most commercially interesting fisheries in the world, economic and biological data that should support optimal harvesting paths are not even available. Pearse (1980) writes that perhaps the best that can be achieved by regulation is to allow a safe catch to be obtained as efficiently as possible.

[21]Exceptions are, for instance, Graves *et al.* (1994); Salvanes and Squires (1995); and especially Sutinen and Anderson (1985).

7. *Establishing ownership* by forming cooperatives strengthen social pressure, and may rule out "rape, ruin and run" behaviour that way.

The aim of closed seasons and gear restrictions is to reduce the effectiveness of harvesting or increase the real cost of fishing. Conrad and Clark (1987) refer to these policies as the economic perversity of purely biological conservation. The effects are easily demonstrated with Figure 2.5. If TR_1 indicates the total revenue curve for the fishery, then under conditions of open access, the bionomic equilibrium is at E_1. If the regulator wants to control effort, and bring it back to, say, the E_2 level (which is optimal in the context of a static model as it equates current marginal benefits and costs of harvesting), then one possible way to do this is to rotate the total cost curve TC_1 upwards to TC_2. Imposing inefficient harvesting techniques[22] and closing the fishery for long times such that the vessels remain idle will increase the costs of fishing. While these policies may meet the goal of stock conservation ($X = X^*$ in the new equilibrium), they will still cause economic inefficiency and wasting of resources, i.e., excessive working hours for fishermen and large expenditures on capital and fishing equipment to achieve the desired level of harvest. Economic efficiency requires that the level of annual harvest reflects the true value of the fish in the sea and that harvest takes place at the lowest possible cost. By increasing the cost of harvesting, the regulation policy reduces the income level of fishermen as the rent is still zero.[23]

Closed seasons may have an additional effect, which is also likely to occur when the policy is simply aimed at limiting the number of vessels in a particular fishing area: it will easily lead to a phenomenon called 'capital stuffing'. When the ships lie idle in the harbour, the harvesting capacity of existing vessels in the longer run is increased by investing in more powerful engines, larger nets or by adding more or better electronic devices to find the schools of fish more efficiently. This has two effects. First, the danger of

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[22]For instance, until the 1950s the Alaskan government barred certain fishers from using engines to propel their boats, such that they had to use sailboats (Tietenberg 1994).

[23]Conrad (1995) remarks that gear restrictions such as minimum mesh size may also have an ecological basis, as it is hoped that young fish will escape the harvest and provide the necessary growth and recruitment to the fishery in later periods.

overharvesting may still exist as there are no binding constraints on total catch. Harvest capacity may increase and more fish are caught in a given period.[24] This is what Anderson (1995) refers to when he argues that input restrictions tend to be "rubber yardsticks". Second, scarce resources are wasted as capacity becomes excessive or operational decisions of participants in the fishery are distorted. The reason why excessive investments in harvesting capacity are likely to be undertaken is that fishermen still have strong incentives to start harvesting at full capacity, because being too slow at the beginning of the season may imply that their annual share in the total quota will be small.

The fundamental problem of open access fishery is that fishermen have no incentive to take into account the value of fish left in the sea or the shadow price of the resource. Conrad and Clark (1987) demonstrate that a management agency can force fishermen to recognize this cost by imposing the shadow price as a tax on harvests. This tax reduces the revenue to the fishery for each unit of effort employed. This is illustrated in Figure 2.5 by the new revenue parabola TR_2 [$TR_2 = (P - t)y$, where t is the per unit tax]. Fishermen now set $TR_2 = TC_1$, yielding the desired amount of effort E^*.

While the resulting outcome is theoretically efficient (in the context of a static model) and does not involve tedious monitoring of fishing effort, some problems are associated with the taxation approach. First and most importantly, as made clear by Munro and Scott (1985), this regulation measure may politically be unfeasible. The tax transfers all of the economic rent from the fishing industry to the government, and fishermen will use all their political power to prevent this kind of policy from being implemented. Second, the management agency may have great difficulties in computing the varying shadow price, which depends on factors as demand for fish and biological processes. Also, enforcement of a tax on harvest is hard, and collection may be difficult (Hartwick and Olewiler 1986). Rettig (1995) writes that taxes are often inflexible, which may be hard to reconcile with fluctuating stocks of fish.

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[24]Anderson (1995) describes this as a game between the manager and the fishing industry, where the industry always has the last move.

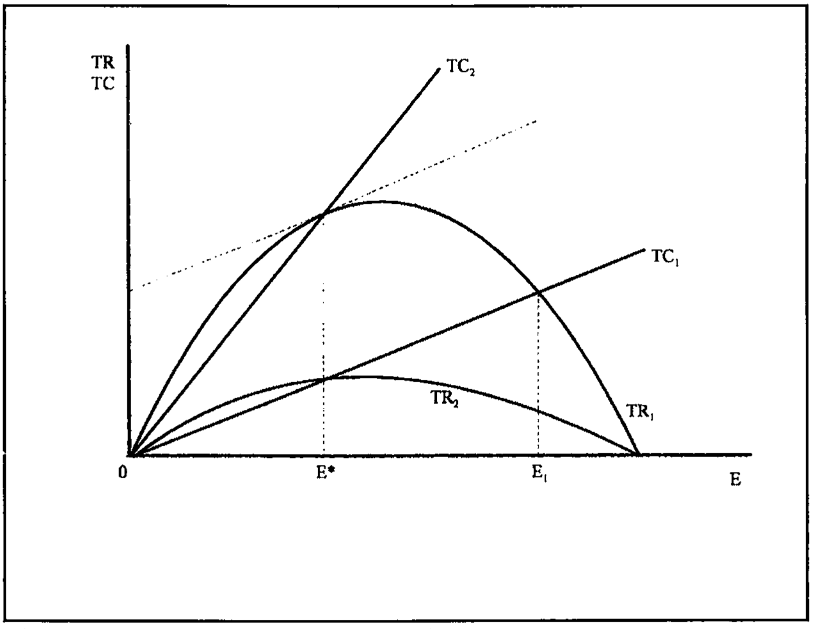

Fig. 2.5: Instruments for fishery policies

An alternative to a tax on harvests would be a tax on effort. Effort is often measured as the number of vessel days devoted to fishing during a particular year (Conrad and Clark 1987). The effect of such a policy would be very similar to the situation of increasing harvesting costs by imposing inefficient regulations. The difference is that resources are not wasted in an inefficient harvesting process, but collected by the government instead. At least two different taxation schemes are possible. First, a head tax can be imposed, levied on every fisherman in the sector. An example is a simple license fee f. This shifts the total cost curve up ($TC_2 = CE + f$).[25] Second, a unit tax on effort t can be imposed: $TC_2 = (C + t)E$, which rotates the total cost curve upwards. In principle, both taxes could lead to optimal harvest levels.

[25]Note that, if the tax is not to be paid in every period, it will be a sunk-cost for the fishery, which does not influence the marginal amount of fishing effort.

Apart from the difficulties associated with taxing described in the previous paragraph, in reality it is difficult to define what "effort" is in this respect, as fishermen have the incentive to simply substitute types of effort that are not taxed for types that are taxed. Empirical studies of substitution possibilities between restricted and unrestricted inputs in fisheries, such as Squires (1987), Campbell (1991) and Dupont (1991), indicated that fishing harvest technology is typically not the fixed-proportions type, as usually assumed for convenience in the literature.[26]

Another means of regulating a fishery is implementing a quota scheme. The authority simply limits the quantity of fish that can be harvested in a given period of time.[27] The moratorium on harvesting herring as mentioned in section 2.4 is an extreme example of such a quota. The Common Fisheries Policy (CFP) of the European Union is also based on a quota system. Typically, first the total allowable catch (TAC) is determined, which is based on biological, economic and often political considerations. Next, this TAC is distributed over the member states, which use it as a basis for their national policies. National quotas may be distributed amongst the fishermen (as in the Netherlands), or the fishery can simply be opened up, and be open until the national quota is reached.

While a quota system may result in optimal harvesting levels (provided that the authorities have access to all relevant data, and that monitoring and enforcement of the policy is not too costly), a quota system will not always result in efficient allocation of effort. If the fishery is opened up as long as the TACs are not reached, it is possible to end up in a situation where fewer fish are caught with more effort, as firms will rush to capture as much as possible of the quota before others get the fish first. This may again result in the paradoxical 'capital stuffing' situation, where fishermen invest in more powerful engines and increase their harvesting capacity, for an ever-shorter

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[26]Note that similarly, when output is taxed, fishermen have the option to substitute taxed species for untaxed species in a multi-species fishery.

[27]It is also possible to implement quota on effort. Apart from the question what factor inputs need to be restricted, which is very similar to the question of what inputs should be taxed, the problems are the same as those to be discussed for harvest quota.

fishing season. Next, investments in extra capacity are necessary in processing and retailing sectors to handle the large irregular quantities of fish that are dumped on the market for an ever shorter time interval.[28] Rents are likely to dissipate, and the situation is very much like the open access fishery, with the main difference that fish stocks are potentially protected from over-exploitation by the quota.

The open access problems can be overcome if property rights are allocated at firm or fisherman level. If fishermen have the rights to harvest a certain quantity in a specified time interval, they will decide to use their effort such that, for instance, harvesting costs are minimized if discounted prices are constant, or that supply is concentrated in periods of high demand and high prices. Economic efficiency at the firm level is obtained, but from society's point of view it is still possible to improve the allocation of effort as it is not clear that the most efficient (low-cost) fishermen are doing the harvesting. It is possible to concentrate the quota with low-cost firms by cumbersome administrative procedures, by auctioning them off, or by allowing trading of the quota.

As mentioned above, a quota establishes a property right, and as such it has a value. The price of the quota is the value of the *in situ* resource, which is simply the market price minus the marginal harvesting cost.[29] Firms with low costs could bid more for a quota at an auction, and thereby acquire them. Likewise, if quotas are tradeable, low-cost firms will buy them from high-cost firms, thereby making everybody better off. In equilibrium, the price of transferable quotas is equal to the rent of the resource (Clark 1980;

[28]Tahvonen and Kuuluvainen (1995) describe the wasteful harvesting of salmon in Sweden. Salmon was abundantly available in a short time for low prices, but at Christmas, when prices are high, it had to imported.

[29]Enforcement of quota rights is a necessary condition for quota prices to reflect scarcity rent. For example, ITQs for flatfish species were introduced in the Netherlands in the early eighties, and prices of these quotas rose sharply in the later 1980s after the government had made it clear that enforcement would be taken more seriously. Without monitoring and enforcement, *de jure* property rights are of little value.

Anderson 1995),[30] and the impacts of the quota and optimal tax system are identical in the sense that socially efficient allocation of effort is the result if perfect information about biological and economic conditions is available and enforcement causes no problems.[31] [For long-run industrial equilibrium in such an ITQ fishery, and the effect of changing prices and capital costs on the optimal size of the firm, see Hannesson (1996)].

However, there is a distributional difference.[32] Distribution of quotas is in a sense distribution of income, which may result in political complexities very similar to the allocation of pollution allowances over and within countries in a system with tradeable pollution permits. Both auctioning off the quota and the tax approach generate public revenues, and income is transformed from the fishery to the government, which may result in considerable resistance from the sector (Tietenberg 1994). If instead quotas are distributed among fishermen on the basis of historical harvesting (a sort of "grandfathering approach"), current fishermen receive a kind of gift, at the expense of society as a whole. Since the relatively small fishery sector is better organized than the amorphous group of tax payers, and has more to lose from an auction than any tax payer has to gain, public choices theories predict that the interests of the sector prevail.

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[30]If problems with disinvestment exist, for instance because vessels are not easily absorbed in (or transferred to) other fisheries, firms compete to gain enough rights to continue operating and make at least a contribution toward covering their fixed costs. Hence, fishermen look at their average variable cost rather than average total cost, and prices for individual transferable quotas will be higher (Anderson 1995).

[31]Enforcement costs depend on social norms (which determine the degree of self-enforcement) and observability of behaviour. Salvanes and Squires (1985) argue that, even for a decentralized regulation scheme based on transferable quotas, information on the underlying technology of the fishing vessels may be necessary because of interrelated production processes (i.e., multiple species harvesting) and to reduce incentives for underreporting of the actual catch and discarding of less valuable species. They propose constructing a typical firm for monitoring and enforcement.

[32]Besides the interpretation in the text, there is also considerable attention in the literature on the effect of transferability on industry structure. More specifically, the fear has been expressed that eventually the rights end up in the hands of a few large firms (OECD 1993). A recent analysis of the effect of transferable catch quotas on the structure of the Dutch fleet, however, provided not much support for this hypothesis (Davidse 1995).

Wilson (1982) is critical about the implications of conventional fishery economics for policy makers. He points to the lack of attention for transaction and informational costs, and the simplified and unrealistic assumptions with respect to fisherman behaviour [an issue elaborated upon by Feeney *et al.* (1995)] and fish biology. Wilson concludes that the social costs of unregulated fishing are typically less than might be expected on the basis of accepted theory (for example, fishers switch from one species to another when the population gets depleted and harvesting costs increase) and that attempts to regulate will usually imply higher costs than anticipated. These two effects limit the range of economically feasible management options.

2.5.3 Interaction among species

Modelling the dynamics of populations with differential equations (see previous sections and, for example, chapter 5) or difference equations (see chapter 6) implies a neglect or simplification of the ecological interaction among species, as mentioned above. While this may not pose serious problems when only one species is subject to exploitation, Clark (1990) argues that with ever increasing demand on renewable resources, such single species models become increasingly inadequate. He divides interactions between exploited populations in two broad classes: biological and economic interactions. An example of the latter was the case of harvesting cost of blue whales in a multi-species fishery, where the stock size of one species affects the marginal harvesting cost of another species [see section 2.3; Anderson (1977) refers to this as an example of a *technical interdependency*] Biological dependencies concern the competitive, commensal and predator-prey relations that species have with other species. Considering interactions among species in economic models implies working with multiple-state variables. Analytical results tend to get fewer because of the associated mathematical complexities involved, and often we need to rely on numerical solutions for approach dynamics.

Most multi-species models that have been developed in recent years are partial models, in the sense that many important relationships and interactions are ignored. This may be due to incomplete knowledge of (marine) ecosystems. Clark (1990), for example, writes that no marine ecosystem has probably been studied in sufficient depth to warrant the use of sophisticated multi-species models.

An important feature of (partial) multi-species models is whether selective harvesting is possible or not. When multiple species are captured with the same gear, differentiation may not be possible. In those cases, however, in which ecologically interacting species can be harvested selectively, the standard theory developed in this chapter seems adequate, although it should be slightly modified to allow for these interactions. For example, Hartwick and Olewiler (1986) analyze the case of a simple predator-prey model with sharks and tuna. Not surprisingly, they conclude that it is socially desirable to subsidize shark harvesting such that the steady-state population of tuna rises and more tuna is available for human consumption. Similarly, Flaaten and Kolsvik (1996) examine a predator-prey relation where both predator and prey are commercially interesting species (here the predator is cod). In the context of a multiple cohort model with the opportunity cost of predation in the objective functional, they conclude that the optimal age of starting harvesting is significantly reduced: Harvesting cod yields not only a commercially valuable product, it is also a kind of investing in the stock of the prey species (in this case: shrimp, herring, capelin and little cod). Unlike the conclusion reached by Hartwick and Olewiler, there is no need for subsidies to induce extra cod harvesting. In fact, they conclude that the optimal age in the Northeast Arctic is still high compared with the actual age of harvesting.

More recently, Flaaten and Stollery (1996) have examined the economic cost of predation of Minke whales on economically valuable species such as herring and capelin. Because modelling the interactions among prey species (i.e., modelling a more complete multi-species model for the ecological system) proved to be very difficult (see Clark's remark above) their model neglects these effects. It is shown that the cost of predation depends on the management regime that the prey species is subjected to (for example, think of constant effort as opposed to constant catch policies), which is to be expected. In an empirical section it is shown that the economic cost of whale conservation, in terms of fishery benefits foregone, may be considerable. This is elaborated upon in chapter 6.

The case where selective harvesting is not feasible proves to be more difficult. Consider the case of combined harvesting of two ecologically independent populations x and z with differential productivity. If E represents effort devoted to joint harvesting, then $\dot{x} = rx(1\text{-}x/K)\text{-}q_1Ex$ and $\dot{z} = \gamma z(1\text{-}z/M)\text{-}q_2Ez$, where r and γ denote intrinsic growth rates and q_i, $i=1,2$, is the (possibly) different catchability coefficient for the two species. Clark (1990) demonstrates that when such species are harvested jointly, the less productive population may be driven to extinction, whereas the other species continues to support the fishery. The ratios r/q_1 and γ/q_2 are referred to as biotechnical productivity. As a general rule, populations with a relatively low biotechnical productivity (i.e., species with low growth rates and/or high catchability coefficients) are subject to extinction under joint harvesting conditions, provided that the cost-price ratio of the other species is sufficiently low to ensure sustained profitability of effort devoted to the harvesting process. Extinction may occur under conditions of open access, but it may also be optimal (even with zero discounting) for a profit maximizing regulatory agency.

Finally, Anderson (1995) mentions another aspect of catching several types of fish simultaneously. Under a quota system this may cause problems as it may result in discarding fish for which no quota rights are held. Anderson describes a number of ways to address this issue, such as retroactive trading (i.e., buying and selling quotas after the harvesting has occurred) and allowing fishermen to exceed current quotas with any overages today deduced from next period's allowable catch.

2.5.4 Uncertainty in fisheries

In addition to the types of uncertainty addressed in section 1.2, fisheries management involves uncertainty in the reproductivity of the stock. Pindyck (1984) argues that the natural rate of growth of the stock is stochastic for virtually all resources. In a model with well-defined property rights and endogenous prices, he has demonstrated the role of uncertain asset growth. The standard equation of motion (equation 4 in section 2.1) is replaced with:

$$dX = (G(X) - y)dt + \sigma(X)dw \tag{28}$$

with $\sigma'(X) > 0$, $\sigma(0) = 0$ and $dw = \epsilon(t)(dt)^{0.5}$. Here $\epsilon(t)$ is a serially uncorrelated and normally distributed with unit variance, implying that $w(t)$ is a Wiener process. The current stock X in this model is known, but the instantaneous change in the stock is partly random. Pindyck's objective function is as follows:

$$MAX_y \int_0^\infty (P - C(X))ye^{-rt}dt \tag{29}$$

Straightforward solving of this problem with the deterministic equation of motion [equation (4)], gives the following expression for the optimum stock:

$$\frac{\dot{P} - \dot{C}}{P - C} + G'(X) - \frac{C_X y}{P - C} = r \tag{30}$$

Pindyck solves the model with the partly random equation of motion while using stochastic dynamic programming, and obtains the following solution for the stochastic model:

$$\frac{E(\dot{P} - \dot{C})}{P - C} + G'(X) - \frac{C_X y}{P - C} = r + \sigma'(X)\sigma(X)A(X) \tag{31}$$

where $A(X)$ is thought of as an index of absolute risk aversion. The interpretation of (30) is that the market interest rate is augmented by a risk premium, equal to the increase in stock growth variance attributable to the marginal *in situ* unit times the index of risk aversion (Pindyck 1984). Fluctuations reduce the value of the stock, and because their variance is an increasing function of the stock level, there is an incentive to reduce the stock. From (30) it is evident that the expected capital gain required to hold one unit of the resource *in situ* is higher, and *ceteris paribus*, the rate of extraction is higher and less of the stock will be held *in situ*.

Pindyck has demonstrated that rent and extraction rate are affected by random growth in two other ways as well. First, since $G(X)$ is concave, stochastic fluctuations in the stock level reduce the expected rate of growth, which is an implication of Jensen's inequality. This results in increased scarcity, and reduced extraction. Second, since the cost function $C(X)$ is typically convex, fluctuations in the stock level increase expected extraction cost over time, again an implication of Jensen's inequality.[33] This effect reduces rent and increases the rate of extraction. Pindyck then concludes that, given a particular current stock level X, the net effect of uncertainty on the current rate of extraction is ambiguous.

More ambiguity arises if, in addition to stochastic growth of the resource, a stochastic stock of pollutants is modelled. Olsen and Shortle (1996) have elaborated upon the Pindyck paper by introducing a stochastic equation of motion for a stock of pollutants (s) adversely affecting growth and (perceived) quality of fish:[34]

$$ds = (z - \delta s)dt + \sigma_1(s)dw \tag{32}$$

$$dx = (G(x,s) - y)dt + \sigma_2(x,s)dw_2 \tag{33}$$

where $z(t)$ is the emissions rate of the pollutant, δ is the expected rate of decay, and $\sigma_1(s)dw$ describes random disturbances in the level of the pollutant. It is assumed that the variability of the disturbance increases or is independent of the stock of pollutants. It is also assumed that the variability of the growth rate increases with the stock of pollutants. Olsen and Shortle find that the new source of uncertainty adds a term to the RHS of the equation for the optimal stock of fish, analogous to Pindyck's risk premium. Unlike Pindyck's "risk premium", however, the sign of this extra term is ambiguous. It depends on the correlation coefficient between both stochastic processes and the change in the value of the fishery due to a change in s. This implies that the required rate of return of the resource in situ is increased or decreased, hence that

[33] Note the similarity between these arguments and Pindyck's (1980) stochastic exploration model, discussed in chapter 1.

[34] Their model is partly based on a deterministic model by Tahvonen (1991).

optimum population size can go up or down. Due to Jensen's inequality there are two more effects of a randomly fluctuating stock of pollutants. First, since the growth function of the fish stock is assumed concave in x and s, the expected growth rate is reduced. This implies that optimal harvesting is reduced. Second, with consumption benefits concave in the pollution level, the expected benefits of future consumption are reduced, which tends to enhance current harvesting. As in Pindyck's case, the net effect is ambiguous.[35]

2.6 Forestry issues

Traditionally, an important issue in the economics of forestry is calculation of the optimum rotation of a stand of single-aged trees. Since this topic is not easily discussed within the context of the preceding sections, I will review briefly some major insights that have developed in this field of research in this section. Tropical deforestation is dealt with in a subsection.

2.6.1 Optimum rotations

For the sake of simplicity it is often assumed that growth of individual trees and forest stands can be accurately described by a logistic growth function. The development of volume and wood quality over time is much more subject to management (silviculture) than fisheries. Define $V(T)$ as the function that describes volume over time. For a long time, maximizing sustained yield (or mean annual increment, MAI) was the focus of foresters. This amounts to simply maximizing $V(T)/T$ with respect to T, which readily solves for $\dot{V} = V(T)/T$. This solution disregards prices, costs and discounting, and is therefore generally rejected by economists. For economists, the traditional forestry problem ("when to cut a tree?") is formally specified as follows:[36]

[35]For information on the optimal emissions rate under uncertainty, the reader is referred to Olsen and Shortle (1996).

[36]Note that $1/(1-e^{-rT})$ is a convenient way to write $1+e^{-rT}+(e^{-rT})^2+\ldots$

$$Max_T \frac{PV(T)e^{-rT} - C}{1 - e^{-rT}} \tag{34}$$

where P is the (fixed) stumpage price per unit of wood, C are (fixed) regeneration costs and T is the optimum rotation cycle that is to be found. Hence, we begin with bare land and try to find the rotation length that maximizes the present value of revenues from current and future harvesting cycles (it is implicitly assumed that forestry is the optimal land use, but the model can easily be extended to allow for exogenous land rents or incorporation of alternative forms of land use). With P, $V(T)$, C and r constant, it is easy to see that T is the same for all future harvest cycles.[37] The first order condition for this problem is:

$$P\dot{V}(T) = rPV(T) + rW \tag{35}$$

The interpretation of this condition (the so-called *Faustmann* solution) is straightforward. The optimum harvest cycle is characterized by the requirement that marginal benefits of postponing harvesting one more period (more wood to sell at price P) are equal to the marginal cost of delaying the harvest. The opportunity cost involves two components: first, delaying current harvesting implies that potential interest income is foregone, and second, there are costs of delaying harvesting of all future rotations (delaying harvesting necessarily implies that re-planting starts later too). The term rW (where W is the maximum present value of the stand when the land is to stay in forestry in perpetuity) is also called the implicit rental cost of the land. When land is in unlimited supply (Hartwick and Olewiler 1986), or if C is high relative to P (Bowes and Krutilla 1985), the land rent approaches zero. However, in general solving for a single rotation without consideration of the opportunity cost of land results in incorrect solutions (see Samuelson 1996 for a lucid discussion).

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[37]McConnell *et al.* (1983) have expanded the Faustmann model by allowing for non-constant prices and costs, such that site rent changes over time. They find that with constant discounted price and increasing (decreasing) discounted costs over time, the rotation length increases (decreases).

Bowes and Krutilla (1985) demonstrate comparative statics of the Faustmann solution by rewriting (34) as $P\dot{V}(T)/(PV(T)\text{-}C) = r/(1\text{-}e^{-rT})$, where the LHS is interpreted as the relative growth rate in net harvest revenues. The LHS and RHS of this expression can be plotted in a graph as in Figure 2.6.

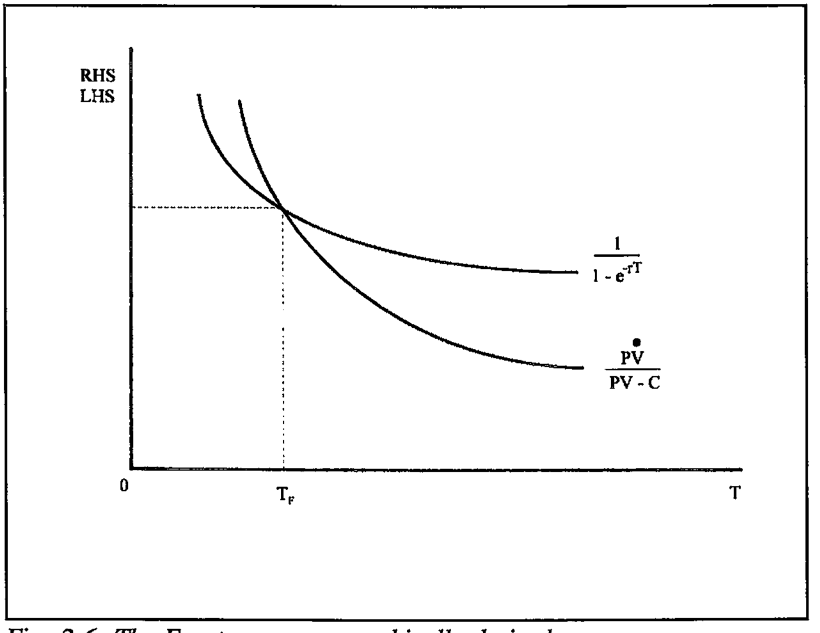

Fig. 2.6: The Faustmann age graphically derived

Obviously, the Faustmann age occurs at the intersection of the two curves. Now, (i) a higher price shifts the LHS-curve downward such that the intersection occurs at a younger age, (ii) a higher cost shifts the same curve upward, such that rotation age is increased, and (iii) a higher r shifts the RHS curve results in an upward change of the RHS curve, such that optimum rotation length is reduced. The intuition behind these results can easily be understood with equation (34). Although the process of discounting usually implies that economic rotations are shorter than the rotation length that maximizes sustained yield, Binkley (1987) has demonstrated that this is not a general result. As argued above, high clearing or regeneration costs tend to postpone harvesting, so that they are incurred at a later date, not only for the current rotation, but for all rotations that follow. Specifically, Binkley (1987) has shown that the economic rotation is greater than the MSY rotation when the discount rate is less than the inverse of the MSY rotation.

The Faustmann solution does not take non-market benefits into account. This implies that the optimum rotation age as prescribed by the model may be sub-optimal for management of, for example, public-owned forests. Hartman (1976) has generalized the Faustmann model by adding a flow of non-timber benefits, related to the age of the standing stock. These benefits could include recreation benefits or nature conservation. If these benefits at age n are called $B(n)$, then $\int_0^T B(n)e^{-rn}dn$ represents the present value of these flow benefits for a rotation length T. The problem reduces to:

$$MAX_T \frac{PV(T)e^{-rT} + \int_0^T B(n)e^{-rn}dn - C}{1 - e^{-rT}} \tag{36}$$

The first order condition of this problem is called the Hartman rotation age:

$$P\dot{V}(T) + B(T) = rPV(T) + rW \tag{37}$$

The optimum rotation is where marginal benefits of delaying harvesting [which now include $B(T)$, the flow of amenity benefits during delay] are equal to the marginal costs of delay. The land rent reflects the value of land for timber production and flow services.

Note that the effect of non-market benefits on rotation length is ambiguous. At one extreme, with $B(n)$ very large and increasing with stand age, it may be optimal to never harvest the stand. In general, with $B(n)$ increasing in stand age, the Hartman rotation is longer than the Faustmann rotation. However, with $B(n)$ declining in stand age, the Hartman rotation is shorter than the Faustmann age [Bowes and Krutilla (1985) mention forage production as a possible flow benefit which may decline as the stand matures]. Not surprisingly, flow benefits do not affect optimum rotation length when $B(n)$ is independent of stand age. With $B(n) = B$, harvesting and subsequent planting do not affect the flow of services, hence the Faustmann and Hartman rotation coincide.[38]

Economic literature provides more extensions of the Faustmann model. Van Kooten *et al.* (1995) have done so by incorporating the forest's ability to "store" carbon dioxide, and as such mitigate the greenhouse effect. They assume that carbon uptake is proportional to the growth of a forest, which implies that growth of the forest is valued, i.e., changes in the stock, instead of the stock itself. They conclude that incorporation of this externality in general results in slightly longer rotations. Reed (1984) modified the Faustmann model by including an age independent probability of forest fires that would destroy the stand. Consistent with intuition, such a risk would add a risk premium to the discount rate, which reduces the length of the optimum rotation.

As mentioned above, McConnell *et al.* (1983) have relaxed the restrictive assumption of constant prices in Faustmann models. They argued that historic prices are subject to a rising trend, which is likely to continue in the future. Apart from the question whether prices follow a (long-run) trend, there is abundant evidence that short-run stumpage prices follow a stochastic process, be it "efficient" (i.e., Washburn and Binkley 1990) or not (Hultkrantz 1995). Assuming the latter, Brazee and Mendelsohn (1988) demonstrate that the present value of expected returns are increased significantly when, instead of working with expected prices according to the Faustmann tradition, the landowner applies the so-called reservation price approach based on asset sale models or search models. This implies solving a dynamic programming problem for (age-dependent) reservation prices. Reservation prices are based on the expected return that can be earned by delaying harvesting. This approach is applied in chapter 11 of this thesis.

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[38]An interesting feature of the model with flow-benefits is that the age of the stand matters when we do not start out with bare land. In the Faustmann model, immediate harvesting is prescribed when the current stand age is older than the Faustmann age. However, in the Hartman model it may be optimal to delay harvest further or never harvest at all (Bowes and Krutilla 1985). The intuition is similar to the one provided by Cropper (1988) with respect to optimal extinction and inherited stock size (see 2.6): When the inherited stand is young, discounting may prevent us from waiting for high amenity values from old growth forests. However, when by chance an old-growth stand is inherited, the current high benefits of the amenity flow may justify further conservation.

2.6.2 Tropical deforestation[39]

Tropical deforestation is a subject that has received ample scientific and popular attention in recent years.[40] Tropical forests serve a variety of functions, among which conservation of biodiversity, local and global regulatory functions and provision of useful forest products. Many of these benefits accrue to people in other countries. This gives rise to the usual problems of taxing and subsidizing externalities. Theoretically, three different levels of efficient harvesting of tropical forests can be distinguished, with decreasing exploitation intensity: (i) efficient harvesting at private (logging) firm level, where discounted (expected) profits are maximized, (ii) efficient harvesting at national level, where regional and national externalities as watershed protection and protection against soil erosion and nutrient loss are incorporated into decision making, and finally (iii) global efficient harvesting where global externalities such as biodiversity protection and global regulatory functions (for example, the forest as store of carbon dioxide) enter the analysis. The Western world should aim to persuade developing countries to adopt the latter option, but some evidence exists that current deforestation proceeds at a pace faster than the private optimum (Barbier *et al.* 1994).[41] An often-mentioned reason may be perverse government policies (e.g., distortive taxation, uncertain tenure rights, short concession contracts, industry protection but

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[39]This section is based on: Bulte E.H. en D.P van Soest (1996), "Ontbossing en de Handel in Tropisch Hardhout", *Economisch Statistische Berichten* 80, nr. 4034, p.1044-1047.

[40]For a critical review of some empirical studies in this field, see Kummer and Sham (1995). A major concern is that in many cross-sectional studies (percentage) forest cover serves as a proxy for deforestation, because time-series data on specific countries are often lacking. If anything, forest cover represents cumulative deforestation, rather than recent or current deforestation. This implies that conclusions about deforestation may be invalid when either the initial forest stock or the period during which past deforestation has occurred differs between different countries in the sample.

[41]It should be recognized that few studies have been directed at establishing socially optimum deforestation rates and steady states of forest cover, which basically implies that the benchmark against which current deforestation has to be measured is often lacking. A notable exception is a study by Ehui and Hertel (1989) for Ivory Coast.

also pricing and exchange rate policies). For an overview, see Repetto and Gillis (1988).[42]

According to the FAO (1993), tropical deforestation advanced during the 1980s at a pace of 15 million hectares annually. This would imply that each year approximately 1.5% of the total stock of tropical forest was removed. For many, this is a cause of great concern. Conventional wisdom is that there is a strong relation between tropical deforestation and the trade in timber. Closer examination of the data, however, suggests that this relation is in fact weak and mostly indirect. Empirical research into the timber trade and deforestation (for instance, reported in Barbier *et al.* 1994) does not provide a strong or even significant connection. The reason is that commercial logging is not the primary cause of deforestation, as often suggested. According to Amelung and Diehl (1992) at least 85% of deforestation is due to advancing agricultural conversion, and at most 10% due to logging. Logging may (severely) damage forest areas, but mostly the forest is able to regenerate itself such that at least a tree cover is maintained.

About 80% of the logged wood is for domestic consumption in the form of fuelwood. Barbier *et al.* (1994) estimate that 17% of the volume of wood is for industrial purposes. Only a third of this industrial wood is exported. Hence, less than 6% of the wood produced is traded internationally and less than 1% of tropical deforestation can be directly linked to the timber trade ($0.1 * 0.17 * 0.3 = \pm 0.005$). This suggests that trade restrictions may not be effective instruments to combat deforestation, in spite of recent international agreements (see e.g. ITTO 1992). Instead, many researchers (e.g. Jepma 1995; Myers 1994) have indicated that tropical forest conservation can only be achieved by reducing population growth (but see Kummer and Sham 1994) and illiteracy, investments in productivity growth of agriculture, labour-

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[42]Deacon (1994) empirically analyzed the relationships between deforestation and population pressure, income growth and insecure tenure rights, with special attention to the latter possible explanation. He has found some support for the hypothesis that (lagged) population growth contributes to deforestation and (very) little evidence for the hypothesized role of tenure security. Since there exists a strong theoretical support for the claim that tenure insecurity should imply a (higher) risk premium which tends to accelerate harvesting, this is an interesting field of future research (see also Dasgupta and Heal 1974; and Long 1975). For an overview of empirical work in this field, see Brown and Pearce 1994).

intensive industrialization, land reform and establishing or enforcing secure tenure rights. In the words of Myers (1994): "*The source problem is an amalgam of non-forestry problems, ... hence the overall problem [of deforestation] must be tackled largely through non-forestry measures*".

Even though the direct link between deforestation and commercial logging is weak, there are clear indications that a more indirect relation exists. To transport the logged wood from isolated forest areas to ports or markets, commercial foresters have to construct a road network. It has been observed that this road network facilitates advancement of slash-and-burn farmers. Hence, agricultural conversion proceeds more rapid in secondary, or over-logged forests, as compared to pristine, undisturbed primary forests (see chapter 3). This suggests that commercial logging acts as a catalyst or pacemaker for the deforestation process by providing the necessary infrastructure.[43] In chapter 3, I will explore this typical aspect of deforestation (primary forests are turned into secondary forests by logging and are next subject to encroachment) in more detail.

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[43]Note that it is not easy to make recommendations with respect to trade policy from this observation, because it is uncertain what the effect of trade restrictions is on harvesting. First, trade restrictions in the form of sustainability requirements (possibly corresponding with cutting fewer trees per hectare) could provoke usage of extra hectares to compensate for any loss of revenues. Access to these extra hectares subject to selective logging is facilitated for encroachment. Second, trade restrictions that reduce the value of forest as a standing stock increase the probability that the present value of alternative land use options exceeds the present value of sustainable forestry. In the long run, this could result in excessive conversion of forests for alternative purposes, and a lower steady-state forest stock. For an interesting empirical study of policy evaluation with a general equilibrium model, see Thiele and Wiebelt (1993).

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CHAPTER 3

TROPICAL DEFORESTATION, HIGH DISCOUNT RATES AND SLASH-AND-BURN AGRICULTURE[1]

Abstract

In this chapter the validity of two widely held presumptions is examined using a two-state variable model, that distinguishes between primary and secondary forests. First, the supposedly adverse impact of shifting cultivation for conservation of tropical forests is addressed. With a two state variable model, we demonstrate that (the threat of) encroachment acts as a "natural brake" on the pace at which concessionaires open up primary forest areas. Hence, while encroachment may inflict severe ecological damage on secondary forest areas, it may also promote conservation of virgin forests. Second, conventional wisdom implies that high discount rates accelerate depletion of tropical forests. We show that this result does not necessarily hold in a two state variable model.

3.1 Introduction

The rate of deforestation and forest degradation in tropical countries caused by human interference has been high and increasing over the past two decades. According to FAO (cited in Amelung and Diehl, 1992), deforestation, defined as the total removal of tree cover, has increased from 0.6% per year in the second half of the 1970s to 0.8% in the 1980s. This would indicate an increase in the deforestation rate of 30% in a decade. Myers (1994) states that the deforestation rate reached 1.8% in 1991, and he also

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[1]This chapter is based on two published articles: (i) Bulte E. and D. van Soest (1996), Tropical Deforestation, Timber Concessions and Slash-and-Burn Agriculture-Why Encroachment May Promote Conservation of Primary Forests, *Journal of Forest Economics* 2, pp. 55-66, and (ii) Bulte E. and D. van Soest (1996), A Note on High Discount Rates and Depletion of Primary Forests, *Journal of Agricultural and Resource Economics* 21, pp. 341-350.

asserts that the deforestation rate in 1994 must be even higher. In economic literature, encroachment by shifting cultivators and high discount rates applied by commercial logging firms are often noted as important causes of excessive deforestation. In this chapter, the validity of these claims is investigated.

Several types of economic activities are important causes of the depletion of forest resources: agriculture (shifting cultivation, permanent agriculture, cattle ranching), logging, mining and generation of hydropower. Of these causes, the agricultural sector is generally believed to be the single most important contributor to tropical deforestation: shifting cultivation alone is probably responsible for about 40 to 60% of total deforestation (Lanly 1982; Brünig 1989). Myers (1994) claims that shifting cultivators were responsible for 61% of total deforestation in 1989 and that this share appears to be rising over time. The timber industry is responsible for about 10% of total deforestation (Lanly 1982; Brünig 1989). As one course of action, many authors have concluded that encroachment must be stopped if efforts to combat deforestation are to be successful. Myers (1994) elegantly summarizes:

"*A broad based approach is needed to overcome the economic, social, political and institutional marginalization of the shifted cultivator which would involve the redistribution of existing farmlands, reform of land tenure systems, build up of agricultural extension services, improvement of credit facilities and provision of agrotechnologies. ... The source problem is an amalgam of non-forestry problems, ... hence the overall problem [of deforestation] must be tackled largely through non-forestry measures*" (Myers 1994, p. 40).

In this chapter we examine the validity of two pieces of conventional wisdom with respect to conservation of tropical forests. Using a two state variable model, we analyze whether (i) shifting cultivation implies nothing but bad news for nature conservationists, and (ii) whether high discount rates are necessarily detrimental for nature conservation. Although the models we develop below are still far from perfect (and difficult to solve analytically), it is our opinion that they provide a more realistic starting point for analysis than single state models that do not distinguish between primary forest areas and secondary forests.

As mentioned in section 2.6.2, and more elaborately discussed below, the differences between primary and secondary forests, and especially the difference in accessibility for shifting cultivators, is of crucial importance in the deforestation process. Primary forests are untouched forests, where trees have matured fully and the timber extracted from these forests can be sold at high prices in international markets. Primary forests have no net growth: they are in ecological equilibrium where growth equals decay. By definition, primary forests are turned into secondary forests if they are selectively logged (see for example, Kummar and Sham 1995). In contrast to primary forests, secondary forests display net growth because half-grown trees are exposed to more sunlight and face less competition for other scarce resources.

It is widely acknowledged that the issues of encroachment and commercial, selective logging are not independent. Logging, for instance, increases the attractiveness of agricultural activity in the tropical forest area by providing access to previously inaccessible areas. The existence of a road network facilitates travelling into the forest and increases potential agricultural rents because of the increased possibilities to transport agricultural surpluses to local and regional markets (Grut 1990; Grut *et al.* 1991; Horta 1991; Jepma 1993; Southgate *et al.* 1991). Furthermore, land clearing costs are lower on logged-over forest lands, because loggers have already removed some of the larger trees (Panayotou and Sungsuwan 1994). For these reasons, some authors argue that logging encourages forest destruction through encroachment. In the words of Amelung and Diehl (1992):

"*The opening up by the forestry sector can only be regarded as the main source of forest disturbance, if it was clear that otherwise potential users face prohibitive costs of entering virgin forests. ... In countries, in which the share of shifting cultivators in deforestation is high, logging can be considered as a necessary first step of destruction by opening up forest areas*" (Amelung and Diehl 1992, p. 120).

There is empirical evidence supporting the claim that shifting cultivators use the logging sector to gain access to primary forest areas. Although the rate of deforestation caused directly by the logging sector is generally believed to be small (as indicated above), logging is the primary cause of forest modification.[2] More than 70% of the primary forest areas brought under exploitation are first degraded by the commercial logging sector (Amelung and Diehl 1992). Furthermore, according to the FAO (cited in Sun 1995), deforestation rates due to agricultural conversion are eight times greater in logged-over forests than in undisturbed forests. Barbier (1994) reports that in many African countries around half of the area that is initially logged is subsequently deforested, while there is little, if any, deforestation of previously unlogged forest land.

In the first part of this chapter (section 3.2) we analyze the differences between conservation of secondary and conservation of primary forests. More specifically, we demonstrate that a trade-off may exist between the two issues, when encroachment is included in the analysis. Obviously, the only sensible way to model shifting cultivation is with aid of a model that distinguishes between primary and secondary forests.

In the second part of this chapter, using a slightly different model, we focus on the supposedly adverse effects of high discount rates for nature conservation. For this model, apart from the difference in accessibility for shifting cultivators, we focus on another difference between primary and secondary forests: the latter category displays net growth whereas the first category does not. Conventional wisdom implies that high discount rates discourage sustainable forestry. High discount rates typically accelerate harvesting, depress investments in sustainable resource management, and reduce the weight attached to the needs and desires of distant future generations. In the economic literature, the process of discounting has been labelled ethically indefensible (Ramsey 1928), a polite expression for rapacity (Harrod 1948) and is believed by some to advance doomsday (Koopmans 1974). At first sight, the higher the discount rate, the more pronounced the negative impact on intergenerational equity and sustainability. Since it has been argued

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[2]Forest modification is defined as the conversion of virgin forests into productive closed forests or other forms of land use (Amelung and Diehl 1992). Forest modification may be due to selective logging.

by many that current deforestation is excessive and socially wasteful (e.g., Barbier *et al.* 1994; Brown and Pearce 1994),[3] and since intertemporal harvesting paths are for an important part driven by discount rates (Hotelling 1931; Dasgupta and Heal 1979), a logical conclusion would be that the timber industry applies a discount rate that is too high.

However, the observation that high discount rates have adverse effects for intergenerational equity and sustainability is not generally valid in resource economics. Farzin (1984) demonstrated that if the backstop price of a resource is a function of the interest rate or if production is capital intensive, the depletion period of a mine may increase or decrease as the interest rate shifts. With respect to forestry, Price (1991) argued that if discount rates are lowered, the opportunity costs of investment funds are also reduced, thereby making forest exploitation more profitable. Hence lower discount rates may enhance the case for short-term exploitation of forests.

In section 3.3, we obtain similar results for tropical forestry without making any assumptions about the capital requirements of substitutes and without exploring the consequences of changes in the discount rate on the opportunity costs of investment funds. Instead, we analyze the influence of the discount rate on the way in which primary forests are converted into secondary forests. It is explicitly recognized that primary forests are converted into secondary forests by the timber industry and we model the interlinkages in a simple two-state variable model.

An important assumption in the first model is that the commercial logger has imperfect control over the behaviour of shifting cultivators. The second model is driven by the assumption that the government (as the owner of the forest) has imperfect control over the harvesting decisions of the logging firm that is granted a concession for harvesting, such that the firm has a certain freedom to allocate its intertemporal supply. An important assumption that relates to both models is that logging firms maximize their profits in a competitive industry. We assume that all firms are equal, so we model the logging sector as a representative firm.

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[3] To evaluate whether harvesting is "excessive" or not, one needs to relate it to a benchmark. It is often unclear what this may be; invariably, subjective opinions will play a role.

This chapter is organised as follows. We develop a simple two state variable model of encroachment in section 3.2, providing a numerical solution to illustrate the relation between the harvesting of primary forests and encroachment. In section 3.3 we construct another two state variable model, but here we will disregard the complexities introduced by shifting cultivators (the second model is easily extended to incorporate the key features of the first model). The conclusions follow in section 3.4.

3.2 A Two State Variable Model

Tropical commercial logging typically involves a contract between the government, as the owner of the resource, and a private firm. The firm is granted the right to harvest and manage a certain parcel of forest for a specific period of time, a "concession". In principle, the concessionaire has the sole right to use the resource. In practice, however, control and monitoring costs to safeguard the area from encroachment (i.e., small scale agricultural conversion) are high. This implies that, once shifting cultivators have gained access to the forest, they will share its use. The effect of encroachment on logged-over forest areas is that under-aged trees or tree species that are currently not commercially interesting but potentially valuable, are removed. This implies that the forestry sector is confronted with a much younger, less valuable collection of trees for succeeding rounds of logging. For the concessionaire, this can be considered as a cost of opening up an area. Recognition of this will influence decision making. This effect will be illustrated more formally in this section using a simple model.

In order to highlight the effect of encroachment on degradation of secondary and primary forests, we make several assumptions. First, we assume that the impact of encroachment on the logger's decision can be adequately analyzed by evaluating the optimal depletion time of the primary forest (T_1) and the secondary forest (T_2), respectively. The interpretation is that postponing depletion is beneficial for nature conservationists because society can enjoy the stock and flow services provided by the forest for a longer time. Likewise, reducing the optimal depletion period corresponds with a less desirable situation. Obviously T_1 must be smaller than T_2.The specification of the objective function of the logger is as follows:

$$Max_y \int_0^{T_2} [P_1(t)\,y_1(t) + P_2(t)\,y_2(t)]\; e^{-rt}dt \tag{1}$$

where P_i, $i = 1,2$ indicates the price of wood from primary and secondary forests respectively, and y_i represents the quantity harvested in forest type i. In the model, harvesting of primary forests is expressed in hectares whereas harvesting of secondary forest is expressed in biomass units. Our focus is on the size of area of primary forests, whereas incorporating the growth potential of secondary forests makes representing this stock in terms of biomass more logical. Apart from the fact that P_1 is a price per hectare and P_2 is a price per biomass units, prices for wood from primary and secondary forests may also differ due to, for instance, different species composition (Grainger 1993) or differences in average stem size. The discount rate r is assumed constant.

The equations of motion of the model will be explained next. With respect to harvesting primary forest, for which net growth is negligible, the model is an extension of the standard mining model (Hotelling 1931, Dasgupta and Heal 1979). It is assumed that all harvesting takes the form of selective logging,[4] and that encroachment is induced by commercial logging. No encroachment is possible unless the forest is first subjected to selective harvesting so there is no encroachment on primary forests, which is consistent with some of the empirical evidence summarized in section 3.1.[5] Thus, the equation of motion of the stock of primary forest is simply:

[4]As mentioned above, the share of the logging sector in deforestation is relatively small. Since the forestry sector is by far the most important source of forest modification, the explanation must be that after the first round of logging there is still some sort of tree cover present. Grut (1990) even asserts that selective logging is "a regime not much more interventionist than the regime of nature itself".

[5]This assumption is perhaps somewhat heroic, but it facilitates the formal analysis. The results will not be affected qualitatively if we would mitigate this assumption and assume instead that there is a difference in the speed at which slash-and-burn cultivators encroach upon primary and secondary concession lands.

$$\dot{x}_1(t) = -y_1(t) \tag{2}$$

where $x_1(t)$ is the stock of primary forest, measured in hectares, and $y_1(t)$ is the area of primary forest selectively logged, also measured in hectares. The dot over a variable indicates a change in time, hence $\dot{x}$ represents a change in the size of the stock.

Now turning to the specification of the second equation of motion, we need to make the translation from the area of primary forests to quantity of biomass in secondary forests. For this purpose we multiply the area harvested in primary forests by a constant conversion factor γ. This constant is derived as follows. Assume that the timber volume per hectare of undisturbed forests equals ψ_1 units of trees, and that it is optimal for the firm to restrict harvesting to ψ_2 trees. Now, γ is given by (ψ_1-ψ_2). If the different age classes of trees are homogenously distributed over the total area (or if firms are restricted by laws or custom to harvest a certain fraction of the stand), ψ_1 and γ will be constant.

To incorporate encroachment in the model we arbitrarily assume that encroachment can be represented as a destructive process beyond the control of the forestry sector that devastates β% of the stock of secondary forest in every period. This is an uncommon assumption, for which no empirical support exists. However, since encroachment is restricted to secondary forest area (x_2), damage due to agricultural conversion (S) is likely to be linked to the size of the forest that has been opened up for access. For any other model specification with $\partial S/\partial x_2 > 0$ similar results can be obtained. The current specification is chosen for mathematical convenience. The second equation of motion of this two state variable model is:

$$\dot{x}_2(t) = \gamma y_1(t) - y_2(t) + (\rho - \beta)x_2(t) \tag{3}$$

where $x_2(t)$ is the stock of secondary forest measured in biomass units. Depleting the stock of primary forest implies accumulating a stock of secondary forest as is evident from the first term on the right hand side of (3). Next, $y_2(t)$ is the quantity of biomass harvested in the secondary forest and ρ is the (constant) growth rate of secondary forests. The assumption of a constant growth rate is for mathematical convenience and does not affect the qualitative results

Invoking the maximum principle and assuming an interior solution gives the following necessary conditions for an optimum solution:

$$P_1(t) + \gamma\mu(t) = \lambda(t) \tag{4}$$

$$P_2(t) = \mu(t) \tag{5}$$

$$\frac{\dot{\lambda}(t)}{\lambda(t)} = r \tag{6}$$

$$\frac{\dot{\mu}(t)}{\mu(t)} = r-\rho+\beta. \tag{7}$$

In these equations, $\lambda(t)$ and $\mu(t)$ are the costate variables, akin to Lagrange multipliers, that measure the shadow price of the associated state variable, and $\dot{\lambda}$ and $\dot{\mu}$ represent the derivatives of these shadow prices with respect to time. The interpretation of (4) is that the marginal benefits of harvesting a unit of primary forest, measured as the sum of direct revenues and future harvests as a secondary forest, are equal to the foregone future timber benefits from leaving it as primary forest. Equation (5) states that marginal timber benefits from secondary forest should equal marginal cost of foregone future timber benefits. Equations (6) and (7) are non-arbitrage conditions: (6) is simply the Hotelling rule and (7) is an extended version of this rule that accommodates the growth of the resource (ρ) and encroachment damage (β). When we assume that the inverse demand functions are linear, i.e. $P_i(t)=\overline{P}_i-\alpha_i y_i(t)$, then substituting (6) and (7) in (4) and (5) and solving the differential equations, we get:

$$P_1(t) - \left(\overline{P}_1 + \gamma\overline{P}_2 e^{(r-\rho+\beta)(T_1-T_2)}\right)e^{r(t-T_1)} + \gamma P_2(t) = 0 \tag{8}$$

$$P_2(t) - \bar{P}_2 e^{(r-\rho+\beta)(t-T_2)} = 0 \tag{9}$$

where $\bar{P}_i$, $i = 1,2$, is the backstop price for wood extracted from forest type i, which will be reached at $t = T_i$ because by definition at time T_i forest type x_i must be depleted and subsequently $y_i(T_i)$ equals zero. This also means that by integrating (2) and (3) and evaluating them at T_1 and T_2 respectively, we find:

$$x_1(0) = \int_0^{T_1} y_1(t)dt \tag{10}$$

$$e^{(\rho-\beta)T_2} \int_0^{T_2} [\gamma y_1(t) - y_2(t)]\, e^{-(\rho-\beta)t} dt = 0 \tag{11}$$

Note that by definition $x_2(0)$ equals zero. Substituting the linear inverse demand function in (8) and (9) and solving for $y_i(t)$, the optimal depletion periods can be derived (see appendix 1).

The model is complicated and analytically solving it in order to illustrate the relation between T_i and β proves to be extremely cumbersome. Therefore, we resort to a numerical solution. Representative results are presented in Figure 3.1. The optimal time of depletion is on the vertical axis and β is on the horizontal axis.

The following observations apply. First, in this model high discount rates (r) correspond with enhanced depletion of both primary and secondary forests. As is evident from Figure 1, optimal depletion periods for a high discount rate are always lower than for a lower discount rate (i.e., the T_i curve for r = 12.5% is located below the T_i curve for r = 10%). More importantly, however, is the trade-off between conservation of primary and secondary forests as encroachment increases: the higher β, the higher T_1 and the lower T_2. The T_1 path is an upward sloping function of β whereas the T_2 path is downward sloping.

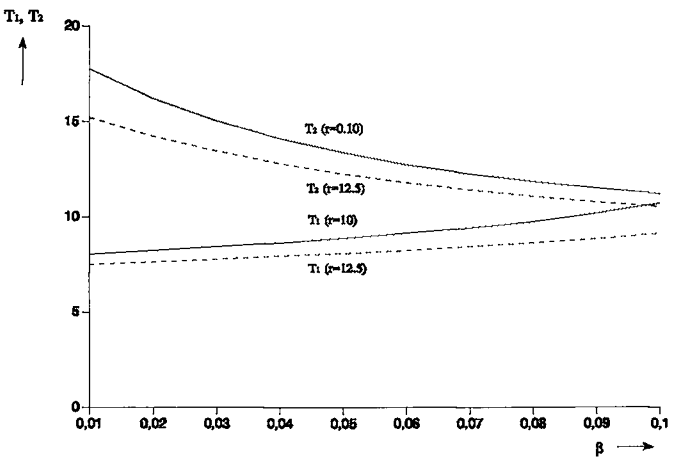

Figure 3.1: Optimal depletion as a function of encroachment (r= 10% and r=12.5%, $x_1(0)$=400, $\bar{P}_1$=40, γ=2, $\bar{P}_2$=20, α_1 =0.3, α_2=0.15.).

The interpretation is that concessionaires want to avoid losing part of their stock to shifting cultivators. Encroachment thus can be considered as a sort of property tax on "owning" logged-over forest areas. Cutting back on encroachment losses can be achieved in two ways. First, by reducing the "supply" of secondary forest by harvesting less primary forest. Encroachment acts as a "natural brake" on the rate of harvesting in primary forests: supply is postponed such that encroachment damage is discounted (hence, $\partial T_1/\partial \beta > 0$). Second, by intensified harvesting of secondary forest areas in order to outrace the shifting cultivators.

The effects are twofold. First, during the depletion period of the primary forest the area of this type of forest will be greater if the threat of encroachment exists. Hence during this period amenity values will be higher, and biodiversity is greater when compared to logging not constrained by encroachment. The second effect is that the quality and quantity of secondary forest areas will deteriorate if more shifting cultivators move to the area. The shifting cultivators themselves will cause more direct damage and in addition the forestry sector will respond by increased harvesting. This may cause a change in species composition and will reduce biomass per hectare. A trade-off exists between conservation of primary forest and conservation of the quantity and quality of secondary forest. Depending on the preferences of the international community with respect to nature conservation, different weights are given to these effects. This implies that, on balance, the destructive process of slash-and-burn agriculture can be considered either beneficial or detrimental.

2.4 A slightly different two-state variable model

Many governments in developing countries consider their forest base to be suboptimally large (Myers 1994) and develop forest zonage plans in which part of the forest area is designated as conversion forests. Conversion forests are destined to be cleared for alternative uses, such as agricultural cultivation (see, for example, the case of Cameroon, as described in Côté 1993). Land can be cleared from its forest cover by clearcutting the entire area as quickly as possible, or it can be transformed gradually such that the net present value of land use is maximized. Of course there are situations in which instantaneous clearcutting maximizes net present value of land, but this is certainly not generally true. This implies that during a gradual conversion phase two types of forests can be discerned; primary and secondary forests.

Suppose that the government owns an area of primary forest $x_1(0)$. If the benefits accruing to society from sustainable forestry are less than the benefits associated with alternative land use (for instance, agriculture), then eventually the forest cover will be cleared. The government will solve the following problem:

$$\max\ U = \int_0^T B(t)e^{-\delta t}dt + e^{-\delta T}\int_T^\infty A(\tau)e^{-\delta(\tau-T)}d\tau, \qquad \textbf{(12)}$$

subject to appropriate state equations, nonnegativity constraints, and the constraint that cumulative extraction from 0 to T equals the total forest stock at $t = 0$. In (1), $B(t)$ are (social) benefits derived from timber exploitation and

$A(t)$ are (social) benefits from the alternative land use option.[6] Finally, δ is the (constant) opportunity cost of capital, used by the government as discount rate. Solving the government's problem yields (*a*) an optimal intertemporal extraction path for the forest stock and (*b*) the optimal period T in which all forest cover is cleared.[7]

We assume that the government aims to achieve optimal deforestation as determined by the optimality conditions that follow from the government's problem by setting the terms of a concession contract with a logging firm. However, limited ability of governments in tropical countries to enforce concession contracts is well documented (for example, Grut, Gray and Egli 1991). An alternative interpretation is that the transaction or enforcement costs associated with enforcing full compliance exceed the benefits of compliance. For that reason we assume that the government is able to set the optimal "depletion time" of the forest stock T,[8] but due to limited capability of the government to monitor the firm's logging activities or output, the harvesting decisions of the firm cannot be fully controlled. The consequence is that the firm has a certain freedom to allocate intertemporal supply, which will result in an intertemporal harvesting path that is optimal for the firm (conditional on the pre-determined depletion time T) but not necessarily for the government. Discrepancies arise if, for instance, harvesting involves external effects or if the firm applies a different discount rate than the government. In the remaining part of this chapter we focus exclusively on the latter.

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[6]We can assume that the government solves for a harvest plan that is socially optimal. Alternatively, and without loss, we can assume that the government maximizes, for example, a political preference function (e.g. Becker 1974).

[7]User cost and its development over time is an important determinant of the optimal timing of switching to alternative land use. We refer to McConnell, Daberkow and Hardie for a model that determines (approximately) optimal harvesting when timber production eventually ends. Since we are interested in optimal management of the firm (which faces a given T) rather than the government, we do not deal explicitly with the complexities of solving the government's problem.

[8]Hence the timber industry is not allowed to harvest after T, and is not allowed to stop harvesting before T. A reason for the latter may be that the government wants to get a steady stream of revenues from forest exploitation. Because of imperfect monitoring, the government can only enforce that in each period a positive harvest takes place.

In a simple model, the discount rate of a private firm may be based on the opportunity cost of capital (δ) and a possible risk premium (σ). In the case of the timber industry in tropical countries, the latter may be a function of the security of its tenure rights (Deacon 1994, Mendelsohn 1991). We assume that the timber industry perceives a constant probability κ of losing its tenure rights due to hostile government policy. When $\sigma = -\ln(1-\kappa)$, then $e^{-\sigma t}$ is the probability of having control over the stock at time t. We define $\sigma + \delta = r$. Due to this risk premium the government and firm will prefer different extraction paths, even if they have the same opportunity cost of capital.

Now the firm's optimization problem can be sketched. For convenience, we do not include encroachment on secondary forests in this model, but extension in this direction is straightforward. The firm has agreed to deforest an area of mature, non-growing primary forest ($x_1(0)$) in T years. Suppose that it is optimal for the firm to log the primary forest selectively such that trees with small diameters are allowed to grow and reach commercially (more) profitable stem sizes. Selective logging turns the primary forest into a forest with net growth. The firm's problem consists of two linked subproblems: (*a*) with respect to the primary stock, an optimal extraction path and *depletion* time (denoted by T_1) must be solved for; and (*b*) with respect to the stock of secondary forest, an optimal extraction path and *starting* time (denoted by T_2) must be solved for. Obviously these problems must be solved simultaneously. Note the difference with the previous model, where T_2 referred to the depletion time of the stock of secondary forest. Due to the set-up of the model, and more specifically the transversality condition that every hectare must be cleared from its (secondary) forest cover at T, the firm's subproblem is concerned with finding the starting time that maximizes net present value of exploitation, rather than the optimal depletion time. Of course it is possible to harvest primary forest and secondary forest at the same time (though not on the same hectare, obviously), as T_1 can be arbitrarily near T and T_2 can be arbitrarily near zero.

Formally, the objective function of the firm is specified as follows:

$$\max_{y_1, y_2} \Pi = \int_0^T [P_1(t) y_1(t) + P_2(t) y_2(t)] \, e^{-\delta t} \, e^{-\sigma t} dt, \tag{13}$$

where Π indicates the present value of the profit stream, T is the concession period as determined by the government, $y_i(t)$, $i = 1,2$ indicates harvesting in primary and secondary forest in period t respectively, and $P_i(t)$ represents the (net) price of wood from forest type i in period t. Again, harvesting in primary forests is measured in hectares logged whereas harvesting of the growing secondary forest is expressed in cubic meters of wood.

Prices are assumed to be net of extraction costs, and revenues are net revenues. If marginal extraction costs are constant, this assumption is harmless, but even with marginal extraction costs that are not constant the qualitative results are generally not affected. However, there is one aspect of ignoring extraction costs that is potentially restrictive in this model: if there are significant economies of scale in harvesting, it may be more attractive for firms to clearcut the entire tree cover and save on exploitation costs, than to harvest selectively and benefit from forest growth for future harvesting. In the remainder of this article we assume that possible economies of scale in exploitation are outweighed by the benefits from harvesting additional forest growth after selective logging, such that selective logging is optimal for the firm. If scale economies dominate growth benefits and we maintain the condition that output should be strictly positive from 0 to T, interior solutions for the firm's problem may be infeasible.[9]

In order to make this an interesting problem, we assume that the firm faces a downward sloping, inverse demand function for wood: $\partial P_i / \partial y_i < 0$. More specifically, in the numerical solution we will assume that the inverse demand function is linear: $P_i(t) = \bar{P}_i - \alpha_i y_i(t)$. In the absence of extraction costs and with constant prices, the representative logger's optimal decision when to remove all commercially interesting trees would simply be to deplete

[9]Suppose that the terminal point of the optimal extraction path ($y_I(T)$) and the starting time ($t = 0$) are specified. Then, at most 1 extraction path will satisfy the conditions that (*a*) $\int_0^T y_I(t)dt = X_I(0)$, (*b*) $y(t) > 0$, $\forall t \in [0, T]$, and (*c*) the necessary non-arbitrage condition that describes the development of the co-state variable over time [see equation 7]. This path may be optimal, depending on the discount rate applied by the firm, but most probably it won't be. Changing the firm's discount rate and evaluating the effect on deforestation makes no sense in this context.

the mature stock in the first period and benefit from the growth potential of secondary forests in all periods that follow.[10]

The equation of motion for the stock of primary forest is equal to the one of the first model [see (2)]. The equation of motion for the stock of secondary forests differs slightly from the previous one [see (3)], because encroachment is not considered here:

$$\dot{x}_2(t) = \gamma y_1(t) - y_2(t) + g(x_2(t)). \tag{14}$$

In this equation $x_2(t)$ is the stock of timber in cubic meters available in secondary forest, $y_2(t)$ is the number of cubic meters of timber harvested and $g(x_2)$ describes secondary forest growth. Invoking the maximum principle and assuming an interior solution gives (4), (5), (6) and the expression below as necessary conditions for an optimal solution:

$$\frac{\dot{\mu}(t)}{\mu(t)} = r - g'. \tag{15}$$

The interpretation of these necessary conditions is similar as before [see section 3.2], and (15) is again an extended version of the Hotelling rule that accommodates growth of the resource. After substituting the solutions of (6) and (15) into (4) and (5), we find

$$P_1(t) - \left(\bar{P}_1 + \gamma \bar{P}_2 e^{(r-g')(T_1-T)}\right) e^{r(t-T_1)} + \gamma P_2(t) = 0, \textit{ and} \tag{16}$$

$$P_2(t) - \bar{P}_2 e^{(r-g')(t-T)} = 0, \tag{17}$$

where $\bar{P}_i$, $i = 1,2$ is the backstop price for wood extracted from forest type i, which will be reached at $t = T_1$ for primary forest and $t = T$ for secondary forest because, by definition, at T_1 and T respectively primary and secondary

[10]Note that, without loss, a so-called bang-bang solution can also be avoided when we model harvesting costs explicitly and assume $\partial C_i / \partial y_i > 0$.

forest stocks must be depleted. Harvesting the stock of secondary forest starts at time T_2. By integrating (3) and (14) we find

$$x_1(0) - \int_0^{T_1} y_1(t)dt = 0, \textit{ and} \tag{18}$$

$$e^{g'T} \int_0^{T} (\gamma y_1(t) - y_2(t))e^{-g't}\, dt = 0. \tag{19}$$

Solving this model we obtain the optimal depletion time of the primary forest T_1 and the starting period of secondary forest extraction T_2 [see appendix 2; we have chosen a linear growth function ($g(x_2) = \rho x_2$) because it facilitates the mathematics considerably without affecting the qualitative results].[11] Next, the optimal depletion paths for primary and secondary forests can be derived. The response of T_1 to changes in r is the subject of analysis in this section. The discount rate r can change, for instance, because the firm perceives its tenure security differently as time passes. There are numerous reasons why (the perception of) tenure security may change over time. For example, the firm may fear that the present political elite loses its dominant position, through elections or a coup d'etat, and that the new government will renege on the contract (for a discussion and empirical study of tenure security, see Deacon 1994). Alternatively, political preferences of the government may be subject to change over time. This latter will especially be important if government decision making is in accordance with public choice theories (e.g., Mueller 1989) and not so much aimed at achieving a socially optimal solution.

[11]With this specification, even though $g(x)$ is a function of γ, it is easily verified that $g'(x)$ and γ are independent terms. This is clearly a violation of standard biological relationships central to selective tree harvesting. However, since the focus is on the transformation of non-growing primary forest into growing secondary forest, this simplification is harmless. It is recognised that the present model specification is not suitable to analyze, for instance, optimal thinning in the secondary forest.

If the government does not want to renegotiate the concession contract because the opportunity cost of capital has not changed (hence, T is constant), the firm will alter the intertemporal allocation of supply and redefine its optimal T_1 using the procedure outlined above.[12]

Given the mathematical results as presented in appendix 2, the effects of an increase in the rate of discount r on the optimal depletion time of primary forests T_1 and on the optimal time at which the firm starts logging secondary forests T_2 can be derived. The model is complicated and analytically solving it in order to illustrate the relation between T_1 and r again proves to be difficult. Therefore, we resort to a numerical solution. We have arbitrarily selected values for the parameters of the inverse demand functions and for $x_1(0)$ and T, as reported in figure 3.2. Representative results are presented in figure 3.2.

As is clear from this figure, the higher the rate of discount, the more logging in secondary forests is postponed while the effect on the depletion period of primary forests is less clearcut; the results are presented for two different values of γ but are robust for other parameter values (as long as the nonnegativity constraints are not violated). The fact that a higher γ leads to lower optimal values of T_1 and T_2 can easily be explained by analyzing equation (14). Ceteris paribus, an increase in γ (which may correspond with a high initial stocking density) will raise the marginal benefits of primary forest exploitation because the investment aspect of harvesting, hence the role of converted primary forest as an input in the secondary forest production process, gains weight. In order to benefit more from the stock of secondary forest the concessionaire wants to accelerate access to this stock. Hence, the higher the conversion factor γ the shorter the optimal rotation of the primary forest, and logging in secondary forests can start at an earlier date.

Second, and more important, the curve that relates optimal depletion to the discount rate is not monotonically declining but has an inverted U-shape. This implies that there exists a range of r-values for which raising the discount rate actually postpones depletion of the primary forest. The reason is

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[12]In fact, the model we use does not investigate the impact of a change in r after a couple of years of harvesting but, instead, determines optimal exploitation paths from period 0 onwards.

that a high discount rate "tilts" the price paths and optimal exploitation paths of both the primary and the secondary forest stock. With respect to the primary forest, on the one hand, this results is an incentive to harvest more in early periods, which is the standard Hotelling result. On the other hand, the shift in the price path of the stock of secondary wood has a countervailing effect on optimal exploitation of the virgin stock. The reason is as follows. Raising r while keeping T and $\bar{P}_2$ fixed implies that the new price path for secondary wood ($P'_2(t)$) will be steeper and necessarily located entirely below the original price path. If the growth rate of $P_2(t)$ increases and the terminal point is identical, then automatically the starting point of the price path must be lower. Hence,

$$\dot{\mu}'(t) > \dot{\mu}(t) \quad \wedge \quad \mu'(t) < \mu(t) \ , \quad \forall t \in [0,T]. \tag{20}$$

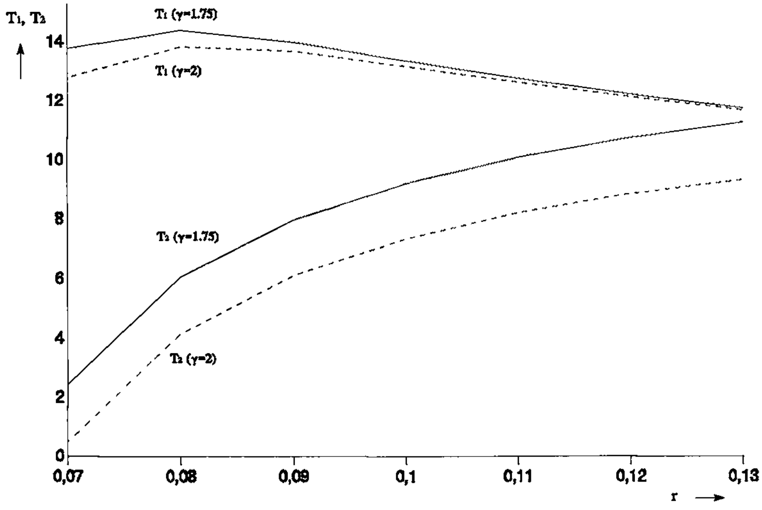

Figure 3.2: Optimal depletion times for γ=1.75 and γ=2 (ρ=0.05, $x_1(0)$=550, T=50, $\bar{P}_1$=30, $\bar{P}_2$=15, α_1=0.3, α_2=0.15)

The implications are as follows. From (4) it is clear that the low level of $\mu(t)$ reduces the marginal benefits of converting primary forest into a productive asset. This slows down the optimal extraction rate of the stock of virgin forest, which explains the inverted U-shaped T_1 path as shown in figure

1. Perhaps this explains why empirical support for the hypothesis that high risk premiums should accelerate deforestation, provided by Deacon (1994, table 4 and 5, p.424), is weak.

From the inverse demand function we know that low realizations of $\mu(t)$ ($= P_2(t)$) correspond with a relatively large supply of secondary wood. To satisfy condition (19), this means that T_2 must be shifted to the future, as the increase in supply per period must necessarily be compensated for by a reduction in the number of periods in which timber is actually supplied. Thus supply is increased and prices are depressed, but harvesting the secondary stock starts later.

3.4 Conclusions

In this chapter we have demonstrated that well established theoretical results, obtained with single state forest models, may lose their validity in the context of a more realistic model with primary and secondary forests and imperfect control.

First, encroachment by shifting cultivators on tropical forest concessions is generally considered a primary cause of deforestation. Drastic measures have been proposed in the literature to reduce the number of people who appear to depend on forest resources. Here we demonstrate that destructive conversion of forest areas for agricultural purposes is only part of the story. Under the assumption that encroachment is confined to accessible logged-over forest areas, encroachment has similar effects as a property tax on owning secondary forests. If the damage due to encroachment is positively related to the size of the secondary forest area, hence agricultural conversion increases as the area of accessible forest increases, then concessionaires will respond by reducing the rate at which they harvest primary forests. This implies that the net effect of encroachment is theoretically ambiguous and needs to be empirically determined. Whether the moderating effect of encroachment on harvesting of primary forests described in this paper will be significant in practice remains an open question. The braking power of squatters is probably small relative to the overall desire of logging firms to enter new areas. Also, if loggers do not consider successive rounds of logging, for instance because concession rights are defined for a short period or because primary forests are more profitable to exploit and abundantly available, the "natural brake effect" will be negligible.

Second, it is well documented that high discount rates are detrimental for natural resource conservation. If supply is restricted, for instance because of a tropical timber concession contract, this general conclusion no longer holds. If the concession period is exogenously determined and fixed and we recognize that depleting a stock of primary forest implies building a stock of secondary forest, then the effect of high discount rates on the stock of primary forest is ambiguous. There is a range of r-values over which an increase in the discount rate actually postpones depletion. The reason is that the marginal benefits of converting primary forests into secondary forests (a growing asset) are reduced. This result follows directly from the revised non-arbitrage conditions for the firm. In addition, the effect of higher discount rates on the stock of secondary forest is that the first period of exploitation is shifted to the future for all r. Whether this phenomenon is likely to occur in reality depends on the strength of the government to enforce concession contracts.

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Appendix 1:

From the first order conditions, the optimal extraction paths $y_1(t)$ and $y_2(t)$ can be determined:

$$y_1(t) = \frac{1}{\alpha_1}\left[\bar{P}_1 - [\bar{P}_1 + \gamma\bar{P}_2 e^{(r-\rho+\beta)(T_1-T_2)}]e^{r(t-T_1)} + \gamma\bar{P}_2 e^{(r-\rho+\beta)(t-T_2)}\right]$$

$$y_2(t) = \frac{1}{\alpha_2}\left[\bar{P}_2 - \bar{P}_2 e^{(r-\rho+\beta)(t-T_2)}\right]$$

The two equations which simultaneously determine T_1 and T_2 are derived by solving equations (10) and (11), using the optimal extraction paths derived above. Solving (10) yields:

$$\alpha_1 x_1(0) = \bar{P}_1 T_1 - \frac{1}{r}\left[\bar{P}_1 + \gamma\bar{P}_2 e^{(r-\rho+\beta)(T_1-T_2)}\right]\left[1 - e^{-rT_1}\right] + \frac{\gamma\bar{P}_2 e^{-(r-\rho+\beta)T_2}}{(r-\rho+\beta)}\left[e^{(r-\rho+\beta)T_1} - 1\right]$$

Furthermore, solving (11) yields:

$$\frac{\alpha_1\bar{P}_2}{\alpha_2\gamma}\left[\frac{1}{\rho-\beta}\left(1 - e^{-(\rho-\beta)T_2}\right) - \frac{e^{-(r-\rho+\beta)T_2}}{r-2\rho+2\beta}\left(e^{(r-2\rho+2\beta)T_2} - 1\right)\right]$$

$$= \frac{\bar{P}_1}{\rho-\beta}\left(1 - e^{-(\rho-\beta)T_1}\right) + \frac{\gamma\bar{P}_2 e^{-(r-\rho+\beta)T_2}}{r-2\rho+2\beta}\left[e^{(r-2\rho+2\beta)T_1} - 1\right]$$

$$- \left[\bar{P}_1 + \gamma\bar{P}_2 e^{(r-\rho+\beta)(T_1-T_2)}\right]\frac{e^{-rT_1}}{r-\rho+\beta}\left[e^{(r-\rho+\beta)T_1} - 1\right]$$

By solving these two equations simultaneously in GAMS, the optimal depletion times of primary and secondary forests are determined.

Appendix 2:

In order to determine T_1 and T_2, equations (11) and (12) must be solved. First, we have to determine the optimal paths of harvesting in primary and secondary forest areas. These paths can be derived by inserting the inverse demand functions into equations (9) and (10), and then solving them for $y_1(t)$ and $y_2(t)$:

$$y_1(t) = \frac{1}{\alpha_1}\left[\bar{P}_1 - [\bar{P}_1 + \gamma \bar{P}_2 e^{(r-\rho)(T_1-T)}]e^{r(t-T_1)} + \gamma \bar{P}_2 e^{(r-\rho)(t-T)}\right],$$

and

$$y_2(t) = \frac{1}{\alpha_2}\left[\bar{P}_2 - \bar{P}_2 e^{(r-\rho)(t-T)}\right].$$

Now the integrals (11) and (12) can be derived using the optimal depletion paths derived above. The result for equation (11) is

$$\alpha_1 x_1(0) = \bar{P}_1 T_1 - \frac{1}{r}\left[\bar{P}_1 + \gamma \bar{P}_2 e^{(r-\rho)(T_1-T)}\right]\left[1 - e^{-rT_1}\right] + \frac{\gamma \bar{P}_2 e^{-(r-\rho)T}}{(r-\rho)}\left[e^{(r-\rho)T_1} - e^{(r-\rho)T_2}\right].$$

The result for equation (12) is:

$$\frac{\alpha_1 \bar{P}_2}{\alpha_2 \gamma}\left[\frac{1}{\rho}\left(e^{-\rho T_2} - e^{-\rho T}\right) - \frac{e^{-(r-\rho)T}}{r-2\rho}\left(e^{(r-2\rho)T} - e^{(r-2\rho)T_2}\right)\right]$$

$$= \frac{\bar{P}_1}{\rho}\left(1 - e^{-\rho T_1}\right) + \frac{\gamma \bar{P}_2 e^{-(r-\rho)T}}{r-2\rho}\left[e^{(r-2\rho)T_1} - e^{(r-2\rho)T_2}\right]$$

$$- \left[\bar{P}_1 + \gamma \bar{P}_2 e^{(r-\rho)(T_1-T)}\right] \frac{e^{-rT_1}}{r-\rho}\left[e^{(r-\rho)T_1} - 1\right].$$

T_1 and T_2 are determined simultaneously by these two equations.

CHAPTER 4

INTERNATIONAL TRANSFERS AND TROPICAL DEFORESTATION[1]

Abstract

International transfers to developing countries are sometimes advocated on the ground that they contribute to tropical forest conservation. Here we demonstrate that the effectiveness (and attractiveness) of this instrument is reduced if the restrictive assumption of deterministic prices is relaxed. With stochastic prices, the steady state forest stock in the absence of transfers is greater, implying that the marginal benefit of additional hectares conserved is less. In addition, the 'wealth effect' of transfers counteracts the so called 'freeing up effect', which implies that per unit of subsidy less hectares of tropical forest is protected. Both effects reduce the attractiveness of transfers as a policy instrument for Western governments to combat tropical deforestation.

4.1 Introduction

International concern about tropical deforestation has resulted in an increased focus on timber trade interventions and other policies aimed at the conservation of forest stocks. In a recent paper, Barbier and Rauscher (1994) argue that, to conserve tropical forests, international (lump sum) capital transfers are to be favoured over trade interventions. More specifically, they assert that "a direct international transfer will increase the long run equilibrium forest stock unambiguously". This result is obtained by balancing marginal benefits of exploitation (and subsequent consumption) and conservation.

The driving mechanism in their model is that imports must be paid for with foreign exchange, earned by selling tropical timber. Foreign aid eases

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the stringency of the foreign exchange constraint. This means that more imports can be bought, which lowers the marginal utility from imported goods. In equilibrium, the marginal utility of forest conservation must also fall. This corresponds with a larger steady state forest cover. Capital transfers may also contribute to forest conservation because they may alleviate poverty. The effect of poverty on environmental degradation is described by, for instance, Jepma (1995) and Pearce and Warford (1993). A justification for providing international transfers are the global externalities associated with tropical forest conservation, such as storage of carbon dioxide and preservation of biodiversity.

The fact that transfers increase the size of the steady state of tropical forest does not justify the actual donation of funds - it is also a matter of marginal benefits and costs for the donor country. This paper aims to extend the model of Barbier and Rauscher (1994) by allowing for uncertainty with respect to future timber prices and risk averse policy makers. Wood prices have fluctuated widely in the past (World Bank 1991) and risk aversion is the rule rather than the exception, implying that this extension is relevant. We find that the marginal benefits of providing transfers are typically lower (and may approach zero) under these conditions, hence the likelihood that it is optimal to actually provide transfers to tropical countries is reduced.

4.2 The model

We assume that there are substantial unpredictable short-run fluctuations in timber demand that result in varying short run prices. Future prices are assumed to be drawn randomly from a stationary distribution. For mathematical convenience we assume that the price distribution is adequately represented by a uniformly distributed density function, defined as:

$$f(P) = \begin{cases} \dfrac{1}{P_u - P_l} & \text{for } P_l \leq P \leq P_u \\ 0 & \text{otherwise,} \end{cases} \tag{1}$$

where P_l and P_u are respectively the lowest and the highest wood price feasible. Suppose the government of a tropical country aims to maximize its expected utility derived from consumption of imported goods (c) and the size of its forest area (N). Imports must be paid for with foreign exchange, earned from sales of timber products (q is the quantity of wood sold in international markets at the prevailing price P), or with international capital transfers (s). The constrained dynamic optimization problem is:

$$MAX_q \ E \int_0^{\infty} U(c(t),N(t))e^{-rt}\,dt \tag{2}$$

subject to:

$$\begin{aligned} P(t)q(t) + s(t) &= c(t), \\ \dot{N}(t) &= g(N(t)) - q(t). \end{aligned} \tag{3}$$

In the equation of motion, $g(N)$ represents growth of the forest stock. If the government is risk averse, the utility function must be concave, hence $U_i > 0$, $U_{ii} < 0$, for i = c, N. A specification that satisfies these requirements is[2]

[2]Because we require explicit solutions of the model, our model differs from the Barbier and Rauscher model in two respects. In the first place, for mathematical simplicity, we ignore domestic use of timber. In the second place, we have specified a utility function explicitly. It could be argued that the current choice of the utility function is restrictive because the government's relative preferences for forest conservation and consumption are assumed constant, while it is possible that relative preferences for nature change as the level of wealth changes. The qualitative conclusions are not affected by this assumption. Also, for the main qualitative results it does not matter whether the utility function is separable or not.

$$U(c,N) = c^{\alpha}N^{\beta}. \tag{4}$$

It is assumed that α and β are positive, and smaller than 1. An important feature of the current specification is that $-U''(c)/U'(c) = -(\alpha-1)/c$, which implies that the Arrow-Pratt measure of absolute risk aversion is declining in the arguments of the utility function (see for example Varian 1992).

Applying the maximum principle and assuming an interior solution gives the following necessary first order conditions (suppressing time subscripts):

$$E\left[\frac{\partial U}{\partial q}\right] = \lambda, \tag{5}$$

$$\dot{\lambda} = (r-g'(N))\lambda - E\left[\frac{\partial U}{\partial N}\right]. \tag{6}$$

where λ(t) is the shadow price of the state variable, or co-state variable. Condition (5) states that the expected marginal utility derived from selling one unit of timber today should equal future timber revenues foregone. Condition (6) is an extended version of Hotelling's rule, stating that the rate of change in the shadow price is equal to the difference between opportunity costs of holding on to a unit $(r - g'(N))\lambda$, plus expected marginal social value of that unit as standing stock $\partial U/\partial N$.

Substituting (3) into (4) gives an expression for the expected marginal utility of harvesting a unit of timber:

$$E\left[\frac{\partial U}{\partial q}\right] = \int_{P_l}^{P_u} f(P)\,\frac{\partial U}{\partial q}\,dP = \int_{P_l}^{P_u} \frac{1}{(P_u-P_l)}\left(\alpha P(c(P)^{\alpha-1})N^{\beta}\right) dP. \tag{7}$$

Furthermore, expected marginal utility of the forest stock is given by:

$$E\left[\frac{\partial U}{\partial N}\right] = \int_{P_l}^{P_u} f(P)\,\frac{\partial U}{\partial N}\,dP = \int_{P_l}^{P_u} \frac{1}{(P_u - P_l)}\left(\beta c(P)^{\alpha} N^{\beta-1}\right) dP. \tag{8}$$

Solving (7) and upon substitution into (5) gives:

$$\lambda = \frac{N^{\beta}}{(P_u - P_l)q}\left[P_u c_u^{\alpha} - P_l c_l^{\alpha} - \frac{1}{q(\alpha+1)}\left(c_u^{\alpha+1} - c_l^{\alpha+1}\right)\right]. \tag{9}$$

Obviously, c_u and c_l correspond with $P_u q + s$ and $P_l q + s$, respectively. Substituting (9) and (8) into (6), and setting all time derivatives equal to zero, we obtain an implicit expression for the equilibrium stock size:

$$\left(r - g'(N_s^*)\right)N_s^*\left(P_u c_u^{*\alpha} - P_l c_l^{*\alpha}\right) = \left[\frac{\beta}{\alpha+1} + \frac{(r - g'(N_s^*))N_s^*}{q^*(\alpha+1)}\right]\left[c_u^{*(\alpha+1)} - c_l^{*(\alpha+1)}\right]. \tag{10}$$

Note that N_s^*, q_s^* and c_i^* denote the equilibrium values of the forest stock, rate of harvesting and consumption with stochastic prices.

The effect of an increase in the amount of the international transfers on the equilibrium size of the forest area can be found by comparative static analysis (see appendix 1):

$$\frac{dN_s^*}{ds} = \frac{-(r - g'(N_s^*))\alpha N_s^*\left[P_u c_u^{*(\alpha-1)} - P_l c_l^{*(\alpha-1)}\right]}{D} + \frac{\left[\beta + \dfrac{(r - g'(N_s^*))N_s^*}{q_s^*}\right]\left[c_u^{*\alpha} - c_l^{*\alpha}\right]}{D}, \tag{11}$$

in which D is the determinant of the system. When we compare this result to the one obtained by Barbier and Rauscher (1994) in the context of deterministic prices, it is clear that the solution becomes much more complex if prices are stochastic.

4.3 A comparison

Comparing the effect of international transfers under conditions of stochastic and deterministic prices requires that a similar procedure is followed for deterministic prices: $P(t) = \bar{P},\ t \in [0, \infty]$. For the equation that is used to derive the steady state forest stock it is easily verified that:

$$\left(r - g'(N_d^*)\right)\left[\frac{\alpha \bar{P}}{\bar{P} q_d^* + s}\right] = \frac{\beta}{N_d^*} \tag{12}$$

Similarly, the result from the comparative statics procedure is simply:

$$\frac{dN_d^*}{ds} = \frac{\beta}{\alpha \bar{P}[r - g'(N_d^*) - N_d^* g''(N_d^*)] - \beta \bar{P} g'(N_d^*)} \tag{13}$$

Analytically comparing these expressions with the solutions for stochastic prices proves to be cumbersome. For that reason we have resorted to a numerical comparison.[3] Representative solutions are provided in Tables 4.1 and 4.2. In these tables $P_u = \gamma P_l$, with $\gamma \geq 1$.

Table 4.1: Steady state forest stocks for deterministic and stochastic scenario. Parameter values: Pl=2, α=0.25, β=0.75, r=10%, ρ=0.05, K=10,000.

N^*	γ=2.5	γ=5	γ=7.5	γ=10
Stoch.	268.1	213.6	184.8	165.9
Det.	261.9	200.0	168.0	147.7

From Table 4.1 it is clear that the steady state forest stock under uncertainty always exceeds the steady state forest stock with deterministic prices. The reason is simply that uncertain prices (i.e. uncertain consumption) will reduce marginal utility of consumption for any risk averse policy maker.

[3] In order to be able to derive numerical solutions we assumed a logistic growth function, commonly specified as $g(N)=\rho N(1-(N/K))$. The deterministic and stochastic price models were programmed in GAMS. The model is available from the authors upon request.

In equilibrium, marginal utility of conservation must also fall, which corresponds with a larger forest stock. This is just an extension of the argument applied by Barbier and Rauscher (1994). Also, from Table 4.1 we learn that increasing γ reduces the steady state stocks for both the deterministic and the stochastic variant. The reason is that increasing γ while keeping P_l fixed increases the expected price, which tends to enhance deforestation.

Table 4.2: Changes in steady state forest stocks resulting from international transfers for deterministic and stochastic scenario. Parameter values: $P_l = 2$, $\gamma = 10$, $\alpha = 0.1$, $\beta = 0.9$, $r = 10\%$, $\rho = 0.05$, $K = 10{,}000$.

dN^*/ds	s=20	s=35	s=50	s=65
Stoch.	6.8	6.4	6.1	5.8
Det.	6.9	6.6	6.3	6.1

From Table 4.2 it is clear that, given a steady state N^* in the absence of transfers, the marginal gains from providing international transfers in terms of additional hectares protected are smaller with stochastic prices. As γ increases and/or α decreases, the difference between the stochastic and deterministic model becomes greater, and eventually, the marginal gain in steady state forest stock approaches zero. This implies that, give any positive marginal cost of providing transfers, the probability that it is optimal to provide transfers approaches zero. The interpretation is as follows. It is well documented that with stochastic prices, risk averse governments should harvest less than under conditions of deterministic prices (in a different context, this has been discussed by, for instance, Lewis 1977; Leland 1972; Sandmo 1971; Robison and Barry 1987). The reason is that (expected) marginal utility of exploitation is reduced when prices fluctuate. This 'brake' on the harvesting process is partly offset by international transfers. This is the 'wealth effect' or 'income effect' of international transfers; because a government has more wealth, it is willing to gamble a little more on timber revenues by increasing exploitation. Or, in the words of Robison and Barry (1987, p.76): "the total purchases of the risky asset increase (..) with increases in initial wealth as there is decreasing (..) risk aversion". The reason is that an increase in risk free wealth shifts up the EV set in a parallel fashion which decreases the equilibrium slope with the iso-expected utility curve.[4] This effect will be more pronounced for (i) low values of α, (ii) large values of s, and (iii) a large (mean preserving) spread P_u - P_l.[5]

This counteracting effect only arises with a declining absolute risk aversion (DARA) specification of risk aversion. If risk aversion is of the CARA type (i.e., constant absolute risk aversion), the degree of risk aversion is not affected by transfers, and transfers unambiguously increase the optimal stock size, as indicated by Barbier and Rauscher (1994). With IARA (increasing absolute risk aversion), the 'wealth effect' even re-enforces the 'freeing up effect', and increased transfers increase the stock size even more than predicted by Barbier and Rauscher (1994). However, arguing that increased wealth makes governments more risk averse is a bit odd, and not nearly as likely as the DARA case (for a discussion, see Robison and Barry 1987; and Blanchard and Fischer 1993).

4.4 Conclusions

If resource prices fluctuate and resource owners are risk averse of the DARA type, which are both quite reasonable assumptions, international capital transfers to developing countries to reduce tropical deforestation are a less effective instrument to combat deforestation than previously thought, or even not an effective instrument at all. The reason is that, because of increased wealth, governments become less prudent in their harvesting decisions. This effect weakens the 'freeing up' effect that predicts that, to restore equilibrium, harvesting of tropical forests will decline since transfers reduce the marginal benefits of consumption. In addition, under conditions of stochastic prices, the steady state of the forest stock is greater than the steady state stock with deterministic prices. This implies that the marginal utility of

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[4]The EV set describes efficient trade-offs between expected value and variance (see Robison and Barry, chapter 4 and 6).

[5]This also implies that the counteracting wealth effect is reduced as wood prices are stabilized, for instance as a result of establishing buffer stocks for tropical timber.

additional conservation is less with stochastic prices. Combining these effects leads us to the conclusion that the attractiveness of international transfers as an instrument to combat deforestation with stochastic prices is reduced.

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Appendix 1

The comparative static solution can be represented by:

$$\begin{bmatrix} A & -B \\ -1 & g' \end{bmatrix} \begin{bmatrix} dq_s \\ dN_s \end{bmatrix} = \begin{bmatrix} C \\ 0 \end{bmatrix} \begin{bmatrix} ds \end{bmatrix}$$

where:

$$A = (r-g')N_s\alpha\left[P_u^2 c_u^{\alpha-1} - P_l^2 c_l^{\alpha-1}\right] + \frac{(r-g')N_s}{q_s^2(\alpha+1)}\left[c_u^{\alpha+1} - c_l^{\alpha+1}\right] -$$

$$\left[\beta + \frac{(r-g')N_s}{q_s}\right]\left[P_u c_u^{\alpha} - P_l c_l^{\alpha}\right]$$

$$B = (r-g'-N_s g'')\left[\frac{c_u^{\alpha+1} - c_l^{\alpha+1}}{q_s(\alpha+1)} - P_u c_u^{\alpha} + P_l c_l^{\alpha}\right]$$

$$C = -(r-g')N_s\alpha\left[P_u c_u^{\alpha-1} - P_l c_l^{\alpha-1}\right] +$$

$$\left[\beta + \frac{(r-g')N_s}{q_s}\right]\left[c_u^{\alpha} - c_l^{\alpha}\right]$$

Note that in this appendix the asterix denoting equilibrium values have been suppressed. Solution can be derived through the application of Cramer's rule: $dN_s/ds = C/D$, where D is the determinant of the Hessian matrix:

$$D = g'A - B \qquad \text{(A2)}$$

The sign of this expression is positive. A necessary condition for an unique solution in the (q,N) space is that the curve $dq/dt = 0$ is positively sloped and cuts the curve $dN/dt = 0$ from below (see Barbier and Rauscher):

$$\left.\frac{dq}{dN_s}\right|_{q_s=0} > \left.\frac{dq_s}{dN_s}\right|_{N_s=0}$$

Hence, $|B| > |g'A|$. This means that the sign of the determinant D is the same as the sign of the term $-B$. The sign of determinant D is the same as the sign of the term $-B$. By rewriting B as:

$$Sign(D) = Sign(-B) = Sign\left\{P_u c_u^\alpha - P_l c_l^\alpha - \frac{c_u^{\alpha+1} - c_l^{\alpha+1}}{q_s(\alpha+1)}\right\}$$

$$= Sign\left\{(P_u q_s + s)^\alpha \left[\frac{\alpha P_u q_s - s}{q_s(\alpha + 1)}\right] - (P_l q_s + s)^\alpha \left[\frac{\alpha P_l q_s - s}{q_s(\alpha + 1)}\right]\right\}$$

it can be found that the sign of the determinant is strictly positive using the extra information derived from applying the second order condition (that is, the second derivative of the hamiltonian with respect to q must be negative).

CHAPTER 5

IVORY TRADE AND ELEPHANT CONSERVATION[1]

Abstract

To protect the elephant, trade in ivory has been banned since 1990. In this paper, we demonstrate that the effect of this ban on elephant populations is ambiguous. Depending on the discount rate, among other things, the trade ban can either increase or decrease optimal population size. If the decision maker's discount rate exceeds a certain threshold, the optimum stock of elephants is greater with a trade ban. Since the threshold is in the range of possible social discount rates, the success of the trade ban in promoting elephant conservation will vary spatially and temporally, depending on the characteristics and level of development of countries with significant elephant populations.

5.1 Introduction

During the 1970s and 1980s, elephant stocks were severely depleted. According to Chadwick (1992), "statistics showed a species on a toboggan ride towards absolute zero." African elephant populations are estimated to have declined from some 1.2-1.3 million animals in the mid 1970s to less than 800,000 by 1988 (Lindsay 1986; Harland 1988). Besides the large decline in elephant populations, scarce resources also appear to have been wasted as illustrated by the movement of ivory prices: in contrast to Hotelling-style models of efficient exploitation, real ivory prices have followed no trend, rising at an unprecedented rate in the 1970s and remaining high but variable thereafter (Milner-Gulland 1993).

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To halt the process of excessive depletion, the official status of the elephant as determined by the Convention on International Trade in Endangered Species (CITES) changed in 1989 from Appendix 2 (monitored trade) to Appendix 1 (trade only in special circumstances). This implied a world-wide ban on trade in ivory, thereby halting the (legal) killing of elephants for this purpose. Previous attempts in the framework of CITES to reduce ivory trade were less effective, as major loopholes in its application existed (Barbier et al. 1990, Milner-Gulland 1993).

Countries with relatively large elephant stocks, such as Zimbabwe, Botswana, South-Africa and Namibia, opposed this measure and have since lobbied to down-list the species, re-opening limited trade in ivory and other elephant products. Their main argument against the endangered species listing is that, in a number of areas, there are too many elephants and not too few, and that numbers need to be controlled to keep elephants from damaging agricultural lands and wildlife habitat. The ivory revenues can then be used for wildlife management and protection.[2] Barbier and Swanson (1990) have provided an economic argument for removing the trade ban, but it is based on a static model and intuition.

In this chapter, the impact of the trade ban on elephant products is evaluated using a dynamic optimization model that determines optimum elephant populations. We disregard potential effects on other species and poaching. These are important omissions because conservation policies for elephant and rhino tend to be inter-linked and because elephant poaching has proved to be important in the past.[3] Yet the model developed in this chapter

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[2]Zimbabwe, for example, spends US$12 million annually to counter illegal poaching of elephants, rhinoceroses and other animals (Chadwick 1992). Ivory trade generated US$10 million and, according to the Southern African countries, paid for most of the protection for all species. However, the link between wildlife management and protection is, in reality, no more than an assumption, although the link could be made real by designing appropriate institutions.

[3]Poaching is not modelled here because it is not clear how the trade ban affects consumer preferences for ivory—one cannot infer that banning legal sales promotes illegal activities. Poaching may decline as it becomes more difficult to market poached ivory, while, on the other hand, government incentives to combat poaching (provide funds for monitoring and enforcement) are smaller, which enhances poaching. Reports indicate that ivory prices were as low as $11/kg in March 1993 (Sack 1996; but see Bakker-Cole 1995), indicating that demand had shifted inward. Without adequate enforcement, harvesting elephants will take place under conditions

allows distillation of some recommendations for policy. The concept of an optimum population size (as perceived by the government) will, to a certain extent, determine the development of actual stocks over time. For example, when governments believe that the actual stock exceeds the optimal stock, it could decide to cut back on its enforcement efforts, as poaching of elephants appears to be strongly related to law enforcement (Anonymous 1994, Bakker-Cole 1995). Alternatively, excessively low stock levels (as measured against the optimum population) may induce governments to spend more resources on enforcement activities. In other words, preferences of host countries matter. Implementation of a trade ban affects both costs and benefits of "owning elephants". Implementation of a trade ban requires governments to re-assess their optimal stock size. In this chapter we show that for some countries, more specifically those with relatively high interest rates, efficient management could prescribe an increase in population size under a trade ban. Moreover, we find that the "critical discount rate"—where the elephant population with and without the trade ban is the same size—is reasonably close to the social discount rate for many countries. Since discount rates used by different African countries are likely to differ (and change over time), the effects of the trade ban on African countries may be diverse.

5.2 The model

Suppose a government wants to maximize revenues from elephant management. Elephants impose a financial cost because they damage agricultural crops (Pearce and Warford 1993) and possibly wildlife habitat (Chadwick 1992). Yet, they are a source of ivory benefits and revenues from eco-tourism. According to Pearce and Warford (1993), elephants play a flagship role in attracting tourists. The objective function for the government can therefore be written as follows (suppressing the time variable):

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resembling open access. The result will be rent dissipation and severe stock depletion (Barbier *et al.* 1990). For theoretical work on poaching and elephant conservation, see Khanna and Harford (1996) and Bulte and van Kooten (1997).

$$Max_y \int_0^\infty \left(H(y) + R(x) - D(x)\right) e^{-rt} dt, \tag{1}$$

where $x(t)$ is the stock of elephants at time t, $y(t)$ is the number of elephants harvested at t, $H(y)$ are net revenues from harvesting elephants (i.e. selling ivory), $R(x)$ are net benefits from tourism, and $D(x)$ is damage inflicted on agricultural lands and other wildlife habitat. Obviously, $R'(x)$, $D'(x) \geq 0$. The dynamics describing the stock of elephants can be represented as:

$$\dot{x} = G(x) - y, \tag{2}$$

where G(x) is the growth function for elephant herds; for bioeconomic models of elephant management, see also Swanson (1994) and Barbier *et al.* (1990).

From the maximum principle and assuming an interior solution, first-order conditions are:

$$H'(y) = \lambda, \tag{3}$$

$$\dot{\lambda} = \left(r - G'(x)\right)\lambda - R'(x) + D'(x). \tag{4}$$

From (3) and (4):

$$\frac{\dot{\lambda}}{\lambda} = r - G'(x) + \frac{D'(x) - R'(x)}{H'(y)} \tag{5}$$

Equation (3) states that the marginal revenue generated by harvesting another elephant now and selling its ivory is equal to the shadow value of retaining the elephant and perhaps harvesting it at a later date. Equation (4) is a generalized version of Hotelling's rule (Hotelling 1931), which, rewritten as in (5), states that the elephant population should be maintained at the level where the growth rate of the shadow value of an elephant (or, it's ivory) equals the dynamic opportunity cost of culling elephants (i.e., reducing the stock). The latter equals the discount rate minus the growth rate of the stock, plus a term that measures the change in future agricultural damages avoided plus lost future tourism benefits from culling another animal today.

Differentiating (3) and equating it to (4) gives:

$$\dot{y} = \frac{\left(r - G'(x)\right)H'(y) - R'(x) + D'(x)}{H''(y)} \tag{6}$$

Once the optimum equilibrium stock x^*, or steady state, is reached, harvest levels will be constant; hence $\dot{y}=0$. The implicit optimal stock size is then determined by solving

$$r = G'(x) + \frac{R'(x) - D'(x)}{H'(y)} \tag{7}$$

If the discount rate equals the growth rate plus a nonmarket value term, the optimal stock size has been reached. Then, from (2), harvest (y^*) should equal the regenerated fraction of the stock [$G(x^*)$].

The effect of a trade ban can be determined by removing the term $H(y)$ from (1) and solving the dynamic optimization problem anew. In that case, enforcement of the trade ban reduces expression (7) to

$$R'(x) = D'(x). \tag{8}$$

The optimal population level now occurs where marginal benefits generated from tourism equal marginal agricultural damage.

Is the level of the elephant stock determined from (8) larger or smaller than that determined from (7)? Alternatively, does the trade ban increase or decrease optimal population size? An answer can only be found by specifying the underlying functions. With respect to $D(x)$, it seems reasonable to assume that agricultural damage and damage inflicted on nature reserves are directly proportional to the size of the stock.[4] Hence, assume $D(x) = \alpha x$, where $\alpha > 0$ is a parameter (see below). On the other hand, the marginal utility of watching elephants is a positive but diminishing function of stock size, as anyone

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[4]This is a simplification since the presence of small groups of elephants may even have beneficial effects, since their feeding habits result in more sunlight on the ground, which favours growth of grasses, the preferred food of grazers. Thus, over some range, a convex curve would describe $D(x)$, while $D(x)$ might be concave over some other range (Western 1989). The current specification is chosen for expositional convenience, and it does not affect the qualitative nature of the results.

who ever spent a week in a Zimbabwean game park would agree. For simplicity, assume that the convex relation between $R(x)$ and stock size is adequately represented by $R(x) = \beta \ln(x)$, with $\beta > 0$. Also assume that the growth of an elephant population can be described by a logistic growth function, $G(x) = gx(1\text{-}x/K)$ where g is the intrinsic growth rate and K is the region's carrying capacity (for a more complex, and probably more appropriate functional form, see Basson *et al.* 1992).

5.3 Estimates of Optimal Elephant Stocks: Kenya

We consider two cases—an individual country that is a price taker (Kenya) and the African continent as a whole. For Kenya, the demand for elephants is given by $H'(y) = P$, where P is the (fixed) price of an elephant derived from the price of ivory. Using the functional forms described above and solving equation (7) for the steady state optimal stock of elephants with trade in ivory gives:

$$x_T^* = \frac{(g-r)KP - \alpha K + \sqrt{8\beta gKP + \left(\alpha K-(g-r)KP\right)^2}}{4gP} \qquad (9)$$

Equation (9) is solved for two recent prices and different values of the discount rate, with the results provided in Figure 5.1. The parameter values behind Figure 5.1 are given in the Appendix. Under the trade ban, the optimum population is simply β/α, or 15,700 elephants (with associated annual cull of 880 animals) irrespective of the discount rate.[5]

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[5]The actual stock of Kenyan elephants was approximately 16,000 elephants in 1989 (Pearce and Warford 1993).

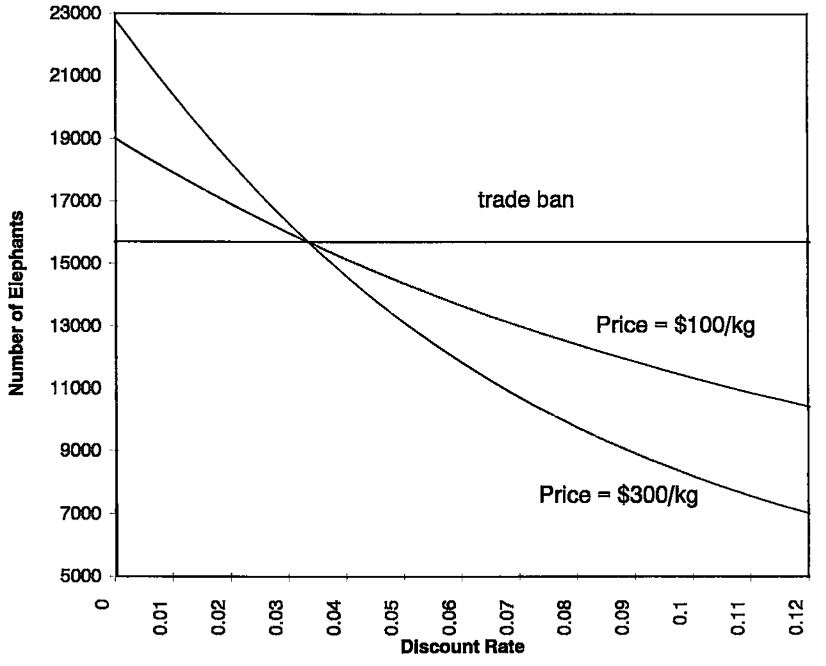

Figure 5.1: Estimated Optimal Elephant Stocks for Kenya for Given Prices and Discount Rates

From Figure 5.1, it is clear that, for both prices and when the discount rate is more than about 3.5%, the optimal stock of elephants under a trade ban exceeds that with trade; at lower discount rates optimal elephant stocks are greatest when trade in ivory is permitted.[6] A real discount rate of 3.5%

[6]In the absence of recreation benefits, damage and stock size dependent harvesting costs, the optimum stock size approaches 0.5*K* (=30,000 animals, see Appendix) as *r* approaches zero. In our model, at the optimum stock size, marginal agricultural damage exceeds marginal recreation benefits, which explains why for both prices x* is (far) below 30,000 animals. Relatively high prices for ivory induce the government to "tolerate" more damage, so we obtain the unusual result that the curves cross at a positive rate of discount.

might be a good approximation of the social discount rate for developed countries. For developing countries, however, r probably exceeds 3.5% by a substantial amount. This would imply that elephant populations should have increased after implementation of the trade ban. This is consistent with actual observations in recent years (Chadwick 1992). If the discount rate applied by the Kenyan government equals 10% and P is US$ 3,060 per elephant (or $300 per kilogram of raw ivory),[7] the optimum population size should almost double after implementation of the trade ban. However, if a social discount rate of 2% would be a more accurate description of the rate of time preference of the Kenyan government, the optimum population size under the trade ban is almost 3,000 elephants smaller than the equilibrium stock with trade. The effect of the trade ban, or its removal, on population size is sensitive to the discount rate.

Ivory prices also play a role. When ivory prices are $300/kg (as in 1989), optimal elephant stocks are higher for low discount rates but lower for higher discount rates compared to an ivory price of $100/kg (as in 1985-86) (see Harland 1988; Barbier and Swanson 1990). Increasing α reduces optimal stock levels, while increasing β enhances them. The results of a sensitivity analysis with respect to the recreation and damage parameters are provided in Table 5.1.[8] The negative values for the critical discount rate in Table 5.1 indicate that, at low values of α and high values of β, there is a chance that the trade ban is always optimal. On the other hand, high values of α (accompanied by low values of β) lead to critical discount rates that are higher than 5%, meaning that a government with a lower rate of time preference than this critical value will favour elephant conservation to a greater extent when trade is permitted. The analysis illustrates the importance to the international community of obtaining good estimates of α and β before making decisions about the benefits of a trade ban in ivory products.

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[7]The conversion from raw ivory prices to elephant prices is based on average tusk weights (see Appendix). For an empirical analysis of ivory prices, see Milner-Gulland (1993).

[8]Substituting appropriate functional forms in (7) gives: $r^* = g(1\text{-}2x^*/K) + 1/P\ (\beta/x^*\text{-}\alpha)$.

Table 5.1: Critical Switching Points of the Discount Rate for Various Values of α and β.

	$1/2\alpha$	α	2α
$1/2\beta$	3.3	5.2	6.1
β	negative	3.3	5.2
2β	negative	negative	3.3

5.4 Estimates of Optimal Elephant Stocks: The African Continent

Next, consider the case of the African continent as a whole. Assume that the inverse demand function for ivory is linear and downward sloping, i.e., $H'(y)=\psi-\gamma y$. In this case, equation (9) cannot be solved in straightforward manner because it is necessary to solve for the optimal harvest level y^* and optimal stock x^* simultaneously. The approach in the case of ivory trade is to find the steady state solution by setting $\dot{x}$ and $\dot{y}$ in equations (2) and (6) to zero and solving both for y:

$$y = \frac{\psi}{\gamma} - \frac{\alpha x - \beta}{gx\left[1 - \frac{2x}{K}\right]\gamma - rx\gamma}, \tag{10}$$

$$y = gx\left[1 - \frac{x}{K}\right]. \tag{11}$$

It is also not straightforward to solve (10) and (11) for y^* and x^* so this must be done graphically and numerically. The graphical results are presented in Figures 5.2 and 5.3, with the parameter values discussed in appendix 1. Again, the optimal stock under a trade ban is given by β/α, although the values of these parameters are different in the case of Africa than Kenya alone (see appendix 1). In the case of Africa, the optimal stock of elephants under a trade ban is 411,200 compared to a stock of approximately 600,000 at the time the ban was implemented (Pearce and Warford 1993).

The $\dot{y}=0$ locus is sensitive to the discount rate, intersecting the growth function ($\dot{x}=0$) from above when discount rates are 5% or less and from below when discount rates are greater than 5%; there is only one intersection point in all of the cases examined here. In Figure 5.2, the $\dot{y}=0$ locus for a discount rate of 4% is illustrated. As indicated in the figure, a stable, steady state solution to the problem exists, albeit a saddle point solution. Upon comparing Figures 5.1 and 5.3, one finds that the critical discount rate in the small country case is about 3.5% compared to about 5% when harvest of ivory affects price. The critical discount rate for the large country case is higher because the large country has the ability to set prices and realise a higher rate of return.[9] The conclusion is that a trade ban may lead to greater elephant conservation in the small-country case of Kenya (where the critical discount rate is low) than in the large-country case with a higher critical discount rate.

It is important to note, however, the potentially restrictive assumption in the "large-country" model that the relation between (potential) recreation benefits and elephant population size in Africa is proportional to that for Kenya—recreation benefits for Africa equal benefits for Kenya multiplied by the ratio of the total African elephant stock to the number of elephants in Kenya (see Appendix). This gave a value of recreation benefits for all of Africa equal to about \$900 million. Clearly, benefits from elephant watching are lower in many African countries where potential for this activity is less well developed (*viz.*, the disturbing influences of wars and the lack of a well-maintained physical infrastructure in some countries). For that reason, the estimate of \$900 million (and the value of β derived from it) serves as an upper bound for our calculations. A sensitivity analysis over benefit values can be used to discover critical discount rates. First, solving equations (10) and (11) simultaneously for r gives:

[9]For example, the price of ivory for the large country case amounts to US\$ 533 per kg, with an associated harvest of approximately 22,000 elephants.

$$r^* = g\left[1 - \frac{2x^*}{K}\right] + \frac{\alpha x^* - \beta}{\left[gx^*\left[1 - \frac{x^*}{K}\right]\gamma - \psi\right]x^*}. \qquad (12)$$

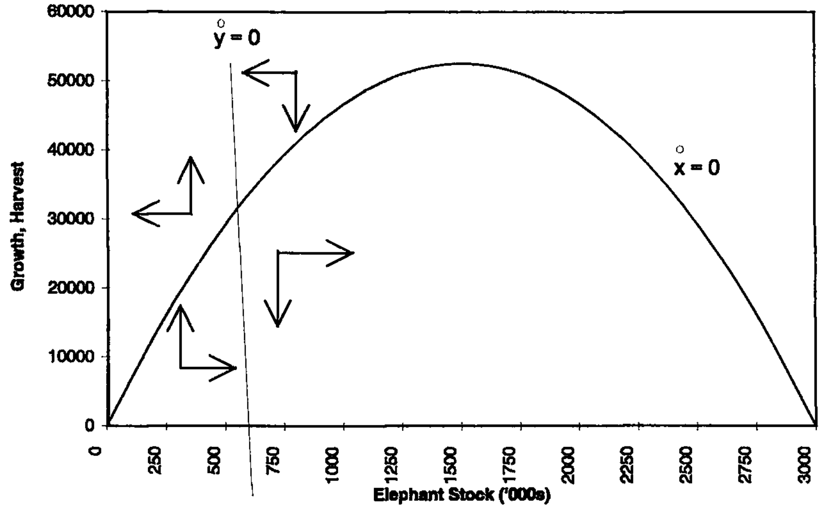

Figure 5.2: Stock and Harvest Dynamics for African Elephants (r=4%)

For different recreational values, it is possible to derive β from the relation $R(x) = \beta\ln(x)$, where x=600,000 and the remaining parameters are provided in appendix 1. Critical r ranges from 5.1% for \$900 million in assumed recreational benefits, to 5.9% for \$500 million in assumed benefits, and to 6.8% for \$100 million in assumed benefits. The result is rather stable with respect to parameter values and the earlier conclusion does not change.

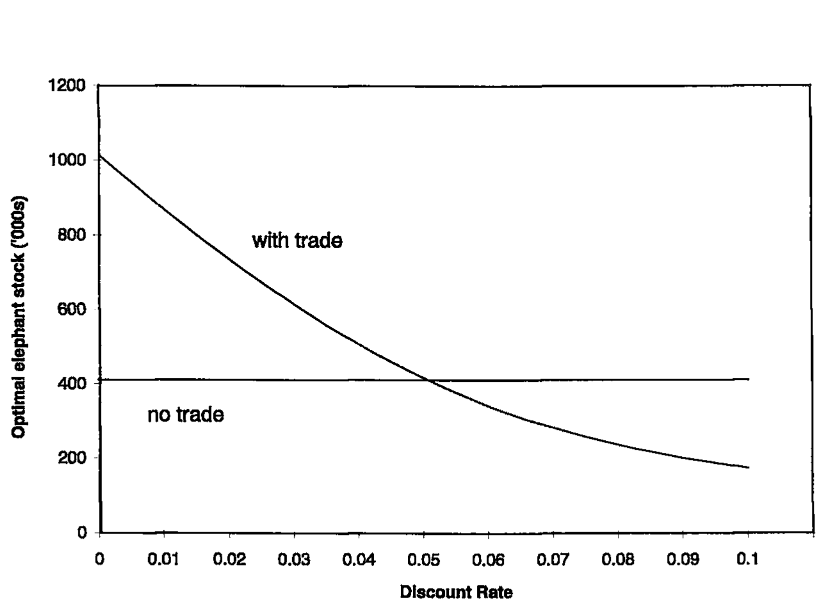

Figure 5.3: Comparison of Optimal Stock Levels for all Africa, With and Without the Trade Ban

5.5 Discussion

A natural next step in the analysis would be to test whether our prediction—countries with high discount rates have increased their stock, while countries with low discount rates have reduced their elephant populations in response to the 1990 trade ban—can actually be observed.[10] Empirical testing of this hypothesis is difficult because the social discount rate, or the rate of interest as applied by the government in inter-temporal cost-benefit analyses, is not revealed on markets. Even market-determined real rates of interest are difficult to determine for many African countries due to poorly functioning financial markets. Governments of many developing

[10]For a discussion of social and private rates of discount, see, e.g., Barbier *et al.* (1990). In this model, the correct discount rate is the social discount rate.

countries have typically tried to "fix" nominal interest rates in the past, mainly for political purposes. Real interest rates (nominal rates minus inflation) have been negative in the seventies and eighties for many countries (International Monetary Fund 1995). These artificial interest rates provide insufficient information about the rate of time preference of governments, so applying such a rate in an empirical analysis would serve little purpose. A more fruitful approach might be to work with indicators of the rate of time preference. For example, a common assumption is that the rate of time preference is related to (material) wealth and uncertainty of the future (Barbier *et al.* 1990), but this is left to future research.

One interesting observation, however, is that the rate of time preference of African governments will not be constant, but will change (more specifically, decline) as the economy and per capita income grows. This in turn implies that, to conserve elephants, it is necessary to frequently re-assess instruments implemented in the past. The same instrument (e.g., a trade ban) that worked well yesterday could be detrimental to elephant conservation at a future date. Whether a trade ban is effective in achieving its goal of species preservation or enhancement depends crucially on the discount rate, which is an object of a country's macro-economic policies, as much as it is on intervention by the international community to protect wildlife species.

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Appendix

Here the parameters used in the model are described. For both the small-country case and all of Africa, α is determined as the value of lost forage, and not the damage that might be caused by elephants trampling crops, since such damages can be avoided by appropriate management practices. As herbivores daily consume about 2.5% of their body weight in forage, one elephant annually consumes as much forage as required to bring 4.7 cows to full maturity (about 36,500 kg of dry matter). Using data for Kenya (see van Kooten, Bulte and Kinyua 1996) and current exchange rates, each cow has a net value of US$35. Hence, the forage displaced is valued at $164.50, which is taken as the value of parameter α.

The intrinsic growth rate of elephants is estimated to be 7%, or $g=0.07$, although that is an optimistic estimate that can only be obtained with appropriate management (Lindsay 1986; Forse 1987). We take the carrying capacity of Africa to be approximately double the highest population estimate of recent times (i.e., $K=3$ million). Kenya currently has some 16,000 elephants, which is about one-half of the numbers of several decades earlier. We assume double the latter amount as an approximation of carrying capacity for Kenya ($K=60,000$).

In the small-country case of Kenya, which is a price taker in the ivory trade (prices given in the text), β is derived as follows. Brown and Henry (1989) estimate that the annual value from viewing elephants in Kenya is $20-25 million. Given that there are an estimated 16,000 elephants in Kenya, then β = $25 million/log(16,000) = 2,582,553 (where log refers to the natural logarithm). For all of Africa, there are approximately 600,000 elephants, but there is no information about their tourism or recreational value. Taking the average estimate of Brown and Henry (1989) and assuming it applies to all of Africa (an unrealistic assumption) gives an upper estimate of viewing/recreational value of some $900 million. Then $\beta=67.6$ million.

Unlike for a small-country, a decision maker who might wish to determine the optimal stock of elephants for the entire African continent would need to take into account the effect of harvest on price. A linear demand function can be constructed from the fact that, in 1986, some 118,600 elephants were harvested and the price of ivory was about $100/kg, while, in

1989, 75,000 animals were harvested and ivory price was \$300/kg (it should be recognized that the price data for 1986 and 1989 may be affected by political events). Average tusk sizes were reported to be 5.6 kg in 1986 and 5.1 kg in 1989. Using these values, raw ivory prices are converted into elephant values of \$1120 and \$3060 for ivory prices of \$100/kg and \$300/kg, respectively; then, the parameter values can be calculated as $\psi=6397.16$ and $\gamma=0.044$.

CHAPTER 6

DYNAMIC AND STATIC APPROACHES TO MIXED GOOD MANAGEMENT: THE CASE OF MINKE WHALES IN THE NORTHEAST ATLANTIC[1]

Abstract

Whales have private and public good characteristics. In this chapter a dynamic model is developed that captures the complexities of managing such a resource. Estimates of the optimal steady state with this model suggest that the current stock of minke whales in the Northeast Atlantic is below the optimum steady state stock. The optimum approach dynamics to the steady state are consistent with drastic measures as the current moratorium on commercial whaling. These findings are not consistent with previous research in this field. The discrepancies should be attributed to the inclusion of non-use values in the present chapter. Finally, we compare the outcomes of the dynamic model with the results of a simple static model. We find that the latter consistently errs on the side of excessive exploitation because of its failure to account for the impact of current harvesting on future benefits and costs.

6.1 Introduction and brief historical overview of whaling

Commercial whaling has taken a severe toll on whale populations, especially in the period when whaling took place under conditions of open access. During the period 1920-1970, for instance, the population of blue whales in the southern ocean was reduced to 5% of its initial size, humpbacks to 3%, fin whales to 21% and Sei whales to 54% (World Resources, 1990-91). To regulate harvesting the International Whaling Commission (IWC) was

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[1]This chapter is based on: Bulte E.H., H. Folmer and W.J.M. Heijman (1996), Dynamic and static approaches to mixed good management: The case of minke whales in the Northeast Atlantic, Department of Development Economics, Wageningen Agricultural University, Discussion paper.

established in 1946. It introduced catch quotum systems in the seventies. Such a system was introduced for the North Atlantic in 1976.

At the 34th annual meeting of the IWC in 1982, it was decided to impose a moratorium on the commercial harvest of all whale species. This meant a total ban on harvesting any whale species under IWC purview for commercial purposes, starting from the 1986 coastal whaling season and the 1985-1986 pelagic whaling season. This ban was intended to continue until the end of the 1990 whaling season. The IWC decided in 1991 and 1992 to extend the moratorium, even for (the few) species that were considered to be able to sustain some commercial whaling. As expected, these decisions were not taken unanimously. The countries involved in commercial whaling used their right to object and voted against the proposals (with Spain as the only exception). Imposition and extension of the moratorium did not imply that whale harvesting ceased completely. Small numbers of minke whales were caught for ostensibly scientific purposes.

Due to the voluntarily basis of complying with the IWC, its stability was fragile. In 1992 Norway decided to "break" the moratorium and restarted commercial whaling on a modest scale the following year. Norwegian whalers harvested 301 minkes in the northern hemisphere in the following season. The 1995 quotum set by the Norwegians amounted to 232 minke whales (Anonymous, 1995). It should be observed that, since Norway filed formal objections to the moratorium, it was not bound to it. The Norwegians were thus in their right to continue whaling.

Minke whales can be found in all oceans, but their recurrence is seasonal as they undertake extensive migration between the tropics and polar seas. The large-scale commercial whaling industry was not very interested in the minke whale until the 1970's low stock sizes and quotas reduced catches of larger species. Minke whales are small baleen whales, growing to a size as large as approximately 10 meters. They compare poorly with many other whale species. For instance, approximately 32 minke whales are needed to yield as many barrels of oil as one blue whale. Moreover, they are fast, and once shot, they sink. Minkes are harvested for meat and fat. Estimations of the global stock size range from 500,000 to (way over) 1,000,000. They are especially 'abundant' in Antarctic waters. According to the whaling industry and IWC, minke whale populations are able to sustain some regulated commercial harvesting. This claim has aroused protest and debate.

The animosity can partially be explained by the "mixed good property" of whales: they involve elements of both private and public goods (Kuronuma and Tisdell, 1993). In the opinion of the whaling industry, the resource should be looked upon as a private good, i.e., harvested and marketed under conditions of exclusion and competition.[2] On the other hand, society is increasingly regarding whale resources as a public good, and has especially related existence values to the stock. A good that possesses this combination of private use values and public non-use values is defined a mixed good. Because of their public good characteristics, harvesting mixed goods like whales in accordance with conventional fishery models will result in market failure (though conventional models can be easily extended to include non-use values, as will be demonstrated below).

One of the objectives of this chapter is to test whether the moratorium on commercial harvesting of minkes is economically justified. Recent analyses of the moratorium exclusively focused on (private) use values, probably because of data availability. Conrad and Bjorndal (1993) use a delayed recruitment model to examine the efficiency of the moratorium for part of the North Atlantic stock of minke whales, and find that the moratorium is inefficient in most circumstances. Amundsen *et al.* (1995) show that this stock was in no danger of extinction, even under conditions of open access. However, if maximizing utility for society as a whole is the purpose of whale management, these findings are not sufficient to abolish the ban on harvesting, as non-use values are disregarded.

The objectives of this chapter are twofold. The first objective is to develop a dynamic framework to analyze management decisions of a natural

[2]For a long time property rights of marine mammals were not defined. Therefore the private good aspects of whale management gave rise to open access problems. It is recognised that this issue has been partially solved by the United Nations 1977 Informal Composite Negotiating Text and the United Nations 1982 Law of the Sea Convention. In 1977 many nations (including Norway) declared souvereign rights within a 200 mile zone. From then onwards, in principle, a national authority was able to manage the fish and whale stocks that inhabit the souvereign area. It thus can be argued that an open access regime has been replaced by a common property regime (see, e.g. Feeney *et al.* 1996). Since, however, whales tend to migrate and don't respect territorial borders, an international governing body like the IWC has an important role to play.

resource when the resource is a mixed good. The results of this model allow a comparison of the current stock and the economically optimal stock. The second objective of the chapter is to evaluate the economic rationale for the moratorium on the harvesting of minke whales by analyzing the approach paths towards this optimum. Because the unilateral "breaking" of the moratorium by Norwegian whalers has triggered off some tension (the U.S.A. threatened to ban Norwegian fish exports and Greenpeace called on tourists to boycott Norway), the issue is certainly not only of academic interest.

A by-product of the chapter is an illustration of the importance of employing an intertemporal rather than a static model. The latter was proposed by Tisdell (1991) and Kuronuma and Tisdell (1993). With respect to the problem of evaluating the moratorium, the simplified static approach boils down to balancing marginal preservation and consumption benefits. We demonstrate that, for this specific case, the static simplification will result in incorrect policy recommendations.

The organization of this chapter is as follows. In section 6.2, an optimal control model is developed that captures the mixed good effect. Solving the dynamic model suggests that the current population of minke whales in the Northeast Atlantic is below the social optimum (though some ambiguity remains because of uncertainty with respect to marginal non-use values). However, this result does not necessarily imply that a moratorium is optimal. To examine the validity of the moratorium, the optimal approach paths are numerically solved in section 6.3. In section 6.4 we apply the static model proposed by Kuronuma and Tisdell (1993) to the minke whale case and compare the results with the findings of the dynamic model. Conclusions and policy recommendations follow in section 6.5.

6.2 The mixed good model

An optimal control model is developed that includes both the public- and the private good features of whale stocks. In this context, it will be assumed that the IWC has the authority to set and monitor the quotas. The objective will be to maximize social value. Under the assumption that the private and public good benefits are separable, and that the cost of harvesting whales C is a function of the number of whales caught (y),[3] the objective function is as follows (suppressing time notation):

$$MAX_y\ W = \sum_{t=0}^{+\infty} \beta^t [H(y) + U(x) - C(y) - P(x)] \tag{1}$$

where $H(y)$ is the benefit from consuming y whales [i.e., the area under the demand curve for whales, so that $H(y)$ - $C(y)$ is the sum of producers and consumers surplus], $U(x)$ is the benefit from conserving x whales as an amenity resource, and β is the social discount factor (equal to $(1+r)^{-1}$, where r is the social discount rate). The usual assumptions prevail: $H'(y) > 0$, $H''(y) < 0$, $U'(x) > 0$, $C'(y) > 0$, $C''(y) > 0$. The term $P(x)$ relates to the economic cost of biological predation by minke whales. Flaaten (1988) claims that reduced harvesting of sea mammals such as minke whales increases predation by these mammals, which increases the cost of harvesting their prey species (capelin, herring and cod). In a recent article, Flaaten and Stollery (1996) estimated that the annual "predation cost" per minke whale amounts to approximately US$ 2,000 (the cost depends, among other things, on the management regime applied to exploit the prey species).[4] The equation of motion of the state variable and the non-negativity constraints are as follows:

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[3]In the Cobb-Douglas production function for whaling, as estimated by Amundsen *et al.* (1995), the stock output elasticity was not significantly different from zero. Possibly, this is explained by the biology of the resource (see Horwood 1990): minkes school in small numbers and therefore have characteristics of a "schooling species" (see Neher 1990). However, in the Spence production function (which was eventually preferred in the Amundsen *et al.* article because the predictions were more in line with actual observations), the stock output elasticity is by definition equal to one. This implies that it *may* be more appropriate to add the stock of whales x as an argument in the cost function: $C(y,x)$. For computational convenience we have omitted this variable, but we will discuss its potential implications in section 6.5.

[4]The predation cost of US$ 2,000 is only relevant given the current stock of the prey fisheries and the current management regime applied in these fisheries. A more correct approximation of this damage function would be to model it as the negative of the present value of economic benefits to the prey fisheries foregone. Hence, for a correct measure we should also maximize the benefits of the prey fisheries. This is beyond the purpose of this paper. However, the current specification implicitly assumes that management of the prey fishery is aimed at keeping the stock of prey species constant at the current stock size.

$$x_{t+1} = (1-m)(x_t - y_t) + G(x_{t-\tau}) \qquad y \geq 0,\ x \geq 0 \tag{2}$$

where m is the natural mortality rate; x_t - y_t is escapement, or the stock of whales left after harvesting; $G(x_{t-\tau})$ is the recruitment function; and τ is the delay in recruitment, reflecting that minke whales do not become sexually mature immediately after birth. Conrad and Bjorndal (1993) and Amundsen *et al.* (1995) specify a generalized logistic function to characterize recruitment:

$$G(x_{t-\tau}) = \gamma x_{t-\tau}\left[1 - (x_{t-\tau}/L)^{\alpha}\right] \tag{3}$$

where γ is the intrinsic growth rate; L is the environmental carrying capacity; and $\alpha > 1$ is a parameter that skews the logistic function to the right. The current value Lagrangian is given by:

$$L_c = \sum_{t=0}^{\infty} \beta^t [H(y) + U(x) - C(y) - P(x)] + \lambda_{t+1}\beta^{t+1}[(1-m)(x_t - y_t) + G(x_{t-\tau}) - x_{t+1}] \tag{4}$$

The necessary first-order conditions for an optimum solution are as follows:

$$\frac{\partial L_c}{\partial y} = \beta^t[H'(y) - C'(y)] - \lambda_{t+1}\beta^{t+1}(1-m) = 0 \tag{5}$$

$$\frac{\partial L_c}{\partial x} = \beta^t[U'(x) - P'(x)] + \lambda_{t+1}\beta^{t+1}(1-m) + \lambda_{t+\tau+1}\beta^{t+\tau+1}G'(x) - \lambda_t\beta^t = 0 \tag{6}$$

and the equation of motion ($\partial L_c/\partial\lambda = 0$). Equation (5) can be rewritten as:

$$\lambda_{t+1}\beta(1-m) = H'(y) - C'(y) \tag{7}$$

implying that the present value of next period's shadow price of the escaped fraction of the resource should equal price minus marginal harvesting cost in the current period. Equation (6) can be rewritten as:

$$\lambda_{t+1} - \lambda_t = P'(x) - U'(x) + \lambda_{t+1}\frac{r+m}{r+1} - \lambda_{t+\tau+1}\beta^{\tau+1}G'(x) \qquad \textbf{(8)}$$

which is an extended version of the standard non-arbitrage condition for renewable resources. The optimal rent path over time depends on (1) the rate of discount; (2) recruitment and mortality of the resource; and (3) the stock-effect on conservation utility and economic predation cost. Harvesting whales reduces the stock and thereby reduces (total) preservation values and predation costs, both now and in the future. A rational decision maker should include these effects when he decides about current harvesting. If the preservation values outweigh predation costs, the efficient rent path is more gently-sloped. This indicates that some of the harvesting should be postponed along an optimal path.

It is possible to use the first order conditions to solve for the optimal steady-state stock of whales that the IWC should try to support if it seeks to maximize social economic value. If the current stock of whales exceeds this optimal stock (x^*), it should be depleted in accordance with the efficiency conditions (7) - (8). We find the optimal equilibrium solution by substituting (7) into (8), and setting the RHS of equation (8) and (2) equal to zero [hence, $(G(x^*)-mx)(1-m^*)^{-1} = y^*$]:

$$r + m = \beta^{\tau}G'(x^*) + (1-m)\frac{U'(x^*) - P'(x^*)}{D(y^*) - C'(y^*)} \qquad \textbf{(9)}$$

We have thus obtained a single implicit equation for the optimal steady state $x_t = x^*$.[5] Equation (9) is a modified *golden rule*, where the second term on the RHS is the marginal rate of substitution between leaving the (surviving fraction of the) stock in situ and harvesting it today. Two observations that

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[5]Equation (9) can be rewritten as $Z(y,x) = 0$, which is an implicit function in (y,x) space. By the implicit function rule ($dy/dx = -Z_x/Z_y$) it is readily demonstrated that this function is upwards sloping for $U'(x) > P'(x)$.

are in line with intuition apply: first, the higher the marginal benefits of conservation [$U'(x)$] or marginal harvesting costs [$C'(y)$], *ceteris paribus*, the higher the optimal equilibrium stock. Second, the higher the marginal benefits of consuming whales [$D(y)$] or the marginal economic cost of biological predation [$P'(x)$], *ceteris paribus*, the lower the optimal stock. This is true because $G''(x) < 0$.

Equation (9) can be solved for the optimum population of minke whales in the Northeast Atlantic x^* if $G(x)$, $U(x)$, $P(x)$, $H(y)$ and $C(y)$ are known. It is possible to find approximations of $C(y)$, $G(x)$, and $P(x)$ in the literature, and to estimate the inverse demand function $D(y)$. Conrad and Bjorndal (1993) and Amundsen *et al.* (1995) model growth of minke whale populations with delayed recruitment models in the tradition of Clark (1976). In this paper we apply their specification of recruitment, implying that $m = 0.1$; $\alpha = 2.39$; $\tau = 7$; $\gamma = 0.15$; and $L = 130{,}000$.[6] For convenience, we initially assume that marginal harvest cost are constant and equal to Norwegian Kroner (NOK) 1,000 [for cost data, see Amundsen *et al.*, (1995)]. We also assume that the marginal economic predation costs $P'(x)$ are constant (i.e., every whale eats the same quantity of commercially valuable fish species) and equal, as mentioned above, US$ 2,000, or approximately NOK 14,000 (Flaaten and Stollery, 1996).

Next, we turn to the inverse demand function $D(y)$. First the variables included in the function will be described briefly. The income variable included in the equation to be estimated is real income per capita (I). Prices and numbers of whales harvested (from 1950 until 1987) are provided by Amundsen *et al.* (1995). In addition to quantity and income, prices of substitutes should be included in the inverse demand equation. For this analysis beef is used as a substitute. Since Norway is a net importer of beef, the average yearly import price of beef (B) was used. Inspection of the data indicates a non-linear inverse demand function. This was confirmed by the Box-Cox test. We therefore resort to a Box-Cox transformation:

[6]This implies that the pristine, or natural population (where natural mortality is exactly offset by recruitment) equals approximately 82,000 whales (see Amundsen *et al.* 1995).

$$z(\pi) = \frac{z^{\pi} - 1}{\pi} \ , \quad for\ \pi \neq 0$$
$$= \log(z) \ , \quad for\ \pi = 0 \tag{10}$$

The Box-Cox model is general, and can take the form of a log-linear and linear specification as special cases. The transformation can be applied to just the LHS of the model (model 1), or the RHS of the model (model 2), or both sides of the model (model 3). Log-Likelihood estimates indicated that model 1 is preferred. According to this specific Box-Cox model, the parameter π is estimated to be 0.60.[7]

Having obtained an estimation of π, we transform the LHS-data by means of equation (10), and estimate a conventional linear model. Estimates of the full model showed that the coefficient of the variable beef was highly insignificant. The estimated model, with the variable beef deleted and with prices expressed in 1994 NOK per whale, reads as (t-values in parentheses):

$$\begin{aligned} z(0.6) &= \underset{(3.6)}{1020} + \underset{(2.2)}{0.0037I} - \underset{(-7.3)}{0.3Y} \\ DW &= 1.7 \ , \quad R^2 = 0.9 \ , \quad F = 37.8 \end{aligned} \tag{11}$$

where Y indicates the number of whales supplied per period, and $z(0.6) = (p^{0.6} - 1)/0.6$. The Durbin Watson statistic indicates no autocorrelation.

Given the inverse demand function, it is possible to solve equation (9) for x^*, provided we have an estimate of $U'(x)$ and the social discount rate r. It is obvious that information with respect to $U'(x)$ is not readily available. The marginal utility of conservation could be derived from e.g. a CVM study by means of questions like: "*Given that there are currently so and so many whales, what is the maximum you would be willing to pay to enlarge the population of minke whales by 1,000 whales?*". Such data, however, are to

[7]We will use the Box-Cox model only to estimate π because except for π, the other parameters are generally not invariant to the "scale" of the analysis (i.e. whether quantities of meat supplied are expressed in kilograms or tons). This means that one should not rely on t-statistics of the variables besides π in the context of Box-Cox models (Davidson and MacKinnon, 1993).

our knowledge not available. Because of this limitation, we will focus on the willingness to pay for stock conservation, and assume that marginal utility of preservation is constant. Admittedly, this assumption is quite strong [see e.g. Loomis *et al.* (1993), Loomis and Larson (1994) and Loomis and White (1996)], and will probably lead to an overestimation of the optimum population. Values for stock conservation will be derived from the literature on willingness to pay for strict conservation of related species, such as blue and gray whales. The WTP for stock conservation are per adult or per household. Assuming the latter, $U'(x)$ is found by multiplying the number of households in the analysis by the total willingness to pay per household, and next divide this product by the (maximum) number of whales, x. In Table 6.1 we vary the total willingness to pay per European household[8] for preservation of minkes in the Northeast Atlantic from NOK 0 to NOK 30 (or, approximately, from US$ 0 to US$ 4, relatively low numbers when compared to estimates found in the literature, as will be discussed below). Equation (9) is numerically solved for various combinations of r and $U'(x)$, and the results are presented in Table 6.1.

[8]The reason for restricting the analysis to European households (approximately 90 million households) requires some explanation. One could argue that whale stocks constitute a global public good, hence that the relevant human population is the world population and not just that of Europe. However, the whale population under consideration is primarily found near the European coast whereas other parts of the world like eastern North America have access to whale populations in their own region. More important, however, is the following reason. Enlarging the spatial scope of the relevant human population would also require enlarging the total stock of whales: If all the households in the world care for all the whales in the world, the analysis should focus on the global whale population instead of dealing with minke whales in the Northeast Atlantic.

Table 6.1: Optimum minke population (in thousands of whales) in the Northeast Atlantic according to the dynamic model. WTP expresses annual willingness to pay for stock conservation in Norwegian Kroners (NOK) per household.

x^*	WTP=0	5	10	15	20	25	30
r=0.00	0	0	6.6	70.0	81.4	82.0	82.0
0.02	0	0	0	66.1	79.8	82.0	82.0
0.04	0	0	0	61.5	78.7	82.0	82.0
0.06	0	0	0	57.2	76.2	82.0	82.0
0.08	0	0	0	52.8	74.3	82.0	82.0
0.10	0	0	0	44.5	73.3	80.5	82.0

As mentioned above, for the current specification of the delayed recruitment function, the natural population is 82,000 whales. From Table 6.1 it can be deduced that the equilibrium population is equal to this natural population for discount rates lower than 8% when the average WTP per household for conservation of the stock of minkes in the Northeast Atlantic is greater than or equal to NOK 25. Note that the optimum stock is very sensitive to relatively low non-use values. Without willingness to pay for preservation, the whale stock should be depleted if maximizing the present value of welfare is the objective of the policy maker, even for low discount rates. This result is due to the significant predation costs imposed by whales.[9]

The literature provides two estimates for the actual stock in this area. First, according to Amundsen *et al.* (1995) this stock consists of approximate-

[9]There appears to be a general societal consensus on the notion that exploitation of renewable resources should be on a sustainable basis. This implies that extinction should be avoided, and that at least a viable population of minke whales should be supported for an infinite period of time. More accurately formulated: the risk of extinction of the species within several centuries should be acceptably low. The model can be elaborated by a condition that relates the current stock x_t to the concept of the minimum viable population (MVP). This would imply incorporating the feature of critical depensation in the recruitment function, such that net growth becomes negative for population levels below MVP. We refer to Soulé (1987) for more information with respect to MVP and (marine) mammals and to Clark (1990) for more information on (critical) depensation in recruitment.

ly 57,000 whales, which would imply that the current stock is sub-optimally low for many (realistic) combinations of preservation values and discount rates as reported in Table 6.1. Second, according to the IWC (1992) [cited in Flaaten and Stollery (1996)] the actual population amounts to 86,000 whales.[10] In other words, the actual population would exceed the natural population of our model, which is clearly not consistent. In other words, this population estimate should not be compared to the population estimates as presented in the table. Instead, it suggests that the delayed recruitment function applied in this chapter and elsewhere in the literature is not correct, and should be modified to allow for a higher natural population. One crude way to achieve this is to increase the environmental carrying capacity L. When we repeat the analysis with such a higher value (for example L = 160,000, such that the natural population is approximately 100,000), then the optimal steady state x^* would be equal to this new natural population for WTP $\geq$ NOK 35 and $r \leq 10\%$. This implies that, again, the optimal stock equals the natural stock for many different combinations of preservation values and discount rates (indicating that whenever the current stock is lower than the natural stock, it is sub-optimally low), although this finding is less clearcut then when the lower population estimate is more apt.

We would like to note that the preservation values for which the optimum steady state exceeds the current stock need not be high. For example, taking the average of the population estimates mentioned above (or 72,500 whales), then for a social discount rate of 10%, the annual willingness to pay for whale conservation per household should be about NOK 20, or less than US$ 3. Of course, it is an open question whether the average European household is actually willing to pay such amounts, but the literature available suggests that this may well be the case. Pearce and Warford (1990), for example, present results of research indicating that the existence value of blue whales equals about US$ 8 per adult, (which would be US$ 16, or about

[10]An anonymous reviewer has pointed out to us that recent (1996) IWC estimates of the minke stock are significantly higher than the ones reported above.

NOK 100, in our analysis because we focus on households).[11] Loomis and Larson (1994) have estimated WTP to increase populations of gray whales, and present numbers that are even higher; according to their work, the WTP for households to increase the current population by 50% and 100% amounts to approximately \$ 17 and \$ 19, respectively. This indicates that actual WTP-values may well be in the vicinity of NOK 100, which would be consistent with the conclusion that the current stock of whales is sub-optimally small, even for high discount rates. This result is robust with respect to the magnitude of $P'(x)$ and $C'(y)$. For example, for $P'(x)$ equal to NOK 28,000 and NOK 7,000 (hence doubling and halving the original value) and $r \leq 10\%$, the optimum population is equal to the natural population when WTP $\geq$ NOK 40 and NOK 25, respectively.

In order to evaluate the rationale of the moratorium, however, the observation that the current stock may be sub-optimally low is not enough; we also need to consider the optimal approach path to the steady state. We turn to this topic in the next section.

6.3 Approach paths to the steady state population

The optimum approach paths to the optimum population are obtained by solving the trajectories that converge to (x^*, y^*). The model described in the previous section would in general not prescribe a moratorium or Most Rapid Approach Path (MRAP) because the Hamiltonian is not linear in its control variable y_t (this is the case because we have a downward sloping demand curve for whales). However, it is possible that the optimum approach path in the x-y phase plane is steep and prescribes zero harvesting even for stock sizes just below the optimum steady state.

It is straightforward to evaluate the rationale of the moratorium for the remaining steady states by analyzing whether the optimal approach path (i.e.,

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[11]The existence values for other natural assets according to Clark (1980) and Herrera (1985), cited in Pearce and Warford (1993), range from about US\$ 7 per adult (for dolphins, seals and otters) to US\$ 15 (for the Grizzly bear), corresponding with almost NOK 100 and more than NOK 200 per household, respectively.

the stable arm) to the equilibria is consistent with periods of extreme low or zero harvesting. For example, let us assume that the low population estimate of 57,000 whales is correct and analyze the case of the following steady state: (x^* = 78,700 , y^* = 420), corresponding with WTP = NOK 20 and r = 4% (if the high population estimate of 86,000 whales is more apt, obviously the model prescribes positive harvesting because then the current stock would exceed the optimal stock). Since the selected steady state is an equilibrium, we can approximate the dynamical system for the state variable [equation (2)] and control variable [implicit in equation (8)] using Taylors expansion:

$$F(x^*,y^*) = F_x(x^*,y^*)(x-x^*) + F_y(x^*,y^*)(y-y^*) + h.o.t., \quad \textbf{(12)}$$

where $F(x,y)$ is the RHS of (2) and (8), and where *h.o.t.* is short for higher order terms. After the appropriate derivations and substitutions [78,700 for x^* and 420 for $(G(x^*)-mx^*)(1-m)^{-1} = y^*$], we obtain the linearized system:

$$\begin{aligned} \chi_{t+1}-\chi_t &= -0.1\chi - 0.9\psi \\ \psi_{t+1}-\psi_t &= -0.012\chi + 0.4\psi \end{aligned} \quad \textbf{(13)}$$

where $\chi = x - x^*$ and $\psi = y - y^*$. The eigenvalues of this linear system are R_1 = 0.42 and R_2 = -0.12. Since one eigenvalue is negative and the other is positive, the steady state is a saddle point equilibrium for the system. The corresponding eigenvector of the eigenvalue with negative value is just V = (1, 0.022).[12]

Next, a point on the linearized separatrix near the steady state equilibrium is chosen, and the Runge-Kutta algorithm is employed to approximate the actual, non-linear separatrices (see Press *et al.*, 1986 or Conrad and Clark 1987). While the time lag between birth and sexual maturity does not have an

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[12]The general solution to the general system also includes the positive eigenvalue R_1 and associated eigenvector W

$$\begin{bmatrix} \chi \\ \psi \end{bmatrix} = aWe^{R_1 t} + bVe^{R_2 t}.$$

However, the only trajectories that converge to the equilibrium are those for which a = 0 (Conrad and Clark 1987).

impact on the steady state (where, by definition, $x_{t-T} = x_t$) it does have an impact on the approach dynamics. Allowing for this, the thus obtained separatrix is steeply upward sloping, and crosses the horizontal axis of the x-y-phase plane at a stock size near 74,000. This implies that a (temporary) moratorium is optimal until the population has grown to 74,000 whales. Beyond that point harvesting should start, and annual harvesting should rapidly increase from 0 to 420 whales as the population grows to the steady state stock of 78,700 whales. The phase plane with stable arm for the case just analyzed is depicted in Figure 6.1.

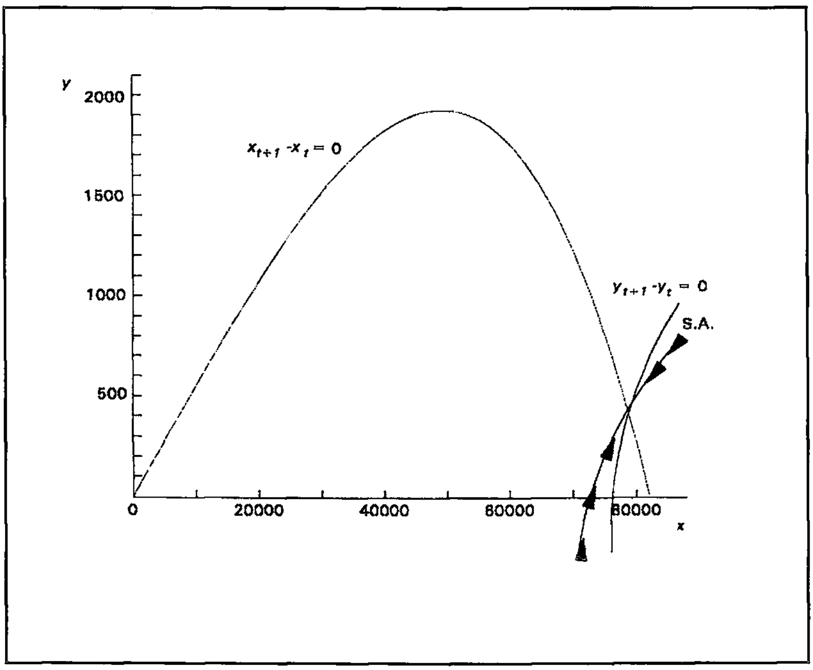

Fig.6.1: The steady state stock and harvest and approach dynamics (S.A. is the stable arm) for the case r = 4% and WTP = NOK 20.

We obtained similar results for other steady states reported in Table 6.1. This finding is consistent with a casual remark by Clark: "It can be shown that the optimal approach to equilibrium is not the bang-bang approach, but in most cases the bang-bang approach is nearly optimal" (Clark 1990, p.239).

An assumption that underlies the optimality of this solution is reversibility in capital investments. Amundsen *et al.* (1996) write that "in addition to whaling, the boats also participate in other fisheries. About 24-45% of total fishing time was devoted to whaling". This suggests that capital in the whaling sector is, at least to a certain extent, malleable. For models with irreversible capital investments, we refer to Clark *et al.* (1979) and Boyce (1995).

6.4 A static simplification

The numerical solution by means of the Runge-Kutta algorithm has the advantage that it is convenient and rather accurate. It has the disadvantage that there is not much intuition to the solution it provides. This is especially unfortunate in this particular field, because whale management is a politically sensitive issue, where politicians have to account for their decisions to critical and perhaps powerful interest groups. Under these circumstances, decision makers may be tempted to resort to models that are easily explained and readily understood by the public.

In this section we will demonstrate what would happen if we had approximated the dynamic model by an annual series of static models, as advocated by Tisdell (1991) and Kuronuma and Tisdell (1993). In the static model, the marginal benefits of (current) preservation and consumption are balanced (see also Longva, 1990). Disregarding harvesting costs and predation costs for the time being for expositional clarity, judging the rationale of the moratorium in the static framework boils down to comparing the marginal benefits of whale conservation (i.e., the marginal benefit to conservationists of preventing whaling harvest) and whale consumption (Kuronuma and Tisdell, 1993). In this chapter we do not assign property rights to the resource to a particular group. Rather, we assume that there is a benevolent planner that allocates the resource to the group where marginal utility of the unit is greatest. In Figure 6.2, the annual catch of minke whales is indicated on the horizontal axis, ranging from zero to x (the entire stock).

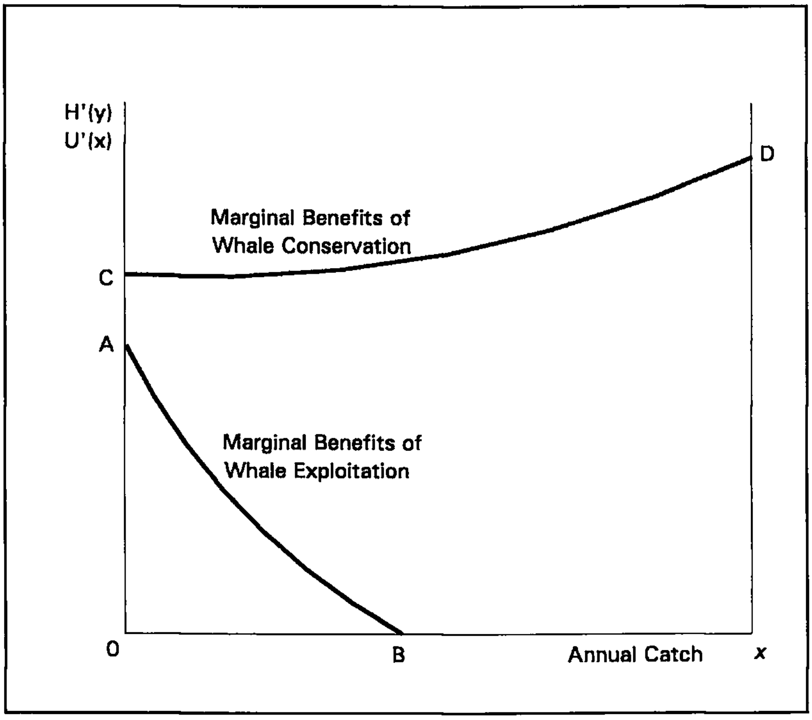

Fig.6.2: Marginal utility of conservation (curve CD) and marginal utility of exploitation (curve AB). According to the static model, a moratorium is justified because the first exceeds the latter for all harvest levels.

In Figure 6.2, the marginal benefit to conservationists of avoiding whale harvesting (curve CD) exceeds the (current) marginal value of whaling (curve AB) for *all* levels of whale harvesting, hence a moratorium is justified. Kuronuma and Tisdell (1993) demonstrate two more cases, that will not be depicted here. First, it is possible that the marginal benefit curves intersect. Then, a quotum would be optimal, as the socially optimal harvest level (i.e., the harvest level at which the curves intersect) is smaller than the harvest level collectively desired by the whaling industry for well behaved marginal benefit curves. Note that the optimal harvest level in the context of a static model for the whaling industry coincides with the intersection of horizontal axis and the curve displaying marginal benefit to whalers, or *B* in Figure 6.2.

Second, it may be the case that the curves do not cross, but rather both intersect the horizontal axis with a "gap" in between (or, along the horizontal axis: OC > OB). Under this condition harvesting of a species as a pure private good is optimal (Kuronuma and Tisdell 1993).

As described above, the economic criterion of strict conservation in a static context is that the marginal utility of harvesting the first whale is less than the net marginal utility of conserving that whale. With marginal utility of conservation constant, we can calculate the WTP that locates the net marginal value of preservation curve (a horizontal line) just above the demand curve for whales. This implies that the minimally required WTP for the moratorium per household is obtained by multiplying the sum of the value of this first whale harvested and the marginal predation cost $[P'(x)+D(1)]$ by the total number of whales (x_t), and next dividing this total by the number of (European) households: WTP = $x[P'(x)+D(1)]/9*10^7$.

If the required WTP thus found exceeds the WTP estimates found in the literature for minke whales or comparable assets, the moratorium is not justified (see section 6.2). The average economic value of the first whale for the whaling industry is estimated by substituting 1 for Y and the 1994 per capita income in equation (11). The price of the first whale harvested thus obtained amounts to approximately NOK 106,429. It should be observed that this price is not exceptionally high when compared to prices paid in the 1980s. For example, the price per whale in 1987 (expressed in 1994 NOK) was about NOK 76,000, corresponding with a harvest of 279 whales.

In order to determine whether the moratorium is efficient, the "reverse analysis approach" is applied. This comes down to evaluating whether the WTP to avoid whaling is such that the benefits of conservation exceed the benefits of exploitation. The minimum necessary WTP to conserve the complete stock per European household, calculated as described above, is approximately US$ 16.5.[13] When the lower estimate of population size,

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[13]For a correct measure of the minimally required WTP per household we should correct for the marginal harvesting costs. Deducting the average harvesting cost in the 1980s from the consumption value of the first whale and calculating the minimally required WTP for strict conservation amounts to a trivial correction: about US$ 15 instead of US$ 16.5 for the high population estimate (cost data taken from Bjorndal and Conrad 1993). Also, it is important to note that the number of whales caught per vessel was low in the mid 1980's: in 1986 and 1987

reported by Amundsen *et al.* (1995) is correct, the necessary WTP reduces to almost US$ 11. Comparing this to the estimates of WTP for conservation of minke whales as discussed in section 6.2, we conclude that the static model is not necessarily consistent with a moratorium: when the higher population estimates are more apt, the moratorium may be not efficient and hence should be replaced by a harvest quotum. For low population estimates a temporary moratorium may be optimal, but with growth of the resource stock the required WTP to support a moratorium will increase, and after a period of zero harvesting the static model prescribes a switch to a quotum system. Obviously, such a switch is advocated *too soon* (i.e., at too low stock levels) when compared to the results of the dynamic model. This further illustrates the importance of applying the correct (i.e. dynamic) model for policy recommendations.

6.5 Conclusions and discussion

The purpose of this chapter was to analyze efficient management of mixed goods. Static and dynamic aspects were considered. A conclusion is that the effect of existence values (or amenity values) on depletion is in accordance with intuition: efficient depletion is characterized by a more gently-sloped rent path, and some of the initial harvesting is postponed. In addition, the equilibrium stock of mixed goods will be larger than the equilibrium stock of a pure private good. In this chapter we have used various values of (constant) marginal preservation utility, and concluded that the current stock may is probably sub-optimally low, depending on the social discount rate and magnitude of the non-use values. This is especially true when the (lower) estimates of current stock size as reported by Amundsen *et al.* (1995) are correct [as opposed to the higher estimates presented by Flaaten and Stollery (1996)].

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for example, the average annual number of whales harvested per vessel was about 7. This suggests that the bias due to the existence of fixed costs that arises, because we determine the average cost of harvesting just one whale, will be relatively small.

The dynamic model in section 6.2 can be expanded upon by allowing for stock dependent harvest costs [$C(y,x)$, with $\partial C/\partial x < 0$]. This would result in optimal population estimates that are even higher than the ones presented in Table 6.1, such that the gap between the current and optimal stock size is underestimated. An interesting topic for future research would be to estimate declining marginal non-use values of whales, along the lines set out by Loomis and Larson (1994) for gray whales, such that the optimum whale population and approach dynamics of the dynamic model as provided in section 6.2 can be solved using more realistic function specifications. For some of the dramatic implications of allowing for declining marginal utility on optimum stock size of minke whales in the Northeast Atlantic, see Bulte and van Kooten (1997).

A second objective of this chapter was to investigate whether maintenance of the moratorium as installed by the IWC is efficient. For this purpose we compute the stable arm of the dynamic model. The numerical simulation indicates that the moratorium is an optimal management strategy whenever the current population is smaller than the optimum population. Again, the result is obtained under the assumption that the marginal utility of conservation of whales is constant. Consequently, the case for a moratorium may be overestimated.

The (prudent) conclusion that the moratorium on harvesting whales is economically efficient is not consistent with results of Conrad and Bjorndal (1993), who argue that a quota system may be more appropriate than a moratorium. However, their analyses focus exclusively on the consumption side of whales.[14] Our result is consistent with a recent paper by Horan (1996), in which he analyses whether the moratorium is justified when ethical

[14]The following remark with respect to this finding is relevant. Allowing whalers to harvest the relatively abundant minke will increase the risk of harvesting rare and vulnerable species. Although the various species are not hard to distinguish, and accidental killings of rare species will therefore be few, whalers will be tempted to shoot at any whale that comes by, regardless of the species. Once processed it is hard to tell the various species apart. Extinction of more rare whale species may be the result. Clearly, control and enforcement of the rule will therefore be important. In addition, enforcement and control will be needed to guarantee that the whaling industry does not over-harvest its minke quota. The extra costs associated with enforcement (that is, the costs over and above the costs of assuring the moratorium) should not exceed any social benefits of switching to the quotum system.

values are included. He assumes that people receive disutility from the actual harvesting of whales, irrespective of the stock size in situ. With this specification, Horan also concludes that a moratorium may be optimal.

Our static approximation of the dynamic model, as proposed by some authors, yields results that are misleading. Because the effect of current harvesting on future harvesting possibilities and preservation utility (and conversely, on future predation costs, but this effect is dominated by the first two effects) is overlooked, the static model incorrectly suggests that the current moratorium should be abolished. This illustrates that there is a danger in "simplifying" dynamic models: the resulting policy recommendations of the simple static model could well be badly wrong.

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CHAPTER 7

OPEN ACCESS, COMMON PROPERTY AND SCARCITY RENT IN FISHERIES[1]

Abstract

For a long time, property rights for marine resources were lacking. Many fish stocks were characterized as open access resources, and not managed efficiently. This resulted in falling stock levels and a declining income for fishermen. In the late 1970's, the policy response to this problem in fisheries was the implementation of 200-mile fishing zones, which enabled the European Union to formulate and implement the Common Fisheries Policy, aimed at (among other things) conservation and distribution of available stocks. For the case of Germany, empirically examined in this chapter, this shift from an open access regime towards a common property regime had favourable outcomes. The trend of falling prices was reversed. The conclusion was that intertemporal efficiency had increased as a result of (inter)governmental policy. Apparently, a wedge was formed between price and marginal harvesting costs, which we have interpreted as a return of scarcity rent as a component of prices.

7.1 Introduction

For a long time, marine fisheries in Europe and elsewhere in the world were managed as open access[2] resources. The outcome of open access

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[1]A different version of this chapter is previously published as: Bulte E.H., H. Folmer and W.J.M. Heijman, *Environmental and Resource Economics*, 6(1995):309-20. The main results are summarized in Bulte E.H., H. Folmer and W.J.M. Heijman, *Economisch Statistische Berichten*, p.330-333.

[2]In an open access regime each potential user has complete autonomy in utilizing the resource. Under a so-called common property regime a group of owners can control the resource and exclude others from using it (see section 2.4).

management, however, was far from sustainable and efficient. Although global fish catch in tonnes did not decline until 1990, the composition of species changed substantially. Valuable species, such as herring and flounder, were replaced by less valuable ones such as skates and dogfish. In addition, the catch declined in several major fishing areas and fishermen's incomes were depressed.

Economic theory predicts that under conditions of open access, a natural resource will be exploited at such a pace that the scarcity rent dissipates completely and that, eventually, total costs equal total revenues (Gordon 1954, Schaefer 1957, Bjorndal and Conrad 1987). Open access implies that, as long as there is an excess profit, new entrants are attracted, thereby reducing overall profits. The rationale for rent dissipation is that no fisherman is willing to "invest" in stock maintenance since the expected benefits are likely to be reaped by others, which implies that the future is discounted at an infinite rate. Also, other drawbacks of open access resource management have been documented. First, the fish stock is often severely depleted, and extinction may be the result. For instance, Spence (1973) found that the stock of blue whales was reduced to undesirable levels because of the open access character of the species.[3] Second, in addition to lost profit and possible stock extinction, open access fisheries usually result in an inefficient structure of the fishing industry, as is argued by for instance McKelvey (1983) and Lipton and Strand (1989).

It is now widely recognized that open access results in biologically and economically suboptimal outcomes. That is why nowadays fisheries are regulated. Iceland and Norway were the first European countries to declare a substantial exclusive fishing zone. In 1977 the European Union followed this initiative and declared sovereign rights within a 200-mile zone, thereby

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[3]Likewise, Bjorndal and Conrad (1987) argue that North Sea herring was saved from possible extinction by the closure of the fishery at the end of the 1977 season. Wilen (1976), however, studied the dynamics of open access fur seal hunting and found that rent dissipation may well be compatible with stock survival.

anticipating the 1982 Law of the Sea Convention.[4] By establishing exclusive zones, it was expected that the open access characteristics of marine resources would be reduced. In fact, the open access regime was replaced by a common property regime, under which a group of owners was expected to regulate exploitation. This exclusive zone has been very important to the European fishery. The nine Atlantic E.U. member states produce more than 5 million tonnes of fish every year, over 80% of which are harvested in the exclusive zone (Salz, 1991).

To be effective, a conservation policy must be pursued for the entire area inhabited by the fish stocks concerned. Since 1977 such an area has existed for the European Union (for some stocks, cooperation with third nations, as Norway, is required for effective policy formulation). It was, however, not until 1983 that the Common Fisheries Policy (CFP), which aimed at both the distribution of fish stocks among member states and the maintenance of the stocks, was implemented.[5] Total Allowable Catches (TACs) were determined, based on mostly biological and political considerations (see chapter 8), and then divided among member states. These states in turn allocate the national quota to their fishermen. There are differences, however, in how individual countries do this. In some countries, fishermen are allowed to harvest until the national quota is reached, while in other countries fishermen are allotted individual quotas, possibly transferable. Initially the TACs were fixed at levels which corresponded with existing catches. The levels of the catches have been reduced in a stepwise manner. Likewise, there has been a shift in the structural policy from stimulating

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[4]Already at the third United Nations conference on the Law of the Sea in 1977 a document called the Informal Composite Negotiating Text was produced with the recommendation that coastal ocean states may have exclusive economic zones where appropriate.

[5]The CFP consists of policies for structures, markets, external fisheries and conservation [for more information the reader is referred to Holden (1994)]. There are two reasons why formulation and implementation of the CFP did not immediately follow the establishment of the Community sea. There was a time-consuming debate between the UK and France about an exclusive zone for England and it took several years before matters concerning the determination of TACs and their allocation were settled in a politically acceptable way for all member states.

vessel construction to reducing fishing capacity in Union waters (Holden, 1994).

According to Schmidt (1993), measures taken to halt overfishing and make fisheries efficient have been largely unsuccessful. Recent research has provided evidence for this proposition. According to estimates of the F.A.O of the United Nations, the global revenue loss because of inefficient fisheries is in the order of US$ 15-30 billion per annum. The income foregone because of suboptimal fisheries management in the European Union alone amounts to about US$ 2.5 billion annually (Schmidt 1993). Holden (1994) argues that the CFP suffers from four major defects. First, the CFP has not been conceived as a coherent, integrated policy. Conflicting objectives arose, for instance, between the structural and conservation policies. Second, the policy has no specific objectives which hampers effective implementation. Third, the fundamental basis has been biological rather than economic, and fourth, the policy suffers from the decision-making processes of the Union where every decision is a political compromise. The importance of ineffective and inefficient regulation is demonstrated by Dupont (1990). Dupont (1990) analyzed three sources of rent dissipation along the west coast of Canada, namely capital stuffing, fleet redundancy and fleet composition, and found that potential resource rents had dissipated to a great extent.[6]

In this chapter, we discuss a small aspect of the CFP. The effectiveness of regulation in terms of return of rent in Germany is analyzed. The main objective was to test whether shifting from an open access management regime to a common property regime enhanced intertemporal efficiency. It should be observed that, in contrast to most of the analyses of rent dissipation, this paper on rent return was based on empirical data. No attempt was made to model processes of production and consumption of the resource. Instead, resource scarcity issues were dealt with exclusively by regarding the outcomes of the underlying processes. Observations of prices (which reflect the outcome of harvesting and demand) were combined with assumptions

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[6]Capital stuffing implies that restricted inputs in the harvest process are replaced by unrestricted inputs, which increases harvesting costs and reduces rent. Fleet redundancy occurs when more than the optimal number of vessels (i.e., the number that minimizes costs) is allowed to participate in the harvesting process. Finally, rents may dissipate because harvesting does not take place by the most efficient vessels.

regarding harvesting costs, in order to draw conclusions about scarcity rent over time.

The paper is organized as follows. In section 7.2, the notion 'scarcity rent' is explained briefly. In addition, the method applied to test the main hypothesis is described. Section 7.3 is a case study of German fisheries. The results illustrate that government policies produced some beneficial effects. In section 7.4 conclusions are drawn.

7.2 Scarcity rent in fisheries

Under the assumption of constant extraction costs per unit of fish and constant demand for fish, implementation of effective policy will result in a U-shaped price path. First, in the absence of governmental control and with abundant stocks, a profit is made. Then, as more and more new entrants appear, dissipation of rent occurs until price per unit equals harvesting costs per unit (phase 1 in Figure 7.1). For a certain period of time, prices are expected to equal marginal costs. During this period possible future benefits of harvesting are completely disregarded and "overfishing" is likely to occur (phase 2). If the government intervenes successfully, prices will go up and rent may return (phase 3).[7] Finally, the rent increase will tail off and settle at a level determined by marginal costs and government regulation (this phase is not included in Figure 7.1, but see section 2.2 for steady state analysis). Under these (strong) assumptions, testing of effective government policy is straightforward, and essentially comes down to analyzing whether such a U-shaped pattern actually exists.

The shaded area indicates scarcity rent. The assumption of constant marginal harvesting costs, however, may be too simple. First, it can be expected that technological developments constantly reduce the cost per unit of catch (bigger trawlers, new technologies for detecting and harvesting fish). Second, it can be expected that as the stock of fish declines, the costs of har-

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[7]It is possible that, after a certain period of time, effective policy implementation results in an increase in the *in situ* stock, such that harvest quotas can increase. Increasing supply will result in lower prices, counteracting the process described above.

vesting a unit of fish go up. These are countervailing effects. Therefore some additional assumptions are needed to empirically analyze policy effects.[8] They are discussed below.

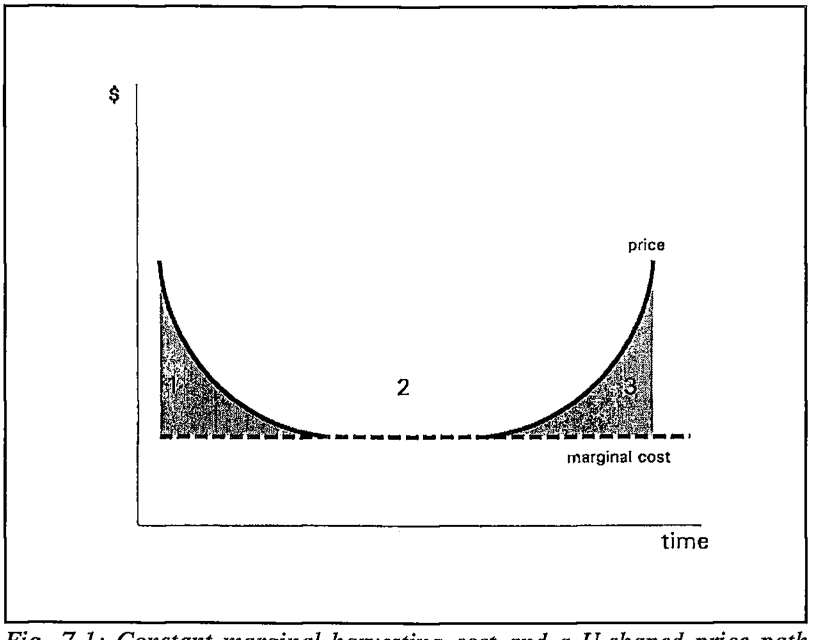

Fig. 7.1: Constant marginal harvesting cost and a U-shaped price path implies increased efficiency. The shaded area indicates scarcity rent.

In fisheries theory, it is common to distinguish between schooling fisheries and searching fisheries. Some fish populations have a propensity to school in large numbers during the course of migration or reproduction activity, or as a defence against predators. Well-known examples are salmon, mackerel, anchovies and herring. The widespread use of modern fish-finding devices makes locating schools easy and cheap, even when (because of overfishing) the stock is smaller. From this follows that in the case of schooling fisheries, the marginal harvesting costs decline over time, though at a decreasing rate.

[8]Note that marginal harvest costs will only be constant under stock depletion if technological change exactly offset the increasing difficulty of harvest as the stock shrinks.

In the case of searching fisheries the stock of fish (the population level) contributes significantly to the harvesting costs. Examples are plaice and sole. In this case there are two opposing tendencies. First, ongoing innovations constantly reduce the cost of effort[9] (although possibly at a declining rate), which lead to a constant downward pressure on marginal harvesting costs. Second, there is the increase in harvesting costs owing to increased scarcity. The resulting marginal harvesting cost curve is likely to be U-shaped. The marginal harvesting costs of both schooling and searching fisheries are depicted below in Figure 7.2.

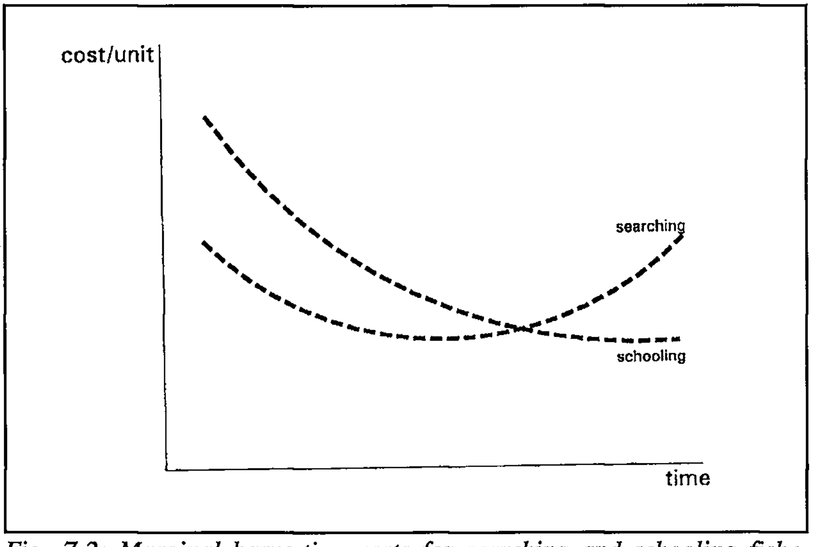

Fig. 7.2: Marginal harvesting costs for searching and schooling fisheries.

The assumptions made so far about marginal harvesting costs enable us to say something about the price path. For searching fisheries, it is U-shaped because of the U-shaped pattern of marginal costs. If scarcity rent returns because of successful policy intervention, the rate of price increase must be sharper than can be expected on the basis of the "naturally" increasing marginal costs, since the rent drives a wedge between price and (rising)

[9]Note that as a result of regulation, ships will spend more time idle in the harbour, which will enhance the opportunities for investment.

marginal harvesting costs. This is illustrated in Figure 7.3, where the moment of policy intervention is indicated as t^*. If the parameters of the harvesting cost function are assumed to be constant over time,[10] the policy effect can simply be measured by comparing the actual price increase in the common property period with the price increase on the basis of the "natural" development (as estimated for the open access period).

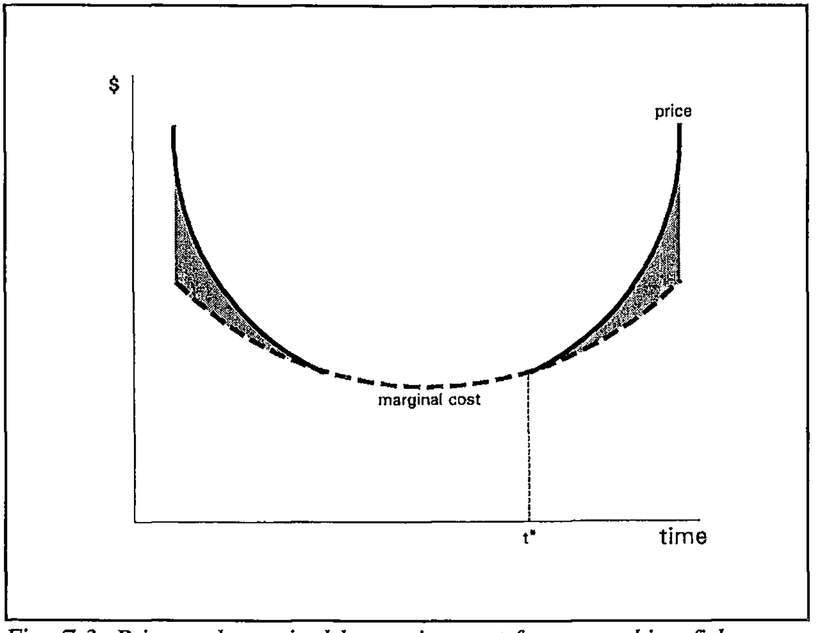

Fig. 7.3: Price and marginal harvesting cost for a searching fishery

For schooling fisheries, increased scarcity is expected to have a minor effect on harvesting costs. Therefore prior to policy implementation, the price is expected to fall constantly, although possibly at a declining rate, implying a convex, downward price path. Because of shifting from open access towards common property, policy intervention, if successful, will drive a wedge between price and marginal harvesting costs. The resulting price path can thus be convex downward, horizontal and U-shaped, but it should be upwardly biased compared with the pre-policy period (see Figure 7.4 for an example).

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[10]This issue will be discussed in section 7.3.

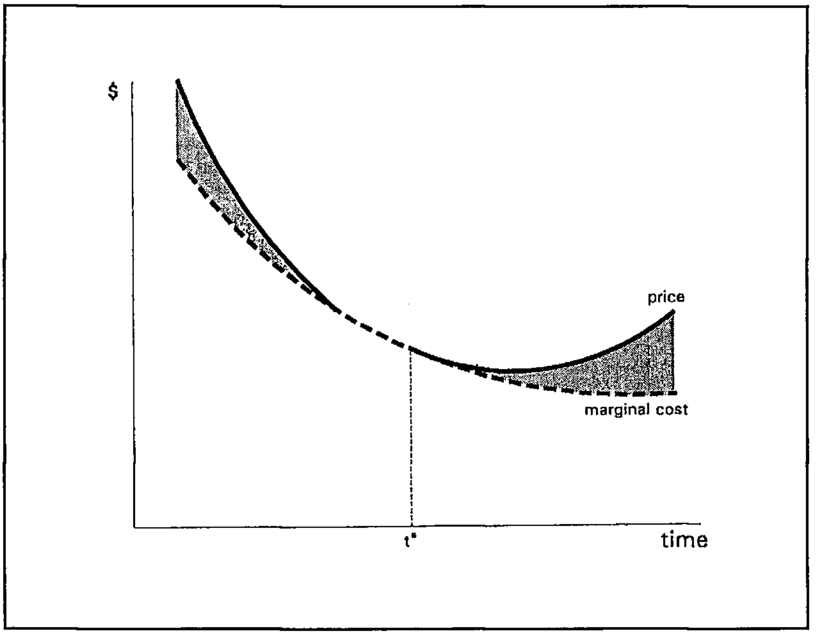

Fig 7.4: Price and marginal harvesting cost for a schooling fishery.

The most general function describing a U-shaped price path is (1), where α_1 is negative and α_2 positive.[11]

$$P_t = \alpha_0 + \alpha_1 t + \alpha_2 t^2 \qquad (1)$$

The simple model is applied to both schooling (herring, mackerel) and searching fisheries (cod, haddock, plaice and flounder). Based on the above-mentioned considerations with respect to marginal harvesting costs, α_2 is expected to be smaller for the schooling species.

[11]The simple quadratic model was also applied by Slade (1982), who attempted to reconcile the theoretical predictions of exponentially rising prices with empirical findings of falling prices of exhaustible resources. It should be observed that in contrast to Slade, the purpose of this paper was not to demonstrate the possibility of an exponentially rising rent path accounted for by a U-shaped price path, but rather that the time path of scarcity rent was U-shaped itself.

7.3 Case study: German fisheries

The hypotheses formulated in the previous section were tested for Germany, which was selected because of the availability of data.[12] The analysis was confined to six important commercial species: herring, mackerel, plaice, flounder, haddock and cod. Annual national statistics for the period 1967 - 1991 served as sources of data. The German fishery sector experienced some major changes in the 1980's. More specifically, the sector shrank by about 60%, partly because harvesting in the Northern Atlantic was restricted after implementation of the 200-mile exclusive zone.

The period considered might seem strange at first sight. After all, open access has long existed, which would support the hypothesis that a substantial part of the rent had vanished before 1967. A closer examination of the history of fishing, however, shows that global fish catch really increased from the fifties onward (see World Resources Institute, 1993). This also holds for Germany. The short period under consideration might thus be justified by the fact that large scale fishing operations are a relatively recent phenomenon. In addition, the increase of global fish catch from 1967 to the late eighties indicates that marine fisheries were profitable and attracted new capital continuously. This suggests that, in particular in the beginning of the 1967-1991 period, it must have been possible to capture a rent.

The policy effect will be analyzed by estimating model (1) on the basis of the data set for the open access period (1967-1982) and by comparing the actual $p(t)$ and predicted $\hat{p}(t)$ for t after 1983 in terms of the studentized residuals. $\hat{p}(t)$ is predicted on the basis of model (1) estimated for the open access period only.[13] A studentized residual smaller than 2 in absolute value

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[12]This does not imply that the conclusions drawn relate to Germany only. They may also apply to European developments if trade in fish is free, and prices equalize. A more appropriate approach to test the success or failure of the European policy would be to do a similar analysis for each of the Atlantic European member states or by pooling cross section and time series data for all these member states. Lack of data makes this impossible.

[13]Analysis of the autocorrelation indicated a first order autoregressive model (AR(1)). Maximum-likelihood was used to handle this problem.

is taken as an indication of no policy effect. Since there was no clear-cut moment of policy intervention and because of possible time lags for biological reasons (see Holden, 1994), policy effects are not expected to materialize immediately after 1983 when the CFP took shape. This uncertainty about the timing of the policy effects will be dealt with as follows. For each observation beyond 1982 with a studentized residual smaller than 2 in absolute value, it is assumed that the policy effect has not yet materialized. The observation is added to the open access data set and the procedure is repeated for the next observation. If from a given year onward a sequence of studentized residuals larger than 2 shows up, we will conclude that significant policy effects have materialized in that particular year.

The following results were obtained. First, estimating model (1) for the entire period gives significant negative coefficients for α_1 and significant positive estimates of α_2 for all the species selected (this result is not reported in a table). The expected U-shape as described in the previous sections is thus found. (It is of interest to note that the implied U-shape does not materialize for the coefficient as estimated for the open access period, as reported in Table 7.1.)

Second, Table 7.1 shows the estimated quadratic price paths for the selected fish species in Germany. Figures in brackets below the coefficients are t-values. A t-value larger than two in absolute value is considered to be significant. Log.lik indicates the value of the log likelihood function, which is a goodness of fit indicator, SR is short for studentized residual, and the figure in brackets following SR is the year concerned. ρ indicates the autocorrelation factor of the data set, and * indicates the year that a significant SR showed up for the first time. The models shown in the upper part of Table 7.1 relate to the pre-invention period and hence need not follow a U-shaped pattern which, as argued in section 7.2, is expected to hold for the period as a whole. All models given in the Table show a downward sloping trend for the pre-invention period, which is consistent with the arguments spelled out in section 7.2.

Table 7.1: Regression estimates for the price paths of selected fish species in Germany

	Herring	**Mackerel**	**Haddock**	**Cod**	**Plaice**	**Flounder**
α_0	0.51 (7.56)	0.54 (8.37)	0.12 (15.5)	1.53 (11.2)	1.18 (14.1)	1.16 (13.7)
α_1	0.03 (2.35)	0.005 (3.60)	0.002 (1.24)	-0.099 (-3.35)	0.025 (1.28)	-0.026 (-1.31)
α_2	-0.002 (-3.71)	-0.002 (-3.91)	-0.002 (-1.91)	0.004 (2.72)	-0.002 (-1.91)	0.0005 (0.56)
ρ	0.34 (1.32)	-0.023 (-0.91)	0.18 (0.65)	0.09 (0.33)	0.25 (0.96)	0.29 (1.25)
Log.lik.	41.9	30.8	19.8	92.4	36.0	29.2
SR(83)	0.29	1.94	0.33	0.30	0.73	-1.42
SR(84)	0.03	0.53	0.59	0.04	0.25	0.69
SR(85)	0.76	-0.40	2.57*	0.95	2.02*	0.46
SR(86)	2.70*	0.35	6.01	2.73*	5.17	3.28*
SR(87)	4.68	3.13*	8.52	3.80	8.69	6.14
SR(88)	4.04	5.68	7.88	3.16	7.16	4.87
SR(89)	4.58	5.07	6.28	3.38	6.38	5.03
SR(90)	4.34	5.69	6.85	3.98	6.23	3.79
SR(91)	3.70	6.91	6.70	3.40	8.96	N.A.
DW	1.7	2.0	1.9	1.7	1.7	1.8

The results for all the species concerned described in the lower part of Table 1 are clear: price development in the common property period was "upwardly biased". For haddock a number of outliers showed up starting in 1985. For

herring, cod, plaice and flounder the starting point was 1986. For mackerel a number of outliers were found from 1987 onward. Hence, for all species a wedge was developed between actual price and expected marginal harvesting costs, which we have interpreted as the return of rent as a component of prices. The analysis clearly indicates that there was a time lag (because of several reasons mentioned above) before policies became effective.
Conclusions with regard to policy effectiveness, however, should be drawn with caution. Since policy intervention and other relevant variables are not explicitly included in the model, factors other than policy may have contributed to the U-shaped price path. The price of gasoline, for instance, is a factor that has great influence on marginal harvesting costs and hence the price of fish. Gasoline price, however, did not rise in the period in which the increase in price was most profound. On the contrary: real prices fell, especially from 1987 onwards (Salz 1991). This is additional support for the conclusion that policy has been effective, since falling gasoline prices result in an underestimation of the true increase of the rent. Another factor that should be taken into account in the context of rent underestimation is increasing stock levels (as a result of the quota system) that could possibly mitigate the stock effect on harvesting costs. Since the analysis assumed constant coefficients for the harvesting cost function, the true increase in marginal harvesting costs was overestimated, which automatically implies that the true increase in rent was underestimated.

Another factor that should be taken into account is product specific capital and non-salvageable costs. Product specific capital implies capital that can be used only for one or a limited number of activities. If these activities cease, the fixed costs associated with the capital cannot be salvaged by shifting these to other activities (see Munro and Scott, 1985). If applied to fisheries, and if non-salvageable fleet costs exist, fleet redundancy becomes important. This might imply that implementing a quota system results in a temporary excessive capacity and hence excessive costs. These might be reflected in a rise in the price (higher costs/unit of fish). However, because of the earlier-mentioned stepwise reduction in quota which made structural fleet adjustments possible, this effect is likely to be small. Harvesting costs could also rise in the short run because of investment in new technologies and power capacity as a consequence of more time spent idle in the harbour, thereby possibly increasing harvesting costs in the short run (i.e capital stuffing, see section 2.5). One way to investigate both claims is to analyze the net returns of German fisheries. Under the assumption of excessive harvesting costs because of product specific capital or new investments, one would expect a falling, or at best a stable net income. It appears, however, that net income has developed positively in recent years (Salz, 1991). In addition, Salz (1991) argues that the size of individual vessels has been relatively constant, thereby further reducing the validity of the second claim. The claim that government and Union policy has resulted in more efficient harvesting does not seem to be invalidated by this category of factors. However, it is recognised that it would be useful to supplement the analysis as presented in this paper by explicitly modelling and estimating harvest technologies, along the lines of Dupont (1990).

7.4 Conclusion

In this chapter we have examined the effects of policy implementation, that followed declaration of sovereignty in a 200 miles zone, in terms of the return of rent in German fisheries. The method employed consisted of estimating a model on the basis of a time series of prices for the open access period and to estimate studentized residuals for the following common property period. Studentized residuals larger than 2 were taken to imply significant policy effects.

The results show that for all six species selected, studentized residuals exceeded 2 for a series of observations. Scarcity rent was found to have returned as a component of prices. There are even reasons to expect that the true increase in rent was underestimated in this analysis, although this is far from sure because we have not modeled (and analyzed) the production side of fishing explicitly.

Despite this finding of increased intertemporal efficiency, there is evidence that the outcome achieved is not yet fully efficient, so that potential rent is still dissipated to a certain extent. Large potential gains in income can be made by more radical changes of the sector (see section 1 and Schmidt

1993).[14] According to a report from the Commission to the Council and the European Parliament on the Common Fisheries Policy (cited in Schmidt 1993), the income foregone through sub-optimal fisheries management in the E.U. has been calculated at around $2.5 billion annually. In addition, it can be severely doubted whether over-fishing has been successfully halted.

An interesting topic for further research may be to analyze whether similar results also hold for other member states of the European Union. According to Salz (1991), the fisheries sector in many countries (Belgium, Ireland, United Kingdom, Spain, Portugal, France, the Netherlands) has performed well in recent years, at least those sectors that are not dependent on fisheries rights in other parts of the world.[15] On the other hand, there have been great differences among countries in species harvested, techniques applied, organization of the sector and national policy, so some differences are expected to arise. Especially with respect to rent dissipation due to implementation of inefficient and ineffective policies, resulting in capital stuffing, fleet redundancy and sub-optimal fleet composition, substantial differences between various member states are expected to exist. Detecting these differences, however, requires a more elaborate model than the one applied in this chapter.

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[14]These "radical changes" usually include (temporary) cutbacks in fishing effort, and loss of income for fishers in the short run. See chapter 9 for one possible approach to facilitate implementation of such policies.

[15]Spain, for instance, lost the right to fish near Namibia and Norway.

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CHAPTER 8

POLITICAL PREFERENCES AND FISHERY MANAGEMENT[1]

Abstract

Models of optimal fishery management traditionally assume social welfare optimization as the goal of the managing agency. However, public choice theories suggest that politicians do not choose policies to maximize welfare for society. Instead, policies reflect the interests of politically powerful groups. In this paper, we examine fishery management under alternative political preferences. We find that the steady state values in the political optimum may be both greater and smaller than steady state values in the efficient case, varying according to the relative weight of different interest groups.

8.1 Introduction

After implementation of exclusive economic zones in the late 1970s, *de jure* open access for many fisheries came to an end (see chapter 7). Typically, a national regulatory agency or supra-national organization determines harvest levels and is responsible for enforcement of regulations. A common assumption in the normative economic literature on fishery management is that this organization chooses instruments to maximize the net present value of economic benefits accruing to society as a whole. In the case of a fishery, these benefits would include scarcity rent due to the fishery, consumer surplus, and rents for factors of production that are not supplied at a

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[1]This chapter is based on Horan R.D., E.H. Bulte and J.S. Shortle (1996), Political Preferences and Fisheries Management, Penn State University Working Paper. It is part of a more elaborate paper: Horan R.D. J.S. Shortle and E.H. Bulte (1996), Fishing for Dollars: the Impact of Political Preferences on Fisheries Management, Penn State University Discussion Paper.

constant cost to the fishing industry (with exogenous prices for inputs and outputs, this amounts to maximizing rent to the fishery). Examples include Bjorndal (1988); Clark (1990); Neher (1990); Boyce (1995) and several chapters in this thesis. Models along these lines determine *economically optimal-*, or *efficient* harvests and stocks.

However, other possible benchmarks exist. Typically, agencies consider the welfare of politically powerful interest groups as well as equity issues (Gardner 1987; Shortle and Laughland 1994; Mueller 1989). Anderson (1984) discusses trade-offs between producers and consumers of fish which may be taken into account by policy makers. Clark (1985, p.144) also lists many possible objectives of fisheries regulation and adds that "in real life there is no question that distributional questions tend to dominate the decisions of management agencies, which must deal face to face with the people whose livelihoods are affected by regulations." In particular, legislation may be developed to increase the well-being of those who provide labour services in the fishing industry. Hartwick and Olewiler (1986, p.292) write that "much existing regulation in the fishery is designed to sustain the fishery and increase the income of fishermen, not to reach a socially optimum equilibrium." Decision making with respect to the allowable catch is undoubtedly a political process (Holden 1994), with a range of interest groups involved. Coull (1993) argues this point, showing how politically set quotas for haddock by Norway and the European Union have consistently exceeded the advice of scientists in past years.

In this chapter, we examine fishery management under alternative political preferences. In particular, following previous literature on regulation in other sectors [e.g. Bullock (1994); Rausser *et al.* (1982); Peltzman (1976); Becker (1974); and Shortle and Laughland (1994)], we consider the design of policy and steady state outcomes when the regulating agency seeks to maximize a political preference function (PPF).[2] The underlying argument is that

[2]Broadly speaking, public choice models can be divided in so called "voting models" and "interest group models" (van der Zee 1997). According to Potters and van Winden (1996), the latter class of models can be subdivided in 4 types of models: (i) models with an influence function; (ii) campaign contribution models; (iii) models with asymmetric information; and (iv) models with a composite utility function. The political preference function approach adopted in this chapter is one possible specification of the latter type, and has been applied most extensively

the agency's preferences are an equilibrium outcome of the interaction of interest groups and political decision makers in political markets. We define *politically optimal* levels of harvest and stock as levels maximizing this PPF. The influence-generating activities of the different interest groups and the process of competition for influence amongst these actors remain a black box.

In this chapter, one politically optimal policy scheme is presented: we have (rather heroically) assumed that the government has direct control over the factors of production in the fishery [for a more elaborate model that includes a two stage game between the government and the sector, and that includes economic incentives, see Horan, Shortle and Bulte (1996)]. We also assume that policy makers have an understanding of biological productivity and current stock levels. For "search" fisheries (fisheries with the biological stock as an argument in the harvest production function, see Neher 1990), we find that the relationship between steady state values in the political and economic optima is ambiguous. Politically optimal steady state values may be larger or smaller than economically optimal values, depending on the agency's preferences over the interest groups. This sheds new light on the often heard claim that many resource stocks are over-fished [FAO (1992)]. "Overfishing" (or underfishing) as measured by traditional benchmarks may be deliberate and the result of politically rational policies.

Interestingly, we find that political preferences with respect to different groups within the economy do not affect stocking decisions for "schooling" fisheries: the economically and politically optimal stock sizes coincide here, provided public and private discount rates are the same. However, investment and labour employment decisions will differ from the economic optimum.

Finally, in addition to favouring some politically powerful interest groups over others within the current generation, politicians that aim to be re-elected may also favour the current generations over future generations. This implies the rate of time preference applied by politicians may exceed the market or social discount rate [Oehmke and Choe (1991)]. We will not return to this issue in the remainder of this chapter.

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in the literature to study the policy making process.

8.2 The Model

Suppose that a fishing agency seeks to maximize a political preference function for a fishery, defined over consumer surplus (*CS*), resource rents (*R*), labour rents (*LR*), and net government transfers to the fishery (*G*). The PPF is written as $U(CS,R,LR,G)$. U is assumed to be increasing in *CS, R,* and *LR*, and decreasing in *G*. In addition, U is strictly quasi-concave. Thus the agency values the economic well being of consumers, producers, and labour employed in the fishery, while at the same time exhibits disutility from transfer payments (or conversely, utility from receipts such as taxes). These groups do not necessarily share the same interests. We assume that the agency's preferences will reflect the outcome of the interplay of these groups in political-economic markets. The resulting preferences are represented by the PPF. Unlike Niskanen (1977), we do not assume that the agency is concerned with maximizing revenues for its own welfare. Instead, following Peltzman we assume the agency values changes in its budget because of the effects this has on taxpayers. We do not explicitly model the process by which the agency incorporates taxpayers' welfare into the PPF. Instead it is assumed that this is the result of interactions in the political arena. Net government transfers may be financed by a reduction in spending on other government programs or an increase in taxes. In either case, there is a loss of welfare for some political constituency.[3]

Let h represent harvests per boat, with n being the number of boats. The inverse demand for fish is $p(Q)$ ($p'<0$). Let l represent labour per boat and let L be the aggregate quantity of labour supplied with labour supply given by $w(L)$ ($w'>0$). Capital investment per firm is denoted by I and $C(I)$ is the cost of investment ($C'>0$, $C''>0$). Capital is denoted by k. Under these specifications, we have the following definitions for the arguments of U:

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[3]The source of funding can have economic efficiency implications. We do not explore this issue here.

$$CS = \int_0^Q P(y)dy - P(Q)Q \tag{1}$$

$$R = P(Q)nh - w(nL)nl - nC(I) \tag{2}$$

$$LR = w(L)nl - \int_0^L w(y)dy \tag{3}$$

The argument G will equal net transfer payments in the fishery. For example, in the case of a per unit tax on harvests, $G=-\tau nh$, where τ is the per unit tax rate.

The partial derivatives of the PPF with respect to its arguments are the marginal political utility associated with the different interest groups. Define:

$$e_{ij} = = [\frac{U_j}{U_i} - 1] = [|MRS_{ij}| - 1] \quad \forall i,j = CS, R, LR, G$$

where MRS_{ij} is the marginal rate of substitution of political utility between sectors i and j. The marginal preferences for sectors (i.e. e_{ij}) will be an equilibrium outcome of the model when the PPF is nonlinear in its arguments. However, marginal preferences are invariant with respect to economic welfare when the PPF is linear in its arguments.

If $e_{ij}>0$, the agency exhibits a marginal preference for sector j relative to i. Alternatively, the agency has a marginal preference for sector i relative to j if $e_{ij}<0$ and a neutral marginal preference between the two sectors when $e_{ij}=0$. Political preferences will be to some extent a response to lobbying efforts and other political pressure by each group.

Due to free rider effects and increasing transaction costs associated with large group sizes, it is reasonable to assume that large, organized groups such as firms will be able to exert more political pressure than larger, but unorganized groups such as consumers [Mueller (1989); Olson (1965); Peltzman (1976)]. This would suggest the agency may have pro-firm marginal preferences relative to consumers. Conversely, it is more difficult to say whether firms or organized groups of fishing labour have more political influence.

8.3 The Politically Optimal Steady state

The PPF is an expression of the goals of fishery management. What is accomplished depends, in practice, on the instruments the agency chooses to use (e.g. tradeable quotas vs. tax on the catch), the ways in which it uses them (e.g. high tax rates or low tax rates), the competitive structure of input and landed fish markets, and the property rights in the fishery. In this chapter we assume that government intervention takes the form of imposing direct control on all the firm's choice variables (*h, I,* and *l*), leaving firms with no degrees of freedom in operation decisions. The agency also controls access to the industry, thus regulating the number of vessels operating in the fishery (*n*).[4] [In reality, intervention might take the form of mixtures of price controls, quantity controls, taxes, and subsidies. For an overview of fishery management instruments and political preferences, see Horan, Shortle and Bulte (1996)]. With price taking in labour and landed fish markets, market clearing implies $nl = L$, $nh = Q$, $p = p(nh)$, and $w = w(nl)$.

Given the above specifications of the model, the agency will choose harvests, the number of boats, labour, and investment to maximize the present value of the PPF subject to the growth of the fish and capital stocks:

$$\underset{h,n,L,I}{Max} \int_0^\infty U(CS,R,LR,G)e^{-rt}dt \qquad (5)$$

subject to

[4]In practice, inputs may be controlled through quotas on input use and restrictions on the type of technology in use. Vessels may be limited through the sale of licenses or administration of costless permits. We will assume the latter situation and so the analysis is simplified as there are no transfer payments to consider.

$$\dot{x} = g(x) - nh$$
$$\dot{k} = I - \gamma k \qquad (6)$$
$$h \leq f(k,l,x).$$

where x is the stock of fish, $\gamma \in [0,1]$ is the constant rate of depreciation for capital, and $f(K,L,x)$ is the harvest production function. Following Neher (1990), for a schooling fishery $f_x = 0$ and for a searching fishery $f_x > 0$. The production function is increasing in all arguments and is strictly concave. The growth function for the stock, $g(x)$, is assumed to have the usual properties (positive and strictly concave over the domain $(0,X)$, with $g(0) = g(X) = 0$ and where X is the maximum stock which can be supported by the ecosystem). The rate of time preference is given by r. Note that the agency's problem reduces to a standard normative (efficient) economic model when the PPF is linear in its arguments, the weights on the arguments are equal, and the rate of time preference is the social rate.

The agency's problem can be simplified by substituting $f(k,l,x)$ in for h and thus eliminating one of the choice variables. This assumes that harvesting is technically efficient, which is what is to be expected in the optimum. Having assumed identical firms, we can aggregate over the costate variable for capital. The current value Hamiltonian for this problem is therefore:

$$H = U(CS,R,LR,G) + \lambda_1[g(x) - nf(k,l,x)] + n\lambda_2(I - \gamma k). \qquad (7)$$

By the maximum principle, the necessary conditions for an optimal solution are:

$$\lambda_1 = -\left[U_{CS} - U_R\right]p'nf + U_R[p - \frac{w}{f_l}] + [U_{LR} - U_R]\frac{w'nl}{f_l} \qquad (8)$$

$$\lambda_1 = -[U_{CS} - U_R]p'nf + U_R[p - \frac{wl + C(I)}{f}] + [U_{LR} - U_R]\frac{w'nl^2}{f} + \lambda_2\frac{I - \gamma k}{f} \qquad (9)$$

$$\lambda_2 = U_R C'(I) \tag{10}$$

$$\dot{\lambda}_1 = (r - g' + nf_x)\lambda_1 + [U_{CS} - U_R]p'n^2 f(\cdot) f_x - U_R p n f_x \tag{11}$$

$$\dot{\lambda}_2 = (r + \gamma)\lambda_2 + [u_{CS} - U_R]p'nf(\cdot)f_k - U_R[pf_k] + \lambda_1 f_k \tag{12}$$

as well as the equations of motion for fish stock and capital.[5]

These conditions reveal several interest-group trade-offs facing the agency. Equation (8) requires labour employment in the fishery such that the political marginal net benefit of additional employment (the RHS) is equal to the political marginal opportunity cost of the additional harvest λ_1. The first term on the RHS is the political value of the price reduction induced by additional employment, $-[U_{CS}-U_R]p'h$. In particular, other things equal, additional labour increases the harvest. Price must fall for the market to clear. Consumers benefit from reduced prices while producers are made worse off. The willingness of the agency to pursue additional employment therefore depends, in part, on the relative political value of consumers and producers (i.e. $U_{CS}-U_R$). The second term on the RHS is the additional rent received by producers due to the increased harvest brought about from additional labour, weighted by marginal utility, $U_R[p-w/f_l]$. This term would be zero in a competitive market with open access. However, it is positive here due to the intertemporal cost associated with harvesting that would not taken into account by the market. The agency's willingness to sacrifice current labour income to conserve the resource for future use is motivated by the political value of future harvests. Finally, the last term reflects the wage bill effect from hiring of additional labour, $[U_{LR}-U_R]w'l/f_l$. The wage may be driven up as more labour is hired, making labour better off but firms worse off. Accordingly, agency choices about employment will depend on the relative political value of labour and producers as well as the political value of labour and consumers.

[5]These optimality conditions are derived respectively from the following conditions: $H_l=0$, $H_n=0$, $H_I=0$, $\dot{\lambda}_1 = r\lambda_1 - H_x$, and $\dot{\lambda}_2 = r\lambda_2 - H_k$.

Equation (9) requires that the number of licenses be such that the marginal net benefits from an additional boat (the RHS) are equal to the political marginal opportunity cost of an additional harvest. The first term on the RHS has the same interpretation as its counterpart in (8). The second term on the RHS is the political value of the per unit profit to an additional boat, $U_R[p-(wl+C(I))/f]$. This term would be zero in a competitive market equilibrium with open access. The third term is the political value of the increase in the wage bill from the addition of another boat, $[U_{LR}-U_R]w'nl/f$. An additional boat increases the amount of labour demanded. If $w'>0$, then wages must rise for the market to clear, reducing producer profits. Accordingly, the agency's willingness to pursue additional labour employment will depend on the relative marginal political value of labour and producers. Finally, the last term represents the benefit associated with the per unit net investment from the introduction of another boat into the industry. The agency must weigh the political gain from profits to additional licenses with the forgone profits to existing licenses which would result as a consequence of additional entry into the fishery.

Equating (8) and (9) yields the following relationship defining the optimal scale of production:

$$[w(\frac{1}{f_l}-\frac{l}{f})-\frac{C(I)}{f}]=[e_{R,LR}]w'nl[\frac{1}{f_l}-\frac{l}{f}] \tag{13}$$

The term w/f_l is the marginal cost of production (*MC*) for a representative firm. Therefore, the LHS is the difference between a firm's *MC* and average cost (*AC*). An economic efficiency requirement is that this difference be zero. To see this, recall that economic efficiency dictates that the marginal weights applied to each sector be set equal to one. The agency therefore would exhibit neutral marginal preferences between sectors (i.e. $e_{R,LR} = e_{R,CS} = 0$) and the RHS of (13) vanishes. In other words, firms would operate at minimum average costs (*MAC*). In this model, it may not be politically optimal for firms to operate efficiently.

It is possible to infer relative political weights according to the PPF by observing actual policies. Suppose that the equilibrium solution to the model is such that $AC>MC$. Assuming that the system is in (or near) the steady state and that $(1/f_l - 1/f) > 0$, the agency will thus exhibit a marginal preference for firms relative to labour at the political optimum (i.e. $e_{R,LR} < 0$). Alternatively, if $(1/f_l - 1/f) < 0$, the agency will exhibit a marginal preference for labour relative to firms at the political optimum (i.e. $e_{R,LR} > 0$). In the absence of output restrictions, firms would not choose to operate at such a point as they would be earning an economic loss. However, if output per firm, total output, and entry into the fishery is restricted such that $P > AC$, then firms would be willing to operate at this point.

If $e_{R,LR} = 0$ at the optimum (i.e. the agency has a neutral marginal preference between firms and labour), firms should operate at *MAC*. However, while firms operate at an efficient scale, the politically determined equilibrium will still not be economically efficient unless it is also true that $e_{R,CS} = 0$. The final case to consider is when $MC > AC$. An equilibrium outcome will require that $e_{R,LR} > 0$. Here, labour is valued more than firms at the margin. There are political benefits to expanding output per boat, thereby reducing the rent per boat, because other things equal, increasing output per boat requires more labour per boat.

Setting $\dot{\lambda}_1 = \dot{\lambda}_2 = \dot{x} = \dot{k} = 0$, the following set of equations along with (13) defines the political steady state, denoted x^*, k^*, h^*, n^*, I^*, and l^* (assuming one exists):

$$r = g'(x) + \frac{\dfrac{nwf_x}{f_l} - e_{R,LR}[\dfrac{w'n^2lf_x}{f_l}]}{-e_{R,CS}[p'nf] + [p - \dfrac{w}{f_l}] + e_{R,LR}[\dfrac{w'nl}{f_l}]} \tag{14}$$

$$(r+\gamma)C'(I) = \frac{wf_k}{f_l} - e_{R,LR}[\frac{w'nlf_k}{f_l}] \tag{15}$$

$$g(x) = nh \tag{16}$$

$$I = \gamma k \tag{17}$$

The political optimality conditions (14) and (15) reduce to the economic optimality conditions (see below) when the agency exhibits neutral marginal preferences (i.e. $e^*_{R,LR} = e^*_{R,CS} = 0$) and the agency's rate of time preference is the social rate.[6] Equation (14), equates the agency's rate of time preference with the marginal product of the *in situ* resource stock in fish production plus a term which we will refer to as the resource stock term. The resource stock term for the political optimum is the rate of substitution between the marginal political cost savings from leaving a fish *in situ* (since less effort will be needed when the stock is large) and the marginal political benefits of harvesting. The larger the resource stock term, the higher the optimal steady state stock level should be (since $g^{//}<0$). Thus higher cost savings have a positive impact on the size of the stock while current rents have a negative impact. When (14) reduces to the economic optimum, it is a general formula for the *golden rule* that is commonly derived in fisheries literature [e.g., see Clark (1990); Neher (1990)].

The larger the marginal political preference of consumers relative to firms in equilibrium, $e^*_{R,LR}$, the larger is the denominator of the resource stock term in (14), other things equal. Given $g^{//}<0$, this suggests that x^* moves inversely with $e^*_{R,CS}$. The political value of harvesting is larger with higher values of $e^*_{R,CS}$ since additional harvests (at lower prices) increase consumer welfare.

The larger the marginal political preference associated with labour relative to firms, $e^*_{R,LR}$, the larger is the denominator and the smaller is the numerator of (14). The denominator is larger because of lower marginal costs in terms of the PPF associated with increasing the amount of labour used in harvesting fish. The numerator is smaller because the marginal political cost

[6]When the marginal political utility across sectors are equal (i.e. $U_{LR} = U_R = U_{CS}$) the political and economic steady states coincide. However, this is not equivalent to the economically efficient case in which the marginal political utility in each sector is constant and equal to one in all periods (i.e. $U_{LR} = U_R = U_{CS} = 1$). Although both situations will lead to the efficient steady state, only the latter will cause the system to approach along an optimal path (see chapter 6).

savings of leaving fish *in situ* will be smaller since a positive weight is applied to the workers who catch fish. A larger stock left *in situ* means that less labour will be required in the future for a given harvest, lowering future wages and representing a loss of welfare to the agency. The net effect of a greater marginal political preference for labour relative to firms is a smaller x^*.[7]

Equation (15) basically equates marginal political costs of capital with marginal political costs of labour. The larger is $e^*_{R,LR}$, the smaller will be the RHS of (15), other things equal. Given $C''>0$, (13) implies that investment and capital will be reduced.

Unfortunately it is not possible to make any general comparisons between the politically optimal and economically efficient outcomes without further specification of the model. As is readily derived, the *economically efficient* counterparts of equations (14) and (15) are:

$$r = g'(x) + \frac{wn\dfrac{f_x}{f_L}}{p - \dfrac{w}{f_L}} \tag{18}$$

$$(r + \gamma)C'(I) = w\frac{f_K}{f_L} - v \tag{19}$$

Equations (18) and (19) along with (16) and (17) determine the efficient solution, x^e, k^e, I^e, n^e, h^e, and l^e. If no other changes occurred (or if the changes were small), then when $U_{CS}^* > U_R^*$ and $U_{LR}^* > U_R^*$, it would unambiguously be the case that $x^* < x^e$ and $k^e < k^*$. Conversely, when $U_{CS}^* < U_R^*$ and $U_{LR}^* < U_R^*$, then we would have $x^* > x^e$ and $k^e > k^*$ with other combinations of weights providing ambiguous results. However, the values of other variables are not constant: they are free to change in order to return the

[7]Note that x^* can never fall below the schooling equilibrium value since the stock term in (14) is always positive. To see this, note that the numerator of the stock term can also be shown to equal $[(r+\gamma)nC'/]f_x/f_k$, which is always positive as is the denominator, λ_1.

system to equilibrium, and since the production function relies on three inputs, it is impossible to know how these variables interact with specifying more fully the substitution effects in production. Therefore, we cannot say in general whether or not the political steady state will exhibit larger or smaller steady state fish stocks, nor can we comment on the extent of capitalization relative to the efficient case. Since the relative size of the fish and capital stocks are uncertain, so are harvest and investment levels.

In a schooling fishery with $f_x=0$, we have that $x^* = x^e$ from equations (14) and (18). However, steady state levels of capital, investment, and labour still differ between the two equilibria. Since x^* is defined by (14), comparison of (15) and (19) shows that when $U_{LR}^* > U_R^*$, we have some combination of $k^* < k^e$, $I^* < I^e$, $l^* > l^e$ or $n^* > n^e$. The reverse situation holds when $U_{LR}^* < U_R^*$. Thus the politically optimal equilibrium of a schooling fishery is, *ceteris paribus*, overcapitalized relative to the efficient solution when firms are valued more than labour at the margin. However, without further specification of the production function, the degree of capitalization in the fishery is ambiguous.

8.4 Conclusions

The standard assumption in normative fishery economics is that governments implement regulations in order to maximize the present value of net benefits to society. This is unrealistic. Public choice theories suggest that governments may maximize a political preference function with unequal weights attached to the economic objectives of various interest groups. In this paper we recognize that, in addition to resource rents, governments may consider labour rents, consumer surplus, and their own costs or revenues associated with regulations when designing fishery policy.

The politically optimal steady state fish stock equilibrium for a searching fishery is shown to differ from the efficient equilibrium. The difference depends on the relative political welfare placed on each sector valued by the agency at the margin. Depending on the structure of the PPF, it is argued that the efficient capital and fish stock may be greater than or less than the political stock. For a schooling fishery, the fish stock levels will coincide. However, labour and capital may be employed at inefficient levels.

In Horan, Shortle and Bulte (1996), we have demonstrated that the implementation of certain policy instruments requires significant restrictions if a steady state equilibrium is to occur under general conditions in the context of a decentralized economy. The use of policies which provide an efficient outcome in the efficient model no longer produce a first best solution in the political model. Additional instruments must be used to insure that the firms, who are the ultimate decision makers in the fishery with respect to input choices, take into account the economic welfare of other sectors which the agency values. For example, with a conventional quota system the government is (theoretically) able to control harvesting and "set" a steady state x^*, so that the firm's problem reduces to minimizing costs, given the quota. However, the political cost in terms of unsatisfied consumers and labourers may be so high that the government may be better of setting quotas at a different level. In contrast, with a combination of a tax on capital and a subsidy on labour costs and the cost of investment, it is demonstrated that the government can mimic the command optimum. Interestingly, political inferiority because of imperfect control over the factors of production may correspond with model solutions that are closer to the economic steady state than the command optimum of the political model.

Although the model presented in this chapter is more general than standard fisheries models, it is by no means complete. For example, arguments for environmental pressure groups and preservation values attached to wild stocks might also be added to the PPF. Inclusion of all relevant actors is especially relevant for future empirical work.

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CHAPTER 9

OVERFISHING, FISHING RIGHTS AND FUTURES MARKETS[1]

Abstract:

Introducing a futures market for fishing rights would increase utility for risk averse fishermen. We use the EV model to analyze possible reductions in (expected) profits for futures trading that would make fishermen indifferent between the situation with and without futures markets. It is found that a futures market for fishing rights enables policy makers to pursue substantial cuts in the size of annual quotas without hurting fishermen. In light of current overfishing and pressure from the sector to avoid dramatic cutbacks in effort, this is an important policy result.

9.1 Introduction

After implementation of exclusive economic zones in the late 1970s, *de jure* open access for many fisheries came to an end. It has been well documented that this did not imply that over-harvesting of fish resources came to an end (FAO 1992). *De facto*, many fisheries still exhibit features of open access management.[2] Conrad (1995) writes that approximation of open access conditions can arise if management regulations are ineffective, for instance because fishermen are adaptive and eliminate or reduce the effect of regulations. Another reason for overharvesting exists. Interest groups pressure governments or supra-national organizations to set quotas at too high levels (see for instance Holden 1994). For example, in 1995 the European Commis-

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[1]This chapter is based on Bulte E.H. and J.M.E. Pennings, A note on Overfishing, Fishing Rights and Futures Markets, *European Journal of Law and Economics*, In Press

[2]Bulte *et al.* (1995) found that establishing property rights to end conditions of open access in Europe has contributed to increased efficiency of exploitation for German fisheries. However, the current situation is still far from optimal, according to many authors (e.g. Schmidt 1993).

sion proposed drastic reductions of fish quota to protect fish stocks, but after a process of negotiations these propositions were substantially mitigated to achieve a balance between ecological and socio-economic considerations (de Graaf 1995). In this paper we recognise that revision of European fisheries policies without particular attention for the (financial) consequences for the sector may fail.[3] We demonstrate how, under an individual transferable quota (ITQ) scheme, harvest levels can be reduced without reducing utility of (risk averse) fishermen.

The European Union annually determines total allowable catches (TACs) for the European fishery sector. Total allowable catch is divided among the member states, who distribute the national quota among their fishermen. Member states have a certain freedom to formulate national fisheries policies, as long as total catch does not exceed the national quota, and the policy does not conflict with European legislation. There are considerable differences in national fisheries policies between different member states. The Dutch government, for instance, decided to implement an individual transferable quota system (ITQ). Initially this system was implemented just for flatfish, but in recent years also for demersal species. Under an ITQ scheme, fishermen are allocated an individual quota, based on their historic catch. Fishermen are allowed to trade their fishing rights, so that, in theory, fish are caught by the most efficient fishermen in the sector: they are able to "outbid" the other fishermen on the quota market, hence quota gravitate towards the most efficient fishers.[4] The price for fishing rights have fluctuated widely in recent years.

Trading quota implies trading *judicial harvest capacity*. In addition, fishermen invest in vessels and gear, the so called *physical harvest capacity*. In the short and medium run, these investments result in fixed costs for the fishermen. For that reason, fishermen are committed, to a certain extent, to buy, sell or lease quota to achieve some balance between physical harvest

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[3]The reader is referred to the vast literature on interest groups and public choice (e.g. Olson 1965, Mueller 1989) for more information.

[4]For an overview of different policy instruments applied for fisheries, see Hartwick and Olewiler (1986).

capacity and judicial harvest capacity. (That is, unless the quota that is handed out to the individual fisherman is just equal to the desired harvest capacity.) Since quota prices are known to be volatile, fishermen face a price risk on the quota market. By introducing futures markets for quota, the fishermen can reduce this risk. In the framework of the certainty equivalent model, we demonstrate that this will increase the utility of harvesting for risk averse fishermen. This implies that, to restore utility neutrality, there is room for the government to combat over-harvesting by *reducing* the size of the quota allocated to the fishermen. This is just the measure of *equivalent variation* applied to fisheries management. In the next section we will briefly explain the underlying model. In section 3 the model is empirically applied to the case of Dutch fishing under the ITQ system. The conclusions ensue.

9.2 The model

Consider a fisher who can lock in the price risks regarding fishing rights with the help of fishing rights futures. The fisher is risk averse and wishes to maximize the expected revenue in the next time period adjusted for risk, where risk is measured by the variance of the expected revenue. We assume that the objective function of this fisherman can be reasonably approximated by the expected value-variance (EV) model (Peck 1975, Kahl 1983, Robison and Barry 1987).[5] The objective function can be expressed as:

$$\pi_{ce} = E(\pi) - \lambda var(\pi), \tag{1}$$

where π_{ce} is the certainty equivalent, $E(\pi)$ is expected revenue and $var(\pi)$ represents the variance of revenues. In (1), λ denotes the risk parameter which, for risk-averse decision makers, is positive, thus demanding compensation for risk bearing (Pratt, 1954).

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[5]For the conditions that justify the use of the EV model and the discussion on the use of the EV model and the general expected utility model the reader is referred to Bigelow (1993); Pulley (1981); Coyle (1992); Meyer and Rasche (1992); Pope and Chavas (1994); Robison (1996); and Tew *et al.* (1991).

If uncertainty in determining the profit is caused by fluctuations in the price of fishing quota P and the output price z, then profit can be expressed as:

$$\pi = \tilde{z}Q - \left[\tilde{P}(Q - h) + P_f h\right] - C(Q) - F, \tag{2}$$

where Q is the quantity harvested (assumed equal to the size of the quota, which is a good assumption for sole fishing in the Netherlands); $\tilde{z}$ is the spot price for the output with expected value z and variance σ_z^2; $\tilde{P}$ is the spot price of the fishery right with expected value P and variance σ_r^2; and h is the quantity of hedged fishery rights, i.e., the quantity bought or leased on the futures market for known price P_f. It is assumed that there is no basis risk; $C(Q)$ are variable costs and F are fixed costs.

Expected profit is:

$$E(\pi) = zQ - \left[P(Q-h) + P_f h\right] - C(Q) - F. \tag{3}$$

The variance of profits is

$$var(\pi) = Q^2\sigma_z^2 + (Q-h)^2\sigma_r^2 - 2Q(Q - h)\sigma_{z,r} \tag{4}$$

where $\sigma_{z,r}$ is the covariance between the price of fish and the fishing right. By substituting (3) and (4) into (1), the optimal holdings of futures contracts is obtained by differentiating the objective function with respect to h and setting the first order condition equal to zero:

$$h^* = Q - \frac{P_f - P}{2\lambda\sigma_r^2} - Q\rho\frac{\sigma_f}{\sigma_r}. \tag{5}$$

where ρ is the correlation coefficient between fish prices and fishing rights. For $P_f > P$, there is a trade-off between (expected) revenues and variance such that $h < Q$. The more risk-averse the fisherman and/or the more price fluctuations in the spot market of the right, the greater the level of hedging for a constant positive difference between P_f and P.

The effect of futures markets on fish quota is graphically illustrated in Figure 9.1.

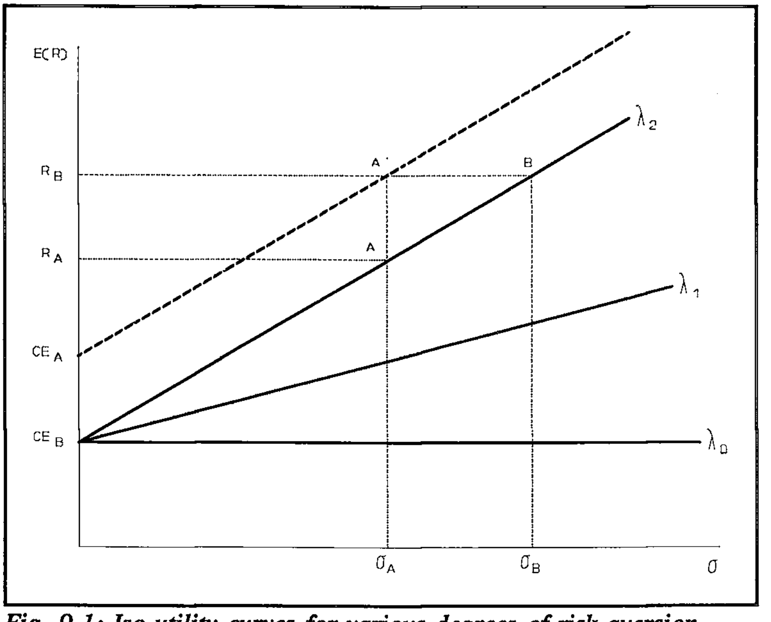

Fig. 9.1: Iso-utility curves for various degrees of risk aversion

In Figure 9.1, σ is the standard deviation of the revenues of harvesting, *E(R)* are expected revenues of harvesting and λ measures the degree of risk aversion: for $\lambda = 0$, the fisherman is risk neutral. Else, the higher λ, the more the fisher's risk aversion. Finally CE_B is the (initial) certainty equivalent. The lines diverging from CE_B are iso-utility curves, indicating a constant utility level for varying expected revenues and standard deviations. Increasing volatility[6] (i.e. moving from left to right in the graph) indicates how much expected revenues should rise to keep utility constant. Obviously, the more risk averse, the higher the risk premium demanded by the individual to make him not worse off with an increased spread, hence the steeper the iso-utility

[6]That is, increased the spread of the price distribution while keeping the mean constant.

line. Risk neutral fishermen ($\lambda = 0$) do not require additional expected revenues to compensate for increased fluctuations, hence the iso-utility line is horizontal.

By buying or selling (part of) the judicial capacity on the futures market instead of the future spot market, the price risk of this capacity is reduced and so is over-all volatility of revenues. Suppose trading quota at a futures market reduces over-all volatility from σ_B to σ_A. This implies that a fishermen with degree of risk aversion λ_2, moves from B to A'.[7] This makes him undoubtedly better off (given a degree of risk aversion λ_2, the new certainty equivalent is $CE_A > CE_B$). To restore utility neutrality, the policy maker next decreases expected revenues with the interval (R_B - R_A) so that the fisherman moves back to the initial iso-utility curve, hence from A' to A. Given a certain price per unit of fish, this reduction in expected revenues can be translated in a reduction of quota allocated to this fishermen. This model is applied for the Dutch situation in the next section.

9.3 Empirical results

Equation (4) in the previous section is used to find expressions for the variance of profits with and without hedging. For example, without hedging we find:

$$var_W(\pi) = Q^2\sigma_f^2 + Q^2\sigma_r^2 - 2Q^2\sigma_{f,r}. \tag{6}$$

Similarly, with hedging we find:

[7]Note that in this graph we assume that the futures price is equal to the expected future spot price. If trading on a futures market implies bearing some cost (for instance because $P_f > P$), then A' will be located below B. As long as the new position of the individual is in the interval (R_B - R_A), however, his utility has increased and the (qualitative) result is unaffected. Note that the individual would not decide to trade on the futures market if the futures price is so much lower than the expected future spot price that revenues fall below R_A.

$$var_H(\pi) = Q^2\sigma_f^2 + (Q - h^*)^2\sigma_r^2 - 2Q(Q - h^*)\sigma_{f,r}. \tag{7}$$

In this section we will calculate the reduction in $var(\pi)$ that results from optimal hedging for the Dutch sole fishery. We have assumed that, if fishers buy a quota, they expect to use it until the Common Fisheries Policy is open for revision, which is in 2003. This implies that they depreciate their quota in this period (we have assumed a discount rate of 10%), and we have used the capital recovery factor to determine annual quota costs. Prices of sole and fishing rights are from Produktschap voor Vis (1994) and Banning (193), respectively. Estimated parameters of interest are as follows: $z = 15.4$; $P = 12.35$, $\sigma_z = 2.6$, $\sigma_r = 3.0$ and $\rho = 0.56$ (implying that $\sigma_{z,r} = 4.37$).[8] We have defined γ as P_f/P, which is a measure for the divergence between the expected spot price and the futures price of the fishing right.

The reduction in supply of fish may increase the price per unit of fish, although we assume constant prices. Similarly, we have assumed that harvesting costs are unaffected by quota cuts. The assumptions of constant prices and constant harvesting costs are for computational convenience and are both conservative: If in reality prices would rise and/or costs would fall when fishing rights are curtailed, the fisher is (partly) compensated for the reduction in quota by high prices and/or cost savings, and even larger quota cuts are possible while keeping utility of fishermen constant. For that reason the results provided in Table 9.1 can be considered as underestimates of the true potential to cut quota.[9]

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[8]It is of interest to note that we have used yearly observations from 1987 onwards. Prior to 1987 enforcement of catches and landings was poor, hence the value of official fishing right was relatively modest. This was reflected in very low prices for quota. After the announcement and implementation of additional enforcement, the price of the right to harvest a kilo of sole rose from fl. 17 to approximately fl. 100. Observations prior to this announcement have therefore been ignored. The development of quota prices over time clearly indicate that there may be considerable differences between de facto and the jure fisheries management.

[9]We argue that the assumptions of constant prices and costs are not very restrictive. First, since we focus on modelling the effects for the Netherlands, the assumption of constant prices is relatively harmless, as trade in fish products among member states will reduce price effects. Second, with respect to constant harvest costs, we refer to Davidse and Beijert (1995), where it is reported that fixed costs, on average, account for more than 75% of total costs.

As a sensitivity analysis we have varied the values for P_f and λ, in accordance with reasonable values as indicated in the literature (Peck 1975, Robison and Barry 1987).

Table 9.1: Possible "utility neutral" reduction in fish quota under various conditions. $P_f = \gamma P$.

	$\lambda = 0.01$		$\lambda = 0.025$		$\lambda = 0.05$	
γ	1	1.1	1	1.1	1	1.1
Q	100	100	100	100	100	100
h^*	51	44.1	51	48.3	51	49.6
$\Delta var(\pi)$	23817	23333	23817	23730	23817	23788
ΔQ	11.3	11.5	34.5	34.5	73.0	73.0

We have assumed that a fisher initially has the right to catch 100 units of fish. If, for example, this fisher has a degree of risk aversion equal to 0.01, it is optimal for this fisher to hedge 51% of his fishing rights on the futures market. This would reduce the variance of revenues with 23817. This makes the fisher better off, which enables the policy maker to cut his quota by more than 11%. Given his hedged quantity of 51 units, this cut will take the fisher back to his original utility level. We conclude that the possible reductions in quota are substantial. For modest measures of risk aversion, significant utility-neutral reductions of fishing rights are possible, when those reductions are accompanied by the implementation of a futures market for fishing rights.

Three caveats should be mentioned. First, the analysis is based on an economic analysis, where quota handed out by the government or rights purchased in previous periods represent an opportunity cost for the owner (see also Hannesson 1996). However, if new policy is aimed at being truly "utility-neutral", the correct basis for the analysis might not be economic accounting but the perception of fishermen. It may be so that fishermen do not consider the fishing rights in their possession as costs. If fishers exclusively consider the costs associated with obtaining new quota, then possible quota cuts would be much more modest. For instance, under the assumption that the share of total quota that changes hands annually (either on the sellers or leasing market) is 15%, and these are the only rights considered as costs by the fishers, the possible reduction in quota size for the parameter values as specified in Table 9.1 ranges from 0% to 5%.

Second, even with modest measures of risk aversion as applied in Table 1, we may obtain negative values for π_{ce}.[10] One possible explanation, consistent with findings in agricultural economics, is that in addition to revenues, fishermen derive utility from the fishing process itself. In other words, fishing may be considered a way of life, yielding benefits outside the scope of simple economic accounting (for example, see Feeney *et al.* (1996) for an elaborate treatment of motivations of fishermen). However, when this additional utility is approximately constant, i.e. not affected by the size of the quota, the results displayed in Table 9.1 are still valid. For relatively large reductions this assumption is probably violated. If utility associated with the fishing process is increasing in the size of the quota allocated, Table 9.1 overestimates the true utility-neutral reductions.

Finally, the analysis is based on the assumption that fishermen are myopic in the sense that they do not recognize that hedging influences the quota cut that is inflicted upon them later: they first determine h^*, taking Q as given, and are next surprised by the government's decision to cut their quotas. There may be reasons for myopic behaviour. For instance, quota cuts will typically be based on hedging performance of the sector as a whole. This implies that optimal hedging for a rational fisherman requires developing expectations of (optimal) hedging for the other fishermen in the sector, which may be costly and cumbersome. In addition, it should be recognized that there is always uncertainty associated with the quota size, even in the absence of considerations brought forward in this paper. Depending on scientific advice on stock size *in situ*, the size of annual quota fluctuates. It may be difficult to estimate the "extra" uncertainty associated with the new policy. It is possible,

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[10] Without futures trading and with a very low degree of risk aversion ($\lambda = 0.01$), we find a positive value: $\pi_{ce} = E(\pi) - 0.01\ var(\pi) = 682$. But for higher levels of risk aversion, such as $\lambda = 0.025$, we find $\pi_{ce} = E(\pi) - 0.025\ var(\pi) = -333$. Note that negative values for π_{ce} do not imply that fishermen earn a negative income.

however, to imagine what the effect is of allowing a sequence of rounds of hedging and quota cutbacks, until an equilibrium is reached. Suppose that it is optimal for a fisherman to hedge 51 units of his initial quota of 100 (see Table 9.1), and that the government calculates that utility neutrality is restored when the quotum is reduced to 89. In the next period, the fisherman is able to increase his utility by hedging less than 51 units. For example, assume that, given Q equals 89, it is optimal to hedge 45 units. In order to restore utility neutrality, the managing agency should cut the quota further. This implies that Q should be reduced below 89, although the second cut will be more modest that the first one. Again, the fisherman will respond by hedging less to increase his utility, which will provoke further quota cuts by the government, until eventually an equilibrium is reached. (It is important to recognise that strategic actions of fishermen are completely disregarded in this reasoning.) For that reason, rational expectations will cause larger quota cuts than the ones predicted in Table 9.1. Formally incorporating rational expectations and strategic actions in the current model will be one of the possible extensions for future research.

9.4 Conclusions

Despite the implementation of exclusive fishing zones that formally ended the era of open access fisheries management, there is still evidence that current fishing effort is excessive. However, combatting overfishing is likely to result in resistance from the sector. Public choice theories predict that a small, well-organized group as the fishery sector may develop sufficient political pressure that it is able to resist painful reorganizing. This implies that an important question for policy makers is how to reduce fishing effort without "hurting" the sector.

In this paper we have analyzed the possibility of cutting quotas combined with implementation of a futures market for fishing rights. A futures market for fishing rights reduces price risk for fishermen, and thereby makes risk averse fishermen better off. Using the EV model we find that considerable quota cuts are possible to bring the fisher back to his original utility level, even with modest measures of risk aversion. The reason is that the price of fishing rights have fluctuated widely in the past, hence there is ample possibility for improvement. The results have been obtained under the conservative assumptions that (i) fish prices do not increase after the reduction in quota size, and (ii) total costs of fishing effort do not fall after the reduction in quota size. If one of these assumptions, or both, is violated, quota size could be reduced even more. Some caveats of the model are recognized and discussed.

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CHAPTER 10

UNCERTAINTY AND FOREST LAND USE: VAGUE PREFERENCES AND IMPRECISE COEFFICIENTS[1]

Abstract

Recently, increasing weight has been placed on non-timber values in forest management. Both the multiple objectives and the parameters that support decision making in forestry are often imprecise and vague. In this chapter, the concepts of fuzzy set theory are explained and then applied to the problem of allocating public forest land on Vancouver Island among competing land uses. Two principal sources of fuzziness are identified—those related to uncertainty in classification (specification of management objectives) and those related to uncertainty concerning how actions affect objectives (imprecise technical coefficients). By comparing the results of classical and fuzzy decision models, we conclude that the latter approach can be judged an improvement over the former. The fuzzy land-use allocation appears to be more consistent with the political decision making process, which relies on consultation and consensus seeking among various interest groups, that has evolved in British Columbia. The analysis also yields insights into the robustness of outcomes and suggests priority areas for further research.

10.1 Introduction

The Government of British Columbia in Canada owns more than 95 percent of the forest land in the province. In the past, these lands had been managed primarily for production of commercial timber from mature stands. During the late 1970s and 1980s, re-planting of denuded, not sufficiently restocked forest lands took precedence over other environmental concerns, but

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[1]This chapter is based on: Ells A., E.H. Bulte and G.C van Kooten, Uncertainty and Forest Land Use in British Columbia: Vague Priorities and Imprecise Coefficients, *Forest Science*, In Press.

during the early 1990s the Government introduced several initiatives to address additional environmental objectives. These initiatives included a Protected Areas Strategy (PAS), Timber Supply Reviews, the Commission on Resources and Environment (CORE) process, a Forest Practices Code, a Forest Renewal Plan, and a Forest Land Reserve (see van Kooten 1995). The policies reflect growing public recognition that forests are more than a source of industrial output. While the importance of commercial timber production to the provincial economy is not in doubt, managing land for multiple uses requires trading off different objectives in forest land management. In this chapter, we examine a particular aspect of land management, namely that of allocating forest land under uncertainty using a zoning instrument. To address uncertainty, we employ fuzzy set theory.

Allocation of forest lands, including protected areas, has defaulted to the CORE, which is charged with finding consensus among various stakeholder groups. The philosophy that emerged during the CORE process was to treat each of the multiple objectives of land use as equally important, so that economic efficiency in the sense of maximizing discounted welfare was not necessarily given preeminent status. Under this condition, traditional multiple objective decision making (MODM) is the most appropriate tool of analysis. The usefulness of classical MODM models is limited, however, because of the following characteristics inherent in the land-use decision making process:

- the objectives of society are ill-defined;
- the values that society attaches to various forest activities (such as recreation or preservation of biodiversity) are imprecise at best, or simply unknown;
- the effects of silviculture and other forest management decisions are uncertain, both from a biological and socioeconomic perspective;
- land-use and silvicultural decisions often pertain to an uncertain and distant future; and
- there is uncertainty about forest tenures, the macro economy, future product prices, and the ability of, or need for, governments to reduce deficits/debts.

In this chapter, we address uncertainty. Uncertainty has to do with the degree of belief or faith in the validity of a particular proposition or datum (Kruse *et al.* 1991). It arises from many sources, including measurement error, lack of judgement, imprecision, unreliability, variability, vagueness, ignorance and ambiguity. The theory of fuzzy sets is used in this study to deal with vagueness. Vagueness is said to occur when an object is known completely but its classification is in doubt because the set to which it may belong is poorly defined (Barret and Pattanik 1989); vagueness refers to the lack of clear-cut boundaries for the set of objects to which the symbol or meaning is applied (Fedrizzi 1987). Or, in the words of Munda, Nijkamp and Rietveld (1992), "fuzzy uncertainty does not concern the occurrence of an event but the event itself, in the sense that it can not be described unambiguously". In addition to this interpretation (which will be elaborated upon in section 10.2), we will use the theory of fuzzy sets to deal with imprecision in parameter values. While there is not necessarily something "vague" or "ambiguous" about these parameters or their interpretation, fuzzy set theory is employed because there is not sufficient support for a proper probability distribution.

Applications of fuzzy set theory in the field of forestry and land-use planning are expanding. Mendoza and Sprouse (1989) proposed a two-stage approach to forest planning, and developed a fuzzy model for more flexible and robust generation of alternatives. The uncertainty in their model arose from imprecise coefficients and was modeled by tolerating some constraint violations. Hof *et al.* (1986), Pickens and Hof (1991) and Bare and Mendoza (1992) compared classical (i.e., crisp) and fuzzy models for describing optimal harvest over time. They found that, by relaxing the constraint of non-declining harvest volume over time, net present value (NPV) could be significantly increased (see also Hof 1993, pp.103-14). Mendoza *et al.* (1993) developed a fuzzy multiple objective linear programming model for forest planning that accommodated uncertainty in the objective function by making coefficients interval-valued. Finally, Tecle *et al.* (1994) developed an interactive fuzzy multi-criterion decision model in which the decision maker is allowed to search the frontier of efficient solutions instead of being confronted with a uniquely preferred solution. Fuzzy set theory was used to deal with a vague objective and constraint.

This chapter differs from previous ones because its scope is broader. In Bare and Mendoza (1992) and Pickens and Hof (1991), for instance, the focus is on timber yield only. We have a social orientation in which timber is but one of many services provided by forests.[2] Obviously, extending the analysis to allow for recreation and preservation benefits requires data that are less precise comparatively than similar data for timber, because they are unobservable in markets and are difficult to measure. While Bare and Mendoza (1992) argue that imprecise (timber) coefficients are better viewed as stochastic rather than fuzzy, in this paper we address imprecision in coefficients as fuzzy measures.

The broader scope also implies that the impact of fuzzifying a crisp model is different. If fuzzifying simply amounts to relaxing a binding constraint of a NPV maximization model, then the predictable result is that NPV will increase! Relaxing a binding constraint will always have this effect. In our analysis, fuzziness actually affects the allocation of land among uses; this is a less than trivial result of the fuzzy approach.

Finally, this is the first chapter in the field that actually combines the existence of vague objectives and constraints, as in Tecle *et al.* (1994) and Pickens and Hof (1991), <u>and</u> imprecise coefficients, as in Mendoza and Sprouse (1989), Bare and Mendoza (1992), Hof *et al.* (1995) and Mendoza *et al.* (1993). Moreover, imprecise coefficients are dealt with explicitly by modelling them as fuzzy numbers instead of incorporating them implicitly in the analysis as admissible violations of constraints, as in Mendoza *et al.* (1993). The approach taken in this chapter gives more insight into the effect of the various sources of uncertainty on land allocation and makes better use of available information.

The purpose of the current research is to capture the uncertainty inherent in describing the socioeconomic impacts of land-use and forest management decisions on Vancouver Island. The major objectives of this study are threefold: (1) to develop a fuzzy multiple objective decision-making model that incorporates uncertainties both in objective specification and parameter values; (2) to contrast the fuzzy models to a classical multiple objective approach where uncertainty is not considered; and (3) to demonstrate the usefulness of fuzzy set theory in the context of a multiple objective decision-making model for land use on Vancouver Island. Vancouver Island

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[2]The papers by Mendoza *et al.* (1993) and Tecle *et al.* (1994) deal with multiple objectives, but are based on NPV maximization models.

was selected because it is a region where land-use conflicts are intense and the recent CORE (1994) land-use recommendations (subsequently adopted by government) have been controversial.

The rest of the chapter is organized as follows. In section 10.2, we provide a brief, formal description of fuzzy logic, followed in section 10.3 by the development of three fuzzy decision support models. The derivation of the required parameters for the models are presented in section 10.4, while the empirical results are provided in section 10.5. Our conclusions ensue.

10.2 Uncertainty and Fuzzy Set Theory

Current literature concerned with modelling uncertainty provides a wide range of definitions both for the concept of uncertainty itself as well as the various types of uncertainty that may be addressed (see, e.g., Krause and Clark 1993; Kruse *et al.* 1991). In this regard, fuzzy set theory has become a particularly fruitful line of research for dealing with vagueness and ambiguity (e.g., Cox 1994; Zimmerman 1991; Barret and Pattanik 1989; Fedrizzi 1987). Dubois and Prade (1993) and Kosko (1992, pp.264-68) have shown that there is a clear distinction between probability theory and fuzzy set theory—that the fuzzy approach to uncertainty is valid in its own right, and that it is different from the Bayesian approach of probability theory. Kosko (1992) provides a formal proof that Bayes theorem is simply a special case of the Subsethood theorem of fuzzy sets. The proof is based on the fact that in probability theory an element is either in that set or not in that set (see below), while in fuzzy set theory it can be an element of both a set and its complement, each with some degree of membership.

In this section, we provide a formal treatment of fuzzy logic by considering membership or indicator functions for fuzzy sets (objective targets) and fuzzy numbers for imprecise values of the technical coefficients in the decision model. This background constitutes the formal foundation for the fuzzy MODMs, without and with imprecise coefficients, that are developed in the next section.

Fuzzy sets and membership functions

An element x of the universal set X is assigned to an ordinary (crisp) set A via the characteristic function μ_A, such that:

$$\mu_A(x) = 1 \qquad \text{if } x \in A \subset X. \tag{1}$$
$$\mu_A(x) = 0 \qquad \text{otherwise.}$$

The element has either full membership ($\mu_A(x) = 1$) or no membership ($\mu_A(x) = 0$) in the set A. The valuation set for the function is the pair of points {0,1}. A fuzzy set $\tilde{A}$ is also described by a characteristic function, the difference being that the function now maps over the closed interval [0,1].[3]

Formally, a fuzzy set $\tilde{A}$ of the universal set X is defined by its membership function

$$\mu_{\tilde{A}} : X \rightarrow [0,1], \tag{2}$$

which assigns to each element $x \in A$ a real number $\mu_{\tilde{A}}(x)$ in the interval [0,1], where the value of $\mu_{\tilde{A}}$ at x represents the grade or degree of membership of x in $\tilde{A}$ (Sakawa 1993). While membership functions can take on a variety of functional forms, linear specifications are often employed.

An example of fuzzy membership is the set of "natural forests," where it is clear that old-growth forests belong to this set with a degree of membership equal to 1. As we consider progressively heavier logged forests, the descriptor "natural" becomes less apt. Partly logged forests are assigned a partial degree of membership in the set "natural forests," something less than one. This is an example of a one-sided fuzzy set, with membership in the set approaching zero as the exploitation pressure increases. A two-sided fuzzy set might be the set of "ponds". A "pond" ceases to be one when it becomes so large that it is better conceived of as a "lake", or when it becomes so small that it is better thought of as a "puddle." It is the researcher's task in these

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[3]The characteristic function should not be confused with probability. Probability deals with the quantification of an uncertain event, while fuzzy set theory deals with the quantification of the uncertainty of the description of the event (see, e.g., Kosko 1992).

cases to construct the relevant fuzzy sets for the sets "natural forests" or "ponds" by specifying threshold values and membership functions, as will be explained below.

The preceding definitions of membership employed the concept of a normalized fuzzy set. A fuzzy set $\tilde{A}$, defined over a finite interval, is said to be normal if there exists an $x \in X$ such that $\mu_{\tilde{A}}(x) = 1$, and $\mu_{\tilde{A}}(x) \leq 1 \ \forall \ x \in X$. A subnormal fuzzy set is normalized by dividing $\mu_{\tilde{A}}(x)$ by its height or greatest membership value.

Set theoretic operations are defined for fuzzy sets. Among these are the concepts of containment, complement, intersection and union. Fuzzy set $\tilde{A}$ is a subset of (contained in) $\tilde{B}$ if and only if the membership function of $\tilde{A}$ is less than or equal to that of $\tilde{B}$ everywhere on X:

$$\tilde{A} \subseteq \tilde{B} \quad \Leftrightarrow \quad \mu_{\tilde{A}}(x) \leq \mu_{\tilde{B}}(x) \text{ for all } x \in X. \tag{3}$$

The complement of $\tilde{A}$ (written as $\bar{A}$) is defined as:

$$\mu_{\bar{A}}(x) = 1 - \mu_{\tilde{A}}(x) \tag{4}$$

The intersection of the fuzzy sets $\tilde{A}$ and $\tilde{B}$ is defined as:

$$\tilde{A} \cap \tilde{B} \quad \Leftrightarrow \quad \mu_{(\tilde{A} \cap \tilde{B})} = \min\{\mu_{\tilde{A}}(x), \mu_{\tilde{B}}(x)\} \text{ for all } x \in X, \tag{5}$$

and the union as:

$$\tilde{A} \cup \tilde{B} \Leftrightarrow \mu_{(\tilde{A} \cup \tilde{B})} = \max\{\mu_{\tilde{A}}(x), \mu_{\tilde{B}}(x)\} \text{ for all } x \in X. \tag{6}$$

Hence, the intersection $\tilde{A} \cap \tilde{B}$ is the largest fuzzy set contained in both $\tilde{A}$ and $\tilde{B}$, and the union $\tilde{A} \cup \tilde{B}$ is the smallest fuzzy set containing both $\tilde{A}$ and $\tilde{B}$.

While both union and intersection of fuzzy sets are commutative, associate and distributive, as is the case for ordinary or crisp sets, fuzzy logic deviates from crisp logic because, if we do not know $\tilde{A}$ with certainty, then its complement $\bar{A}$ is also not known with certainty. Thus, $\bar{A} \cap \tilde{A}$ does not produce the null set as is the case for crisp sets (where $A^c \cap A = \phi$). Thus, fuzzy logic violates the "law of noncontradiction". It also violates the "law of the excluded middle" because the union of a fuzzy set and its complement does not equal the universe of discourse—the universal set X. Thus, $\tilde{A}$ is properly fuzzy iff $\tilde{A} \cap \bar{A} \neq \phi$ and $\tilde{A} \cup \bar{A} \neq X$ (Kosko 1992, pp.269-272).

Fuzzy numbers and alpha cuts

A fuzzy number describes the situation where a parameter value is "approximately m" or "about n." Fuzzy numbers are approximations of a central value and can be represented by "bell" curves, triangular functions, trapezoids, and so on (Cox 1994). A standard form of fuzzy number that allows for computational efficiency is that of the L-R (left-right) fuzzy number. A fuzzy L-R number M is fully characterized by three parameters—m is the mean value of M and σ and β are the left and right spreads, respectively. It is defined as:

$$\mu_M(x) = \begin{cases} L\left(\frac{m-x}{\sigma}\right) & x \leq m \quad \sigma > 0 \\ R\left(\frac{x-m}{\beta}\right) & x \geq m \quad \beta > 0 \end{cases} \tag{7}$$

and written as $(m,\sigma,\beta)_{LR}$. Operations for fuzzy numbers of the L-R type have been provided by Sakawa (1993, pp.26-30). Given the fuzzy numbers $M=(m,\sigma,\beta)_{LR}$ and $N=(n,\gamma,\delta)_{LR}$, the basic L-R fuzzy operators for symmetric fuzzy numbers (where $\beta=\sigma$ and $\gamma=\delta$), are as follows:

Addition:	$(m,\beta)_{Sym} \oplus (n,\gamma)_{Sym} = (m+n,\ \beta+\gamma)_{Sym}$	(8)
Subtraction:	$(m,\beta)_{Sym} \ominus (n,\gamma)_{Sym} = (m-n,\ \beta+\gamma)_{Sym}$	(9)
Multiplication:	$(m,\beta)_{Sym} \otimes (n,\gamma)_{Sym} = (mn,\ n\beta+m\gamma)_{Sym}$, iff $m,n > 0$	(10)
Scalar multipl.:	$k \otimes (m,\beta)_{Sym} = (km,k\beta)_{Sym}$	(11)

The fuzzy number $M=(m,\beta)_{Sym}$ resembles a membership function in appearance and is used to describe a continuous symmetrical quantity distribution about an imprecise parameter. While m is the central value of the fuzzy number M, the width of the spread defined for the set is an indication of the reliability of the estimate. Typically, narrow spreads correspond with "reliable" values for the imprecise parameter.

Although the class of fuzzy numbers discussed above assume symmetry, this does not presuppose symmetry in solution sets. The response in any one parameter value to a change in the fuzzy quantity is strictly a function of the spread (β_{ij}) defined for that number. The spread completely defines the slope of the linear possibility distribution and thus the rate of change in value. It is the net result of all such independent movements that determines the ultimate solution.

Another concept required for model building with fuzzy sets is that of the α-level set. The α-level set A_α is simply that subset of A for which the degree of membership exceeds the level α, and is itself a crisp set (an element either meets the required level of α or it does not).

$$A_\alpha = \{ x \mid \mu_{\tilde{A}}(x) \geq \alpha\},\ \alpha \in [0,1]. \tag{12}$$

A_α is an upper level set of $\tilde{A}$. The use of α-level sets provides a means of transferring information from a fuzzy set into a crisp form. Defining an α-level set is referred to as taking an α-cut, cutting off that portion of the fuzzy set whose members do not have the required membership. Taking different α-cuts allows the decision maker to consider different "realizations" of the problem, based on how much uncertainty he wishes to consider.[4]

A fuzzy multiple objective decision making model with vague preferences and imprecise coefficients can be formulated as a crisp linear program (LP), as illustrated in this section. We acknowledge that classical formulations exist that closely resemble the fuzzy model developed here—the class of "minimum distance models" bears some resemblance. However, the set-up of the problem would be different, and so is its conceptualization in the context of uncertainty.

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[4]If an α-cut is taken for several coefficients individually, then the overall output uncertainty associated with this α-cut is unknown. If the decision maker is more concerned with total output uncertainty than with uncertainty levels in individual model coefficients, this may be an important omission. We thank John Hof for pointing this out to us.

Fuzzy multiple objective linear programming

In the fuzzy MODM model, we are concerned with uncertainty surrounding the definition of satisfactory solution values for each of the objective functions. Although a precise value for each objective is provided by the model, it is unclear as to how well that value represents the concept of a fully satisfied objective. The term satisfactory solution is defined vaguely; it is a fuzzy set. Thus, a goal $\tilde{G}(x)$ or constraint $\tilde{C}(x)$ may be completely satisfied by choice of the solution vector x ($\mu_{\tilde{G}}(x)=1$ or $\mu_{\tilde{C}}(x)=1$), completely unsatisfied ($\mu_{\tilde{G}}(x)=0$ or $\mu_{\tilde{C}}(x)=0$), or somewhat satisfied ($0<\mu_{\tilde{G}}(x)$, $\mu_{\tilde{C}}(x) <1$). Crisp goals and constraints are accommodated in this framework by defining a crisp set as a specialized case of a fuzzy set.

The decision space, $\mu_{\tilde{D}}$, is the fuzzy set defined by the intersection of the fuzzy goal and the fuzzy constraint, and is characterized as

$$\mu_{\tilde{D}}(x) = \text{min.} \ (\mu_{\tilde{G}}(x),\ \mu_{\tilde{C}}(x)). \tag{13}$$

The decision space is illustrated in Figure 10.1.

It follows that, in order to maximize the minimum degree of satisfaction of the goals and constraints, the objective function for the fuzzy linear programming model is:

$$\text{max.}\ \mu_{\tilde{D}}(x) = \text{max.min.}\ (\mu_{\tilde{G}}(x),\ \mu_{\tilde{C}}(x)),\ \ x \in X \tag{14}$$

This maxmin operator is but one of several ways to represent the decision. In the absence of evidence for ranking operators, maxmin was chosen because it is simple and linear, but it may fail to capture the true decision-making process. The weights are implicitly determined by the parameters of the membership functions (because the goals are incommensurable, it is not immediately clear what set of membership functions would imply equal weights). The solution is obtained where the minimum membership value has been maximized, or the lowest level of satisfaction has been raised as high as possible. This decision mimics the situation where there are multiple interest groups involved on Vancouver Island, and the strength and viability of the

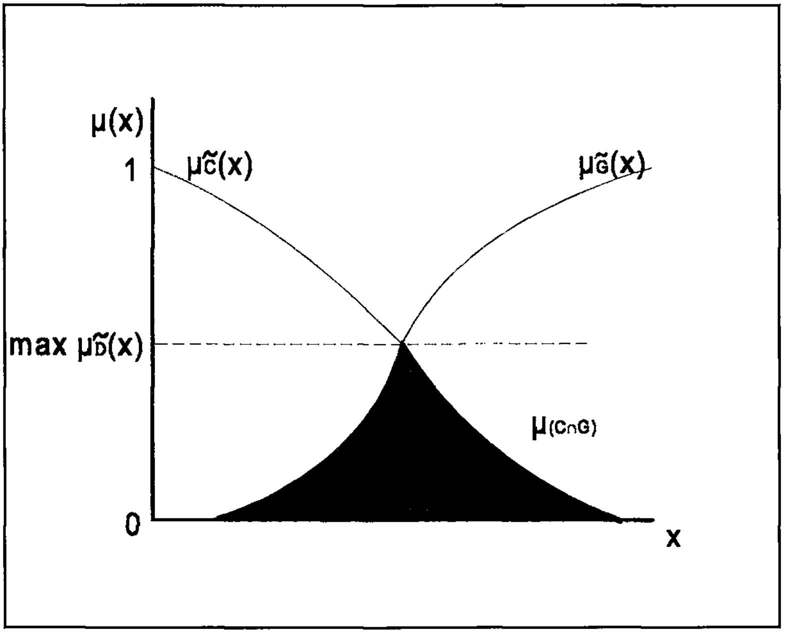

Fig. 10.1: Illustration of a fuzzy decision.

consensus solution will be largely influenced by the level of commitment (satisfaction) of the least committed (satisfied) group.

Model construction places implicit values on outputs, because it is impossible to make a decision or rank alternative scenarios unless such values are assigned. By employing the fuzzy approach, as represented by maxmin, it is possible to avoid translating the objectives into common, perhaps unnatural or inappropriate, units of measurement as doing so can derail a multi-group negotiation problem. Through iteration and continued revision of the membership functions, one would expect to arrive at some set of implied values, although not necessarily measured in dollar terms. Since iteration has not been possible in the case of the Vancouver Island decision, this study can merely suggest a framework for arriving at group consensus or decision. Therefore, despite its limitations and in the absence of good evidence to argue for another operator, the maxmin approach is used here; it is also favoured in the literature (e.g., Hof, Pickens and Bartlett 1986; Hof 1993).

The decision model can now be written as a crisp LP. Suppose that the original fuzzy MODM is as follows:

$$\begin{array}{lll} \text{find} & x & \\ \text{s.t.} & A_i x \tilde{\geq} b_i & i = 1,2,\ldots,N \\ & x \geq 0, & \end{array} \tag{15}$$

where N is the number of goals plus constraints, A_i refers to the crisp parameter values, and $\tilde{\geq}$ refers to fuzzy objective or constraint sets. Assuming linear membership functions, Zimmerman (1991) has shown that (15) can be written as:

$$\begin{array}{lll} \text{Max} & \lambda & \\ \text{s.t.} & A_i\, x - b_i - d_i(\lambda - 1) \geq 0, & i = 1, 2,\ldots,N \\ & \lambda \in [0,1], \text{ and} & \\ & x \geq 0. & \end{array} \tag{16}$$

where $\lambda = \mu_{\tilde{D}}(x)$ and d_i is the spread of a one-sided fuzzy set—for fuzzy set $\tilde{Z}$ this implies that $\mu_{\tilde{Z}}(x)=0$ for $x \leq b_i - d_i$ and $\mu_{\tilde{Z}}(x)=1$ for $x \geq b_i$. This is the maxmin formulation of the fuzzy MODM. Other formulations exist, some of which employ a composite objective function, which is usually additive (see Mendoza and Sprouse 1989).

<u>Fuzzy programming with imprecise coefficients</u>

Now consider the situation where the elements of the matrix A are not precisely known. The j-th element of $\tilde{A}_i$, $\tilde{a}_{ij}$, is described by a fuzzy number. Furthermore, assume that these fuzzy numbers are triangular and symmetric, allowing $\tilde{a}_{ij}$ to be written as (m_{ij}, β_{ij}) with degree of membership:

$$\begin{array}{ll} \mu(a_{ij}) = 0 & ;\ a_{ij} \leq m_{ij} - \beta_{ij} \\ \mu(a_{ij}) = 1 + (a_{ij} - m_{ij})/\beta_{ij} & ;\ m_{ij} - \beta_{ij} < a_{ij} < m_{ij} \\ \mu(a_{ij}) = 1 & ;\ a_{ij} = m_{ij} \\ \mu(a_{ij}) = 1 - (m_{ij} - a_{ij})/\beta_{ij} & ;\ m_{ij} < a_{ij} < m_{ij} + \beta_{ij} \\ \mu(a_{ij}) = 0 & ;\ m_{ij} + \beta_{ij} \leq a_{ij}. \end{array} \tag{17}$$

This fuzzy number is depicted in Figure 10.2.

To capture the effect of uncertainty in the model parameters, we employ an α-cut. This allows the definition of a crisp parameter value derived from the characteristics of the underlying fuzzy set, and permits the use of a standard LP format. From Figure 10.2, an α-cut gives two possible realizations of a_{ij} (a_1 and a_2), which represent, respectively, the lower and upper bounds of the parameter. Consider the lower bound first. The imprecise nature of the technical coefficients is incorporated into model (16) to give the following structure (Lai and Hwang 1994):

$$\begin{aligned} &\text{Max} \quad \underset{\sim}{\lambda} \\ &\text{s.t.} \quad [A_i - (1-\alpha)\beta_i]x - b_i - d_i(\lambda - 1) \geq 0, \quad i = 1,2,\ldots,N \qquad (18) \\ &\qquad \lambda \in [0,1],\ \alpha \in [0,1] \text{ and} \\ &\qquad x \geq 0. \end{aligned}$$

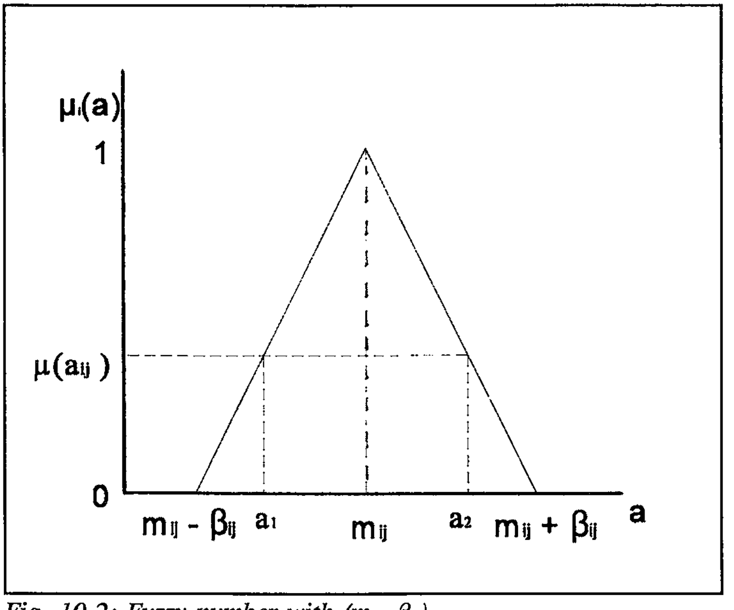

Fig. 10.2: Fuzzy number with (m_{ij}, β_{ij}).

Model (18) permits each element in the parameter matrix to be adjusted to reflect the membership level being considered. Each element $\tilde{a}_{ij}$ is transformed so that $\mu(\tilde{a}_{ij})=\alpha$, the degree of membership of $\tilde{a}_{ij}$ is α. When α

= 1, this MODM formulation is identical to the fuzzy MODM discussed previously. As α decreases, parameter values move away from the centre value, m_{ij}, to values lying below m_{ij} on the defined interval. The solution is now derived using parameter values with a lower degree of membership. This case reflects the situation where the parameters considered most likely are greater than the true parameter values. The model in (18) sets all imprecise parameters to the same degree of membership, one distinct point in the range of possible solutions. The justification for doing so was discussed above.

If we replace $\tilde{A}_i$ - $(1-\alpha)\beta_i$ in (18) by $\tilde{A}_i + (1-\alpha)\beta_i$, then the model uses higher values for the parameters that have the same degree of membership. This represents the situation where the most probable values lie below the true values.

Jointly, these two models provide an upper and lower bound for possible solutions by considering the two extreme points of the fuzzy numbers defined by $\mu(\tilde{a}_{ij})=\alpha$. As the decision maker is less confident that the central value of the fuzzy number is a correct representation of the true value, the α-cut is lowered (this can be visualized by lowering the horizontal line in Figure 2) and the length of the interval separating these bounds increases. This interval identifies the range of possible solutions. Model (18) is restrictive in that it considers only the endpoints of this interval, the extreme cases. By comparing the change to the solution with respect to a change in the α-cut, one can perform a type of sensitivity analysis with respect to uncertainty in the definition of parameters.

The parameters λ and α have a special interpretation. Parameter λ is the degree to which the least satisfied objective has been met. Given the models (16) and (18), and considering a degree of uncertainty specified by α, the least of the objectives is satisfied to degree λ. Parameters α and λ are related in that the level of uncertainty helps determine the solution, but they are not similar concepts; λ deals with satisfaction of objectives—they are vague and remain so in any outcome—while α addresses the problem of imprecise coefficients and allows one to control the level of uncertainty to be considered in a particular trial. If one abandons the strict requirement that imprecise coefficients are described by symmetric, triangular functional forms, it is possible that $\alpha=1$ yields a solution that is itself a fuzzy number (e.g., in the case of trapezoidal functional forms) rather than crisp as in our case.

The maxmin approach developed above will have a solution in which it will be impossible to increase the membership value of one objective without reducing the membership value of another. In this sense, the resulting solution is somehow efficient. The model is now applied to land-use allocation on Vancouver Island.

10.3 A Decision Model for Land Use on Vancouver Island

Vancouver Island consists of nearly 3.35 million hectares (ha), of which 2.4 million ha is publicly owned and has been classified according to timber production potential. During deliberations, the Vancouver Island CORE employed the land use categories "high-intensity resource use", "integrated resource use", "low-intensity resource use", "protected areas" and "settlement" (van Kooten 1995). As public lands are the focus of this analysis, "settlement" lands and other private lands are ignored.

Goals reflecting the general public's expectations regarding forest land use in B.C. are taken from the 1989 Parksville Old-Growth Workshop (B.C. Ministry of Forests 1990). They are as follows: (1) achieve a high revenue from timber harvest; (2) create additional benefits from forest recreation activities; (3) obtain the greatest possible nonuse benefits from forests, as measured in monetary terms; (4) maintain forest employment; (5) collect substantial direct revenues from the forest industry; (6) achieve a high contribution of the forest sector to provincial Gross Domestic Product (GDP); and (7) expand wilderness protection.[5] Vague terminology renders each of these objectives fuzzy, and therefore the values for each cannot be known precisely. As discussed earlier, fuzziness is a measure of how well an instance or value conforms to a semantic ideal or concept. Hence, vagueness can be

[5]An additional objective mentioned was maximizing long-run sustained yield (LRSY). However, since this does not seem a worthy objective in itself (rather it supports the 6 objectives listed in the text), it is not included in the analysis.

modeled through the specification of (one-sided) fuzzy objectives, while imprecision about parameter values can be modeled using fuzzy numbers.

Three types of MODM models for land-use decisions on Vancouver Island are compared to evaluate the usefulness of fuzzy MODM. The first is a crisp NPV maximizing formulation of the multiple goal problem.[6] The second is a fuzzy multiple objective decision model that incorporates the fuzziness of objective values. Finally, a fuzzy multiple objective decision model with both fuzzy objectives and imprecise parameters is considered. The models are static and assume a planning period of 100 years—the assumed rotation age of the working forest. The first step in the modelling process is specification of the parameters.

Description of imprecise parameters

Logging benefits

Logging benefits per hectare are calculated as the difference between the price and the cost of a cubic meter (m^3) of delivered wood. Harvest volume is assumed to be a function of two harvest site attributes: site quality and management intensity.

Site quality is characterized as good, medium or poor. Average harvest volumes by stand age, species and site class, for the B.C. coastal region, are taken from the FOREST6.0 model (Phelps *et al.* 1990a, 1990b). Uncertainty as to the realized harvest from a particular site is captured by specifying a range of possible harvest volumes based on consideration of extremes provided by species composition and consideration of a 20-year spread in harvest age.

In this chapter, an area is assigned to the "high intensity management" category if intensive silviculture (spacing, pruning and pre-commercial thinning) is to be practised. Under "integrated management", it is assumed that basic silviculture (site preparation and replanting) will be performed.

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[6]This approach is standard practice in forest economics to cope with multiple objectives Maximization of social welfare or NPV is achieved by summing the various accounts (e.g., Tecle *et al.* 1994; Mendoza *et al.* 1993).

Land allocated to "low intensity" management provides harvest volume from naturally regenerated stock. No harvest is available from "protected" areas.

Harvest volumes available from each of these nine land allocation categories (3 site qualities and 3 management categories) are described by symmetrical triangular fuzzy numbers of the form (m_{ij}, β_{ij}). The centre value m_{ij} is the arithmetic mean of the extreme values.

Ranges for wood prices, based on species, age and management, are also taken from the FOREST6.0 simulation model, and fuzzy numbers for the price parameters are calculated as for wood volume. These distributions are scaled to reflect the average 1992 wood price of \$70.71 /m^3 for the Coast region (see Price Waterhouse 1993).

Calculation of delivered wood costs follows the methodology outlined above, with the exception that costs vary with management intensity but are constant across site qualities. Two cost values are reported for the B.C. coastal region, one for low cost and another for high cost sites. The fuzzy numbers are based on the mean of the two cost figures and are scaled to reflect an average cost of delivered wood after stumpage fees, rents and royalties of \$65.13/m^3 (Price Waterhouse 1993). These costs do not include costs of silviculture. The Ministry of Forests (1992) provides average cost data for silvicultural activity in 1992. Basic silviculture was applied at a cost of \$21.20/ha, while incremental silviculture represented an added expense of \$20.00/ha. These costs are added in the appropriate management categories. Net logging benefits are calculated as the difference of total revenue per hectare and total costs per hectare. Using the definitions of fuzzy addition, subtraction and multiplication provided in equations (8)-(11), we obtain symmetric fuzzy numbers. The results are summarized in Table 10.1.[7]

Recreation values

Recreation benefits are identified as a goal of land use planning. Recreation plus recreation option value for the Vancouver forest region are estimated at \$111.11 million per year (B.C. Ministry of Forests 1991), for an

[7]The negative value associated with harvest of poor sites is consistent with the current situation: most of the current harvest is obtained from the better quality sites, and there is no margin to allow harvest of the inferior site class.

average recreational value of $33/ha/yr. This value was obtained under the current management, which is denoted as integrated management in this study. Land under low intensity management is assumed to offer little in increased recreational opportunities compared to integrated management, with the same recreational activities being pursued and logging ongoing. Land under intensive management is assumed to produce only 50% of the benefits attainable under integrated management as intensive forestry practices compromise recreational opportunities. Protected areas, with potentially more stringent guidelines as to appropriate recreational activities, will provide only 40% of the benefits received from the integrated management regime. Centres for the fuzzy numbers are scaled to preserve the gross average of $33/ha/yr, with distribution spreads arbitrarily set at $10/ha/yr for all classes. The results are summarized in Table 10.1.

Preservation values

Estimation of preservation or nonuse benefits are based on a survey conducted by Vold *et al.* (1994) that determined the values that B.C. residents place on wilderness protection in the province. The mean maximum annual willingness to pay for a doubling and tripling of wilderness area from a base of 5% were $136 and $168 per household, respectively. We assume that the number of households on Vancouver Island corresponds to the size of the labour force (one labourer per household). Each household on the Island is then prepared to pay $32/yr ($168 minus $132) to increase the amount of protected area to 495,000 ha (15% of the total) from the current 10% level, corresponding to an average annual payment of $26.69/ha of protected area.[8]

Clearly the value of nonuse attributes falls with increasing management intensity, but there is little information for quantifying this relationship. The assumption is that low intensity management provides preservation benefits at 50% of that of protected areas, integrated management areas at 25% the level of protected areas, and land under high intensity management is

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[8]Preservation values are underestimated if part of the benefits accrue to people off Vancouver Island. On the other hand, they may be over-estimated because the marginal value of nonuse attributes is assumed constant while the study by Vold *et al* (1994) indicates that these values may well be declining.

assumed to provide no nonuse benefits. Distribution spreads for these fuzzy numbers are set to allow the range of possible values to begin at 0 and extend to twice the hypothesized value. The results are summarized in Table 10.1.

Forest sector employment

Forest related employment may be generated both by the forest industry and the forest-related tourism and recreation industry. Price Waterhouse (1993) reports 1.18 jobs/1,000 m^3 of wood harvested for the coastal industry. This estimate is reduced slightly to 1.16 jobs per 1,000 m^3 to reflect the fact that some of the jobs associated with the Island harvest are located in mainland mills. The spread for this fuzzy number is set at 0.07, consistent with the variation reported by Statistics Canada (see COFI 1992) for the past decade.

There is little information about the relationship between employment and other uses of the forest. Regionally-based studies yield estimates of 0.0001 to 0.0003 jobs/ha (see Matas 1993; Clayton Resources Lt. and Robinson Consulting & Associates Ltd. undated). The latter figure is used to anchor the job fuzzy number with relatively large spreads to reflect the high degree of uncertainty regarding their genesis.

Direct and indirect government revenue

In 1992, the provincial government and municipalities received \$5.27/m^3 of harvest, while the province collected \$9.05/m^3 in stumpage fees for a total revenue of \$14.32/m^3 (Price Waterhouse 1993). It is assumed that revenues can vary by as much as \$5/m^3.[9]

Indirect revenues are examined by looking at the contribution of forestry to provincial GDP. Forestry accounts for a substantial proportion of provincial GDP, indicating a high dependence on forest operations. Each cubic meter of harvest contributes about \$70 to GDP. An interval of \$20/m^3 is chosen. The results are summarized in Table 10.1.

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[9] In this analysis, *direct* revenues to the provincial government do not include employee taxes paid as a result of indirect and induced employment, and revenues accruing to the federal government are ignored.

Expansion of Wilderness Protection

There is nothing uncertain about the contribution of a hectare of land towards the objective of wilderness protection. It is a crisp parameter—one hectare of land allocated to protected area provides one hectare of protected area.

Table 10.1: Fuzzy Parameter Values, Mean and Spread, in $C per Hectare (except employment in jobs per 1000 cubic meters)

Site Quality	Logging benefits	Recreation values	Preservation values	Employment	Direct G.Rev.	Indirect G. Rev.
High Intensity Management						
good	(197, 324)	(21.9, 10)	(0,0)	(0.0138, 0.0045)	(170, 150)	(833, 343)
medium	(31, 164)	(21.9, 10)	(0,0)	(0.0100, 0.0027)	(123, 69)	(603, 215)
poor	(-26, 104)	(21.9, 10)	(0,0)	(0.0058, 0.0029)	(71, 49)	(349, 167)
Integrated Use Management						
good	(62, 211)	(43.9, 10)	(6.7,6.7)	(0.0131, 0.0040)	(162, 96)	(792, 304)
medium	(9, 143)	(43.9, 10)	(6.7,6.7)	(0.0087, 0.0026)	(107, 63)	(525, 199)
poor	(-23, 75)	(43.9, 10)	(6.7,6.7)	(0.0044, 0.0023)	(54, 45)	(266, 163)
Low Intensity Management						
good	(61, 160)	(43.9, 10)	(13.4,13.4)	(0.0101, 0.0028)	(124, 71)	(608, 221)
medium	(19, 95)	(43.9, 10)	(13.4,13.4)	(0.0061, 0.0018)	(75, 44)	(368, 141)
poor	(-4, 53)	(43.9, 10)	(13.4,13.4)	(0.0033, 0.0015)	(41, 30)	(202, 104)
Protected	-	(26.3, 10)	(26.8,26.8)	-	-	-

<u>Objective target values</u>

The objectives [i.e., the b_i in model (15)] are all modeled as fuzzy "greater than constraints". Thus, the degree of satisfaction increases as the value of the objective function increases. The value deemed to be the lowest possible to generate any satisfaction of the objective defines the lower limit of the constraint interval (b_i-d_i). The value deemed to be the lowest value at which complete satisfaction of the objective is attained defines the upper limit of the interval (b_i). The degree of membership in the fuzzy objective set is given by a one-sided fuzzy number.

Two approaches can be used to determine the upper and lower values for the objectives. The levels may be provided by a decision-maker or an expert in the area, relying on a subjective understanding of both the limits inherent in the system as well as what would constitute a satisfactory level of achievement. In this paper, "employment" is incorporated using this approach. A second approach is to define the upper and lower bounds as the maximum and minimum levels that the system can provide when each objective is considered in isolation. This method may be especially suitable when objectives are less sensitive politically and not restricted to a narrow range *a priori*. It is an objective means to defining the fuzzy constraints and is appropriate when there is little information available regarding the problem, preventing initial specification of unrealistic objectives. In the current analysis, the objective "logging benefits" is incorporated using this approach.[10]

For logging benefits, the level for complete satisfaction is set as the maximum available from the model if only logging benefits are considered. The minimum represents the amount generated from a working forest of 700,000 hectares, even though such a scenario was rejected by CORE as too low. The lower and upper bounds for logging benefits are $48.1 million and $72.0 million, respectively. Recreation and preservation benefit intervals are defined by the maxima and minima available from the system. The lower and upper bounds for recreation are ($49.9 million, $90.1 million), and for preservation ($8.6 million, $60.0 million).

Employment is a politically sensitive issue. We assume that the current level provided by the forest industry is fully satisfactory, even though it will be difficult to maintain current employment in the future as technological developments lead to a decreasing number of jobs per unit of harvest.

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[10]In theory, it would be preferable to use the second method to set initial parameters, and then use feedback information from users to refine the objective intervals, thereby incorporating new information on values or preference structure. The specification of satisfactory levels of achievement for this model employs a combination of the two approaches, without the benefit of any interactive procedure.

Jobs related to recreation are also considered satisfactory at current levels, although, in actual fact, it would probably be less than satisfactory if current levels were simply maintained. However, recreation contributes only a very small number of jobs compared to those related to timber harvest; requiring an increase in this component has little impact in the model. The lower bound for job provision is set arbitrarily at 15% below the current level (i.e., 13,500), on the assumption that it would be politically unwise for government to allow employment levels to drop below this figure. The upper bound is 15,700 jobs.

Maximal values for both direct and indirect revenue are determined by the ability of the system to generate revenues and timber-related GDP. Lower bounds again reflect the political nature of these objectives. It is assumed that a decrease of more than 20% in direct revenues, or of 25% in indirect revenues (i.e., forestry's contribution to GDP), would be politically unsatisfactory. The bounds are ($139.2 million, $174.0 million) for direct revenue and ($640.5 million, $854 million) for indirect revenue.

The final objective is that of wilderness expansion. Any increase in protected areas will most likely come from Crown land. A doubling of protected area on the Island would mean that 660,000 ha would be removed from the working forest, or about 30% of total Crown land. It is assumed that protecting almost a third of the public land on Vancouver Island would allow all PAS objectives to be met; thus, the decision maker is assumed to be satisfied completely at that level of wilderness protection. The lower level for the fuzzy objective is defined as the current area under protection (341,000 ha), a level below current legislated requirements and thus considered unacceptable.

10.4 Empirical Results

A crisp MODM (in the form of an NPV maximizing model), a fuzzy MODM and a fuzzy MODM with imprecise coefficients were constructed and solved.[11] First, the results of the crisp formulations are compared to the fuzzy MODM with crisp parameters, which is then compared with the model where the coefficients are imprecise.

Fuzzy and crisp MODM

The crisp MODM model (or CRISP) maximizes net present value subject to employment and wilderness conservation constraints. NPV is defined as the sum of logging benefits, recreation benefits, nonuse values and direct government revenues. The fuzzy MODM (FUZZY) considers all objectives of the previous section as independent and equal in terms of priority. The land allocations resulting from these models are presented in Table 10.2.

Table 10.2: Simulation Results for Fuzzy and crisp MODM: Land Allocation (hectares).

Site Quality	Management Intensity	Model CRISP	Model FUZZY
	High	223845	209848
Good	Integrated	0	13994
	Low	0	0
	High	891016	363747
Medium	Integrated	0	527269
	Low	0	0
	High	0	0
Poor	Integrated	0	0
	Low	754644	658671
Protected		341000	436976

The CRISP model concentrates good and medium quality sites into high intensity management regimes, and allocates all poor sites to the low intensity system. In contrast, the FUZZY model places the larger proportion of medium quality forest land under integrated management as well as a small

[11] Although linear, the models were solved using a nonlinear approach within EXCEL.

amount of good quality area. Total area assigned to the high management regime is less, and protected area is greater. Given that the B.C. Government has indicated recently that logging should take better account of non-timber benefits, it is interesting to note that the outcomes of the fuzzy model are more in line with government policy than the outcomes of the crisp model. On the other hand, it must be noted that there is very little movement (if any) of high quality land into protected areas in both the crisp and the fuzzy models, which is at odds with the philosophy and intent of the Protected Areas Strategy.

Differences in the allocation schemes are evident in the levels attained for each of the objective functions (see Table 10.3). The logging benefits are greater under the crisp formulation, and as a consequence so are direct government revenues and employment, but logging benefits are of such a magnitude that they dominate other forest services when weighed equally.[12] The fuzzy model provides a higher return on the other accounts.

Table 10.3: Simulation Results for Fuzzy Possibilistic MODM: Monetary and Employment Benefits.

Benefits	**CRISP**	**LOWER** $\alpha = 0.95$	**FUZZY** $\alpha = 1$	**UPPER** $\alpha = 0.95$	$\alpha = 0.9$	$\alpha = 0.80$
Logging ($ mil.)	68.6	54.2	55.3	56.3	57.3	66.1
Recreation ($ mil.)	66.5	64.7	76.7	79.9	82.9	88.1
Preservation ($ mil.)	19.1	21.7	24.0	26.2	28.4	32.6
Direct revenue ($ mil.)	179.2	163.2	166.7	170.0	173.3	179.7
Indirect revenue ($ mil.)	875.9	806.3	814.8	822.7	830.6	846.0
Employment (jobs)	15,066	13,948	14,054	14,151	14,250	14,445

[12]Interestingly, both models provide an annual harvest that exceeds the current LRSY of 11.0 million m^3. This is largely due to the application of intensive silvicultural practices to a substantial portion of the Crown land base, contrary to current conditions.

There is a membership function (μ_i) associated with each of the objective functions indicating the level of satisfaction attained for each objective. Focusing on the FUZZY model, the minimum degree of satisfaction is attained for four of the seven objectives (this table is not reported here). Given that the model provides an efficient solution, the interpretation is that it is impossible to increase the satisfaction level for any one of these four without compromising that of at least one of the other three. The standoff is between logging benefits and employment on the one hand, and preservation values and protected areas on the other. This situation reflects the reality of the conflicts identified in the Vancouver Island land use debate.

Perhaps the difference in performance between the crisp and fuzzy models can be explained by the fact that the former resembles a cost-benefit analysis whereas the latter is more like a true MODM. However, it is impossible to conclude to what extent the divergence between the crisp and the fuzzy models is caused by the difference between crisp and fuzzy modelling *per se*. Interpretation is blurred by the different nature of both models: maximizing NPV subject to constraints in the one and balancing objectives in the other.

A fuzzy MODM with imprecise coefficients

Symmetric fuzzy numbers are used to model the uncertainty surrounding the precision of the parameters in the model. One purpose is to gain some understanding about the sensitivity of the solution to uncertainty in parameter definition. By lowering the value of α (the fuzzy MODM has an implicit α value of 1), the effect of this uncertainty upon optimal land allocation can be explored. At any value of $\alpha < 1$ there are two solutions to consider. The first is from the lower-bound version of model (18), where parameter values take on a less-likely and lower value (the LOWER results); the second is from the upper-bound version of (18), generating a solution based on parameter values that have the same degree of membership in the fuzzy number but higher value (the UPPER model results). The results from the two models are provided in Tables 10.3 and 10.4.

Table 10.4: Simulation Results for Fuzzy MODM with Imprecise Coefficients: Land Allocation (hectares).

Management Intensity	LOWER $\alpha = 0.95$	FUZZY $\alpha = 1$	UPPER		
			$\alpha = 0.95$	$\alpha = 0.9$	$\alpha = 0.80$
Good Site Quality					
High	223,842	209,848	136,697	69,680	0
Integrated	0	13,994	87,145	154,162	223,842
Low	0	0	0	0	0
Medium Site Quality					
High	753,259	363,747	330,668	300,593	213,323
Integrated	137,757	527,269	560,328	590,423	677,693
Low	0	0	0	0	0
Poor Site Quality					
High	0	0	0	0	0
Integrated	0	0	0	0	0
Low	543,453	658,671	645,434	632,048	605,625
Protected Area	552,194	436,976	450,213	463,599	490,022

The most obvious result obtained from the variation of the degree of membership is in the asymmetry of the feasible solution space. While solutions may be obtained for any value of α using the UPPER model, feasible solutions do not exist below a degree of membership level of 0.92 for the LOWER model. Parameter values are unable to provide any level of satisfaction of at least one of the objectives; in this case, the limiting objective is timber benefits.

An unexpected result is that the LOWER model, with $\alpha = 0.95$, provides for over 10% more protected area than does any of the other scenarios considered. The minimum amount is provided by the FUZZY model. The rationale for this is that the LOWER model concentrates the good and medium quality sites into the high intensity management category, a massive shift of almost 400,000 ha as compared to the integrated management allocation level of the FUZZY model. This occurs in response to the lower estimation of both wood yield and wood value. This causes a large reduction in nonuse benefits as the high intensity management category does not contribute to this objective. The shortfall is replaced by the allocation of poor quality area, with it's negative logging value, into the protected area category.[13] Harvest volume declines under this LOWER scenario and job numbers fall slightly (Table 10.3). Monetary benefits are also slightly lower with the largest change observed in recreation benefits; logging benefits are virtually unchanged (Table 10.3).

It is our opinion that the dramatic effects of small parameter adjustments provides an additional reason to model the imprecision associated with parameter estimates explicitly.[14] The evidence from Table 10.4 indicates that, if there is indeed imprecision in the parameters but it is not modeled as such [i.e., imprecise parameters are modeled as if they are crisp, possibly by tolerating small violations of constraints, as in Bare and Mendoza (1992), or Mendoza and Sprouse (1989)], then the results may be distorted. The analysis conducted here enables decision makers to identify sources of imprecision when it comes to land allocation. As information gathering is costly and with limited funds available to overcome parameter imprecision, there is a definite value in knowing which areas to research first. Under the current conditions it seems that it is most crucial to address imprecision in the logging benefit parameters.

The results obtained from the UPPER model as α is decreased are as expected. All parameters of the model increase in value as α decreases, resulting in a higher provision of benefits from each hectare of land considered. Increasing yields and wood values allow less area to be allocated to the

[13]This is likely an unacceptable result if quality of protected areas is important.

[14]In this specific analysis, the solution provided by the FUZZY model is sensitive to overestimation of the true parameter values. If values are realized at a generally lower level than those judged most likely, a large shift in resource allocation is required to obtain the best solution as judged by the maximization of minimum objective satisfaction.

high intensity regime and more to integrated management. The result is an increase in both recreation and nonuse benefits. Harvest volume rises and job provision increases, evidence of the less possible higher per ha yield estimates and a greater number of jobs per unit of harvest.

10.5 Conclusions

Increasing weight is placed on non-timber values in managing forests or making allocation plans for woodlands. The public is also becoming more involved in the planning process. Both trends are evident in British Columbia in the new forest management policies aimed at environmental concerns and a CORE process the relies on stakeholder participation. In most instances, economic efficiency is only one of many competing considerations, with cost-benefit analysis often relegated to a status below that of other concerns, such as employment. Hence, decision models need to be sensitive to the existence of multiple objectives and the fact that the objectives themselves are vague and the parameters that characterize them are imprecise. This study applied fuzzy MODMs with and without incorporating imprecise coefficients, to the problem of allocating public forest land on Vancouver Island, comparing the results of the fuzzy models with a more traditional, crisp approach.

Given the nature of the process that is to be modeled, our conclusion is that the fuzzy approaches can be judged a distinct improvement over the traditional approach of constrained maximization of net present value. For example, the fuzzy MODM allocated about 25% of the land base to integrated timber management, while the traditional (crisp) model concentrated land into the extreme categories of low (natural regeneration) and intensive timber management intensity. Given the intensity of the land use conflicts on Vancouver Island and the context of consensus seeking interest groups, moderation of management intensity seems desirable. The area assigned to the protected category was greater in the fuzzy MODM than in the crisp model, as was the number of direct jobs provided in the forest sector. We conclude that the decision by CORE not to rely on maximization of NPV is confirmed by comparing the results of the crisp and fuzzy models. The fuzzy solution was obtained without needing to specify precise values for objectives, and without an explicit ranking or weighing of the objective functions. The fuzzy MODM also clearly identified those objectives that were in direct conflict with each other, and thus the areas where compromise is required if satisfaction levels are to be increased.

The results from the fuzzy MODM model with imprecise coefficients suggest that the approach of combining fuzzy parameter specifications with fuzzy objectives constitutes an improvement over the fuzzy MODM. However, the model specified in this study was very sensitive to the possibility of lower realizations of parameter values, but this only highlights the importance of modelling imprecise parameters using fuzzy numbers instead of as flexible constraints. The analysis provides insight into what kind of additional information is especially valuable for obtaining robust land allocations—robust in the sense that small mis-specifications of parameters will not cause massive shifts from one land use option to another, which in practice may be costly to achieve.

Finally, areas for future research suggest themselves. The most important of these is that of getting stakeholders/decision makers involved in the development of both fuzzy objectives and fuzzy numbers for the technical coefficients of the decision model. Fuzzy set theory offers a means of combining information from various stakeholders. Research is required to determine how information can be updated when those involved in the decision process are presented with results and, just as importantly, how natural language can be used to develop the required fuzzy measures.

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CHAPTER 11

OPTIMAL HARVESTING AND THINNING WITH STOCHASTIC PRICES[1]

Abstract:

This chapter rigorously incorporates optimal thinning decisions for an even-age stand into an optimal harvesting model with fluctuating stumpage prices. The model optimally determines how often and at what ages to thin. In addition to theoretical results that are analogous to optimal harvesting models without thinning, new theoretical results include increases in the net present of land, stumpage and for some harvest reservation prices. The model is numerically simulated using parameters estimated from Dutch data for *Pinus sylvestris*. The simulation results suggest that the gains from incorporating thinning are significant, when compared to the Faustmann approach, and at least modestly important, when compared to optimal harvesting models without thinning. The simulation results also indicate that expected thinning age may decrease significantly while the expected harvest age increases, when compared to the Faustmann approach.

11.1 Introduction

Thinning is an integral part of forest management. Thinning decisions determine which trees will be left for final harvest, and how much competition these trees will experience. Standard practice for even-aged stands is to initially plant several-fold more trees than eventually will be optimal to

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[1]This chapter is based on: Brazee R.J. and E.H. Bulte (1996), Optimal harvesting and Thinning With Stochastic Prices, Working paper. It also builds on Bulte E.H., J.E. Pennings and W.J.M. Heijman (1996), Managing Resource Price Risk: Futures Markets, Sustainability and Efficiency, *Environmental and Resource Economics* (8), 351-366.

harvest.[2] There are several biological reasons for a high level of initial stocking, followed by thinning. A high stocking level followed by thinning decreases competition for limited resources which may increase stand volume, wood quality and diameter of the remaining trees, as well as reduce wind damage (Schutz and van Tol 1982).

Thinning decisions must be made without perfect information of the future. Due to well-documented volatility (Haight 1990; Haight and Smith 1990; Haight and Holmes 1991; Lohmander 1987; Lohmander 1992; Slangen 1987), stochastic stumpage prices are an important source of uncertainty. Fluctuating stumpage prices impact both the commercial and pre-commercial aspects of thinning decisions, and highlight the difficulties associated with evaluating thinning regimes. For example, it was recently suggested that forest owners in the Netherlands are thinning their stocks at a sub-optimally low level in response to low stumpage prices (Biersma 1994). With stochastic stumpage prices, the validity of this and related claims regarding optimal thinning decisions are unresolved issues.

The purpose of this chapter is to rigorously incorporate thinning decisions into optimal harvesting models with fluctuating stumpage prices. First, we present a model that includes both optimal harvest and optimal thinning responses to fluctuating stumpage prices. Solving this model using dynamic programming extends the theory of optimal harvesting under stochastic prices to the include thinning of even-aged single species stands. The model allows us to optimally determine how often and at what ages to thin.[3] We present new theoretical results that arise from the inclusion of thinning. We then use

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[2]In the Netherlands, for instance, standard silvicultural practice for *Pinus sylvestris* prescribes planting about 3000 trees per hectare initially and thinning this stock back to 300 trees as the stand matures (Schutz and van Tol 1982).

[3]We rely on a fixed thinning intensity at every age. That is, the question of optimal thinning intensities is not addressed. The theoretical extension to include thinning intensities is straightforward. However, since the inclusion of N possible thinning intensities requires the inclusion of N-1 additional states at every age, and these states must be accounted for each at succeeding harvest age, simulating N thinning intensities adds N-1 to the power of the length of the planning horizon minus one. This generates a particular severe case of the "curse-of-dimensionality." Fortunately, the fixed thinning intensity is consistent with standard practice as virtually all silvicultural prescriptions explicitly or implicitly assume fixed thinning intensities.

the model to simulate quantitative results for parameters estimated from Dutch data on pine, the most important commercial species in the Netherlands.

Some of the theoretical results with thinning are analogous to previous qualitative results without thinning. Other qualitative results arise only when thinning is allowed. Analogous results include the existence of an optimal reservation price policy for both harvesting *and* thinning, and that the net present value of both land and stumpage increases with the spread of the stumpage price distribution as do most reservation prices. The new theoretical results from incorporating thinning include increases in the net present value of both land and stumpage, increase in some harvest reservation prices and demonstration that harvest reservation prices are always at least as high as thinning reservation prices. The impact of spread and discount rate on optimal thinning reservation prices is ambiguous, depending on whether commercial or pre-commercial considerations dominate. The simulation results suggest that with the introduction of optimal thinning, the gains in the net present value of land and stumpage may be significant, and that the relevant reservation prices for harvesting may increase significantly.

11.2 Optimal Harvesting and Thinning Model

Key thinning decisions are at what age to thin, how often to thin, and what proportion of stems to cut. Thinning decisions are usually characterized by the existence of both commercial (or short-term) and pre-commercial or (long-term) investment objectives. The commercial objective of thinning is to maximize the short-run profits from selling thinned timber. The pre-commercial or investment objective is to improve the quality of the residual stand to increase long-run profits. When a stand is young, revenue from selling thinned trees is usually low, costs are often relatively high, and net revenues are sometimes negative. Pre-commercial objectives are more important than commercial objectives, i.e. owners thin to increase future profits. As a stand ages, revenue from selling thinned trees increases, and commercial objectives become more important, i.e. owners are more likely thin to increase current rather than future profits.

Over the past decade several papers analyze the effects of fluctuating stumpage prices on optimal harvesting decisions. An important omission to date is the inclusion of thinning responses to fluctuating stumpage prices in single-species, even-aged stands. Previous research that addresses thinning with fluctuating stumpage prices focuses on species selection in a mixed-species stand (Carlsson 1992, Lohmander 1992) and thinning intensity for uneven-age management (Haight 1990).[4] The only previous paper that addresses an even-aged, single-species stand (Teeter and Caulfield 1991) presents simulated thinning results for loblolly pine stands in the southern United States, when sawnwood prices fluctuate. The authors find that there exist circumstances in which the thinning regime with stochastic stumpage prices is different from the thinning regime when stumpage prices are deterministic, and other circumstances in which the thinning regimes are similar. An important limitation in this analysis is the assumption of a fixed, rather than optimally determined rotation age.[5]

The forest owner's objective is to maximize NPV per hectare of land for every age of stumpage. We adopt several simplifying assumptions common to optimal harvesting models. We assume that the land will be used for forestry forever; when one rotation is harvested another is replanted. The

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[4]The primary theoretical and empirical results for multiple species analysis is that mixed species stands are preferred to single species, when species prices are not perfectly correlated and more information about relative prices at harvest age is available at the thinning age (For empirical results, see Carlson 1992; and Lohmander 1992. For theoretical results, see Lohmander 1992). The empirical results for uneven-age stands are that more (and younger) age classes should be thinned more at higher realized prices (Haight 1990).

[5]In particular the question, what would be the optimal timing of thinning if optimal harvesting responds to realized prices, is not addressed. This is a potentially significant omission. Since volume at harvest is usually larger than volume at a thinning, the potential gains from allowing harvest age to respond to realized prices are likely to be much larger than the potential gains from allowing thinning decisions to respond to realized prices. Given the reported size of the potential gains in the NPV for some of the previous simulations, this is a potentially restrictive assumption. Also, the omission may be unusual; most other papers in this line of research on stochastic prices allow harvest age to respond to realized prices.

(risk neutral)[6] forest owner faces exogenous stumpage price fluctuations. Even-age growth and yield functions including volume responses to thinning are known with certainty. T is the maximum harvest age. At age T, which may be arbitrarily large, stumpage that has not previously been harvested, will be harvested regardless of stumpage price. In addition, we assume that the land owner is able to thin or harvest at any of a range of biologically determined ages, 0 to T. Therefore at every age between 0 and T-1, the owner has the option of harvesting, thinning or waiting. For every age k the owners objective can be formulated as a dynamic programming problem:

$$V(k \mid x_{k-1}) = MAX \quad [p(k)Y(k \mid x_{k-1}) + V(0) \ , \quad p(k)Z(k \mid x_{k-1}) + \beta V(k+1 \mid x_{k-1}, 1) \ , \quad \beta V(k+1 \mid x_{k-1}, 0)] \tag{1}$$

where, β is the discount factor; x_k indicates the state of the stand at age k, i.e. it indicates the previous thinning record (c_1, c_2, ..., c_{k-1}) (where 0 represents no thinning, and 1 represents thinning); k is current stumpage age; $p(k)$ is stumpage price at age k; $V(0)$ is the value of bareland; $V(k+1 \mid x_{k-1})$ is the value of land with stumpage age $k+1$, with previous thinning record x_{k-1}; $Y(k \mid x_{k-1})$ is volume available for harvest at age k, with previous thinning record x_{k-1}, and $Z(k \mid x_{k-1})$ is volume available for thinning at age k, with previous thinning record x_{k-1}.

If the land owner harvests at age k, she receives a revenue from harvest of $p(k)Y(k \mid x_{k-1})$ (stumpage price times volume harvested) plus use of the bare land which has value, [$V(0)$]. If the land owner thins at age k, she receives a revenue from thinning of $p(k)Z(k \mid x_{k-1})$ (stumpage price times volume thinned) plus the discounted value of a thinned stand age $k+1$ in the next period, $\beta V(k+1 \mid x_{k-1}, 1)$. If the land owner neither harvests nor thins,

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[6]Bulte *et al.* (1996) have used an analogue of this reservation price approach for a mining problem with risk averse resource owners. In recent years, there has been an increase in the use of price risk management instruments (e.g. futures) in natural resource markets, which is expected to result in more stable spot prices around a Hotelling-style trend. Using an expected value-variance (EV) model, Bulte *et al.* (1996) demonstrated numerically that this stabilizing effect has ambiguous results for the (expected) time of depletion and the utility of exploitation for risk averse resource owners. An important variable here is the degree of risk aversion of resource owners; the less risk averse, the less beneficial price stabilization is.

then she retains the discounted value of an unthinned (at age k) age $k+1$ stand in the next period, $\beta V(k+1|x_{k-1}, 0)$. Given the historical record of previous thinning within the rotation at every age k, the land owner chooses the option that maximizes the NPV of land for age k stumpage.

We assume prices are drawn randomly from a known and stationary distribution. As we discuss in section 11.3, the Dutch data that we base our simulation on supports this assumption of a stationary distribution. However, characterizing the underlying structure of stumpage price fluctuations for different species and regions is an active area of research. As demonstrated in previous research the structure of stumpage prices determines the magnitude of potential gains from optimally incorporating stumpage price fluctuations into harvesting decisions. The underlying structure of stumpage price fluctuations can be broadly characterized as three types: random walk, auto-regressive, and random draw.[7] The reported potential gains to landowners from including stochastic prices in optimal harvesting decisions by a reservation price approach vary from none (random walk, but see Thomson 1992) to substantial (random draw). Two important implications of having a random draw rather than random walk or auto-regressive specification of price fluctuations are mathematical tractability and the magnitude of potential gains from optimal management. With a random walk or auto-regressive formulation, future stumpage price distributions are conditional on current prices (random walk) or current and previous prices (auto-regressive).[8] Although it is possible to incorporate thinning into these models, more complex notation

[7]For details of this characterization, see Haight and Holmes (1991); and Lohmander (1992). In a random draw model the stumpage prices uncorrelated over time and the stumpage price distribution is stationary (Brazee and Mendelsohn 1988; Haight 1990; Hultkrantz 1995; Lohmander 1987). In a random walk the stumpage distibution is unstationary and conditional on the current price (Washburn and Binkley 1990; Thomson 1992). If prices are auto-regressive, the stumpage price distribution is conditional on both current and previous prices; the distribution may be stationary or unstationary (Carlsson 1992; Clarke and Reed 1989; Haight and Holmes 1991; Lohmander 1992; Teeter and Caulfield 1991; Washburn and Binkley 1990).

[8]An explanation for autoregressive prices on wood markets may be that there is a time lag between the decision to sell timber, the harvesting and the final sale. This argument is less valid for stumpage prices, which is what we consider in this chapter.

and analysis is necessary, and the number of states to simulate rises exponentially with the number of conditional stumpage price distributions.

In previous models with randomly drawn fluctuating stumpage prices, a reservation price policy for harvesting is proved optimal.[9] That is, for every age of stumpage the owner determines the reservation price for harvesting. If the actual price is above the reservation price, the owner harvests and replants. If the actual price is below the reservation price, the owner does not harvest that period. The intuition underlying the optimality of a reservation price is that for every stumpage age an owner will either accept or reject any actual price, i.e. for a given age of stumpage, the landowner will never accept a particular price in one rotation, and reject the same price in another rotation. Based on the same reasoning, and using the same techniques, it is possible to demonstrate the optimality of a reservation price policy for both thinning and harvesting.

In Appendix 11.2. A we show by contradiction that for any given stumpage age, the thinning reservation price will always be less than or equal to the harvesting reservation price. This implies that the objective function in (1) can be re-formulated for every stumpage age k as:

$$\begin{aligned} V(k \mid x_{k-1}) \; = & \int_{A(k \mid x_{k-1})}^{\infty} f(p)\,[p(k)\,Y(k \mid x_{k-1}) + V(0)]\,dp \; + \\ & \int_{\alpha(k \mid x_{k-1})}^{A(k \mid x_{k-1})} f(p)\,[p(k)\,Z(k \mid x_{k-1}) + \beta V(k+1 \mid x_{k-1}, 1)]\,dp \; + \\ & \beta \int_{0}^{\alpha(k \mid x_{k-1})} f(p)\,V(k+1 \mid x_{k-1}, 0)\,dp \end{aligned} \tag{2}$$

where at age k, $A(k)$ and $\alpha(k)$ are the optimal reservation price for harvesting and thinning, respectively, and $f(p)$ is the probability density function of stumpage prices.

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[9]Proofs of the optimality of a reservation price policy are provided in Brazee (1987) and Lohmander (1987).

The first integral is the benefits of harvesting at age k, which is the sum of timber revenues and the value of bare land. Timber revenues are the realized stumpage price multiplied by stumpage volume harvested. Stumpage volume harvested at age k is determined by previous thinning decisions x_k. If the actual price in period k exceeds the harvest reservation price, the land owner harvests the stand. The second integral is the benefits of thinning at age k, and harvesting the stand later. If the price is lower than $A(k)$, the harvest reservation price but higher than $\alpha(k)$, the thinning harvest reservation price, the owner should thin the stand. The commercial component of thinning is the revenue from thinning, which is the realized stumpage price multiplied by stumpage volume thinned. Similar to harvesting, stumpage volume thinned at age k depends on previous thinning decisions. The pre-commercial component of thinning is the change in discounted future revenues due to changes in wood, tree or stand quality from thinning. The third integral is the benefits of not thinning and not harvesting, and having stumpage age $k+1$, next period. If the realized price is lower than $\alpha(k)$, the land owner neither harvests nor thins.

The $2^k(T\text{-}1)$ necessary first order conditions are found by differentiating equation (2) with respect to $\alpha(k)$ and $A(k)$ for every k such that $0 < k \leq T\text{-}1$:[10]

$$-f(\alpha(k\,|\,x_{k-1}))[\alpha(k\,|\,x_{k-1})Z(k\,|\,x_{k_1}) + \beta V(k+1\,|\,x_{k-1},1)] + \beta f(\alpha(k\,|\,x_{k-1}))V(k+1\,|\,x_{k-1},0) = 0 \tag{3}$$

and

$$f(A(k\,|\,x_{k-1}))[A(k\,|\,x_{k-1})Z(k\,|\,x_{k-1}) + \beta V(k+1\,|\,x_{k-1},1)] - f(A(k\,|\,x_{k-1}))[A(k)Y(k\,|\,x_{k-1}) + V(0)] = 0 \tag{4}$$

These $2^k(T\text{-}1)$ first order conditions simplify to:

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[10] Sufficient second-order conditions for a maximum are met by the concavity of $V(k\,|\,c_1, c_2, \ldots, c_{k-1})$ in both $\alpha(k\,|\,c_1, c_2, \ldots, c_{k-1})$, and $A(k\,|\,c_1, c_2, \ldots, c_{k-1})$, and the zero cross-partial derivatives.

$$\alpha(k \mid x_{k-1}) = \frac{\beta V(k+1 \mid x_{k-1}, 0) - \beta V(k+1 \mid x_{k-1}, 1)}{Z(k \mid x_{k-1})} \tag{5}$$

and

$$A(k \mid x_{k-1}) = \frac{\beta V(k+1 \mid x_{k-1}, 1) - V(0)}{Y(k \mid x_{k-1}) - Z(k \mid x_{k-1})} \tag{6}$$

The reservation price $\alpha(k)$ is the price at which the sum of the commercial benefits from thinning and the expected present value of future harvesting a thinned stand are equal to the benefits of future harvesting and thinning an unthinned stock. The reservation price $A(k)$ is the price where the revenues from current harvesting are equal to the expected present value of future harvesting and possible thinnings.

This implies that the present value of all future expected revenue starting with bare land is given by the complex formula:

$$V*(0) = -C + \sum_{i=1}^{N} B^i \sum_{j=1}^{2^{i-1}} \Phi(x_{i-1}^j) * \\ \left[\int_{A(i \mid x_{i-1}^j)}^{\infty} f(p)(pY(i \mid x_{i-1}^j) + V(0))\,dp + \int_{\alpha(i \mid x_{i-1}^j)}^{A(i \mid x_{i-1}^j)} f(p)(pZ(i \mid x_{i-1}^j))\,dp\right], \tag{7}$$

where C is planting cost, x_{i-1}^j is the j^{th} state of the i^{2-1} possible states for stumpage at time i-1, and Φ is the probability of occurrence, i.e.

$$\Phi(x_{i-1}^j) = \prod_{g=1}^{i-1} F(w_g), \tag{8}$$

where $F(w_g)$ is the probability of the action taken at time g for thinning history x_{i-1}^j, that is:

$$F(w_g) = \int_0^{\alpha(g \mid x_{g-1}^j)} f(p)\,dp \tag{9}$$

if the stand is not harvested or thinned at time g for x_{i-1}^j, and

$$F(w_g) = \int_{\alpha(g|x_{g-1}^j)}^{A(g|x_{g-1}^j)} f(p)\,dp \qquad (10)$$

if the stand is thinned at time g for x_{i-1}^j. The $2T$-1 necessary conditions of (5), (6) and (7) must be solved simultaneously.

Analogous to optimal harvesting models incorporating stumpage price fluctuation without thinning, a reservation price policy is optimal for both harvest and thinning decisions, the net present value of both land and stumpage and most harvest reservation prices increase with the spread of the stumpage price distribution. Several other new qualitative results which have no analogues without thinning also hold.

First, we have shown that for every age of stumpage the thinning reservation price is less than or equal to the harvesting price. Second, the introduction of thinning increases the expected net present value of both land and stumpage. The rationale for this result is straightforward. Since thinning is optional, introducing thinning increases flexibility through increasing choices. Thinning offers the possibility of increasing future stand values, and realizing early revenues. The landowner will only choose to thin if it increases expected net present value. Third, the introduction of thinning increases harvest reservation prices for some ages. An explanation of this result rests on the previous result, that is, the possibility of future thinning increases the expected net present values of both land and stumpage. For ages near the maximum harvest age, T, optimal harvest reservation prices decrease as the increase in the expected value of bare land from thinning outweighs the increase in the expected value of harvesting at older ages. For younger ages the increase in the expected value of harvesting at older ages dominates the increase in the expected value of bare land, and optimal harvest reservation prices increase. The ranges for higher and lower reservation prices varies from species to species and across price distributions.[11] Finally, the impact of

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[11]Our simulation results indicate that for optimal harvest reservation prices increase for all but the largest ages, and for all relevant harvest ages. For a more detailed description of increasing and decreasing harvest reservation prices without thinning, see Brazee and Mendelsohn (1988).

increasing the spread of the stumpage price distribution on thinning reservation prices is ambiguous. This ambiguity arises from the opposing impacts on the commercial and pre-commercial aspects of thinning. Increasing the spread increases both the expected revenue from thinning and from the expected revenue at harvest. The increase in thinning revenues tends to postpone thinning to capture gains from larger thinning volumes (that is, as long as these gains outweigh the effect of discounting). Higher harvest prices encourage earlier thinning to increase average stem size at harvest. Which of these impacts dominate is an empirical matter.

11.3 Data description

To examine the importance of the qualitative results, we present a numerical simulation. To simulate conditions (5), (6) and (7) requires information on the stumpage price distribution, stumpage volume functions and the responses in stumpage volume and stumpage prices from thinning. Although these information requirements are extensive, adequate data on Dutch forests exist on which to base the simulation.

As discussed previously, the structure of stumpage prices over time is an important determinant of the size of potential gains from incorporating stochastic stumpage prices. To address this issue, we analyze real annual Dutch stumpage prices for pine during the period 1975-1991.[12] The autocorrelation function (ACF) shows two isolated significant autocorrelations at high lags. These can be ignored (SPSS/PC+ 1990). The partial autocorrelation function (PACF) showed no significant autocorrelations at any lag (see Appendix 1). These results support application of the random draw model.[13]

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[12]Additional data for 1969-1974 are available. However, due to heavy storms in the early seventies, market conditions during this period were quite different from the period 1975-1991. Also, the available price data does not distinguish between different species of pine. We simulate volume for the predominant species, *Pinus sylvestris*.

[13]A nonstationary series would have an ACF that remains significant for half a dozen or more lags. Autoregressive processes would have an exponentially declining ACF and spikes in the first one or more lags of the PACF, where the number of spikes indicates the order of the

For simplicity we construct a model with a constant mean and standard deviation.[14]

In empirical research to date for optimal harvesting models, attention has focused on models that have a constant mean unit price. With a constant mean price thinning serves no pre-commercial purpose. However, the assumption of constant unit prices is unrealistic: as forests mature and stem size increases, the price per unit of wood rises. For the simulation we estimate the relation between diameter and price from data of the Dutch State Forestry Service and the Central Bureau of Statistics (see Slangen 1987). Data were collected through questionnaires sent to forest owners and timber merchants during the period 1969-1981. About 11,000 transactions were reported. For most of the transactions information includes tree species, price, sale volume, average diameter, wood quality, phase (stumpage or felled) and location (comparable data are not available after 1981). For simplicity, a linear specification of the relation between price and diameter is estimated.[15] The number of stems per cubic meter (*CT*) serve as a proxy for diameter. The relation for pine species is (in 1985 US$):

autoregression. Neither are consistent with the ACF and PACF for pine.

[14]For the period considered, pine prices exhibit a slight downward trend over time. This trend could be incorporated into the model by reducing the mean of the distribution over time. In the interests of simplicity, we will not include this trend.

[15]Linearity is a crude approximation, as the actual relationship will include jumps in prices as product class changes. However, inclusion of higher order terms failed to capture the effect of product classes. Stumpage quality is assumed to be good. Sale volume and location were statistically insignificant and omitted from the simulation.

$$P(t) = \underset{(42.4)}{17.0} - \underset{(-25.4)}{0.3\,CT} \tag{11}$$

$$R^2 = 0.22, \quad F = 191$$

t-statistics are between brackets. A Breusch and Pagan test (see Maddala 1992) did not reject the hypothesis of homoscedasticity across the diameter distribution.[16]

To simulate forest growth for *Pinus sylvestris*, we use OPTAB, a Dutch growth and yield simulator (see Faber 1989).[17] In OPTAB thinning does not change future stand volumes much, a trait shared by many growth and yield models. However, since aggregate growth is limited to fewer trees, the stand produces more valuable trees. This captures the pre-commercial element of thinning.

Stumpage volume projections were made for hypothetical stands. Starting points for the projections were based on stocking density and volumes for stands using standard Dutch practice (Faber 1989, Schutz and van Tol 1982). The thinning intensity is set at 30%. That is, 70% of the basal area remains after thinning. Good soil quality is assumed. The range of allowable harvest ages in the simulation is 15 to 65. Although any age range is somewhat arbitrarily chosen, stands are rarely thinned or harvested at ages younger than 15 years, and 65 years is almost twice the maximum mean annual increment (MAI) for pine in the Netherlands.

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[16]With heteroscedasticity (more variable prices for bigger trees), the pre-commercial aspect of thinning for forest owners would gain weight. (Heteroscedasticity can also imply that prices vary less for bigger trees, and then the reverse would hold.)

[17]The growth and yield functions in OPTAB are based on observations in (Dutch) stands that are regularly thinned. It is assumed that the program provides a reasonable approximation of the "no thinning" option and the option that only once is thinned.

11.4 Simulation procedure

Simulation of the reservation prices requires simultaneous solution of (5), (6) and (7). Given the computational difficulties and computer time required to simultaneously solve $(T\text{-}1)*2^{T-1}$ non-linear equations, we simplify the calculations in two ways. First, we limit the number of thinnings. With the possibility of thinning at every age, at age k, the stand volume assumes 2^k different states, depending on previous thinning decisions c_1, c_2, ..., c_{k-1}. This also implies at age k there are 2^k different reservation prices for both thinning and harvesting. To circumvent this problem we restrict the number of allowable thinnings to 1 (note that this implies that the important question of how often to thin will not be addressed in the numerical simulation). This reduces the number of different states at age k from 2^k to $k+1$. In addition to the computational advantages, reducing the number of thinnings also allows graphical display of the reservation price paths. Extending the simulation further to allow more thinning is straightforward but tedious.[18]

The assumption of one thinning implies that for the unthinned stand the first order conditions (5) and (6) reduce to:

$$\alpha(k \mid 0,0...0) = \frac{\beta[V(k+1 \mid 0,0,...0,0) - V(k+1 \mid 0,0...0,1)]}{Z(k \mid 0,0...0)} \qquad (12)$$

and

$$A(k \mid 0,0...0) = \frac{\beta V(k+1 \mid 0,0,...0,1) - V(0)}{Y(k \mid 0,0...0) - Z(k \mid 0,0...0)} \qquad (13)$$

For the thinned stand (6) becomes:

[18] We have simulated the flexible model with two thinnings. This did not provide additional insights, except that the marginal gain in NPV of a second thinning is smaller than the gain in NPV of the first thinning. This is consistent with our expectations that, given the short rotations in the Netherlands, the net benefits of an additional thinning decrease as the number of thinnings increases.

$$A(k\,|\,0,0\ldots0,1,0,\ldots0) = \frac{\beta V(k+1\,|\,0,0\ldots0,1,0,\ldots0) - V(0)}{Y(k\,|\,0,0\ldots0,1,0,\ldots0)} \quad \mathbf{(14)}$$

Secondly, we simplify by using an iterative backwards-recursive procedure rather than simultaneously solving all of the first order conditions. The procedure is to initially guess the value of bareland; for simplicity we start with the Faustmann value of bare land. Then given an initial value of bareland, using (12), (13), (14) and (7), we calculate harvesting and thinning reservation prices recursively starting from age T-1 to age 1. A new value of bareland is then calculated using (7). The procedure is then repeated with the updated value of bareland. The iterative procedure is continued until the updated value of bareland does not significantly differ from the old estimate. The proof that this procedure converges to within any arbitrary tolerance to the unique optimum is presented in appendix 2.B.

11.5 Simulation results

The simulation results of the Faustmann model and the flexible management model are presented in Table 11.1. Table 11.1 highlights the effect of thinning on expected NPV and expected harvesting and thinning age. The rows reflect respectively (1) the Faustmann model with thinning; (2) the flexible management model without thinning; and (3) the flexible management scheme with 1 thinning. The estimated standard deviation was approximately 7. Standard deviations of 5 and 9 were reported as an indication of the impact of smaller or larger fluctuations.

Compared to the Faustmann model with thinning, the flexible management scheme without thinning increases the expected NPV of bareland substantially. These gains range from 14% to 36%, and are consistent with gains reported by, for example, Brazee and Mendelsohn (1988). Allowing thinning in the flexible management model provides an additional increase in expected NPV of approximately 7% for our data. As we discuss in section 11.2, both of these results are anticipated and arise from increased flexibility (the landowner has the option to thin besides the option to harvest, depending on stumpage prices). In Figure 11.1, we have depicted the harvest and thinning reservation price paths for $\sigma = 7$. The thinning reservation price path is slightly upward sloping, the harvest reservation price path is downward sloping. The paths converge over time. This implies that the probability of thinning decreases as stumpage ages. This is consistent with standard silvicultural practices.

Table 11.1: Expected NPV in 1985 US$/acre, expected thinning age and expected harvesting age for Pinus sylvestris, interest rate = 3%, maximum harvest age = 65 years.

	N.P.V.	Exp.thin.age	Exp.harv.age
Faustmann with thinning	1407	32.0	40.4
Flexible management without thinning:			
$\sigma = 5$	1597	-	40.1
$\sigma = 7$	1733	-	38.3
$\sigma = 9$	1916	-	38.6
Flexible management with thinning:			
$\sigma = 5$	1718	19.8	45.9
$\sigma = 7$	1860	21.4	46.9
$\sigma = 9$	2053	20.5	46.2

The results in Table 11.1 indicate that the expected thinning age with flexible management is earlier than in the Faustmann model, while expected harvest age under flexible management is later. An explanation for these results is that flexible management increases the expected value of harvest revenues, which in turn, increases the importance of the pre-commercial aspect of thinning. The increased importance of pre-commercial thinning tends to reduce expected thinning age. As discussed in section 11.2, the effect of thinning on the expected harvest age is also consistent with expectations. Thinning increases the quality of the residual stand. This implies that the reservation path shifts upwards, and rotation length is increased.

As described in section 11.2, the *a priori* impact of increasing the spread of the stumpage price distribution on thinning reservation prices is ambiguous depending on whether the commercial or pre-commercial aspects of thinning dominate. The results presented in Table 11.1 provide support for this result. For an increase in standard deviation from 5 to 7, the impact of increasing stumpage price volatility on the commercial aspect of thinning is smaller than the impact on the pre-commercial aspect. That is, landowners tend to thin later to capture larger tinning volumes at the expense of stumpage quality at harvest. The reverse holds for an increase in standard deviation from 7 to 9.

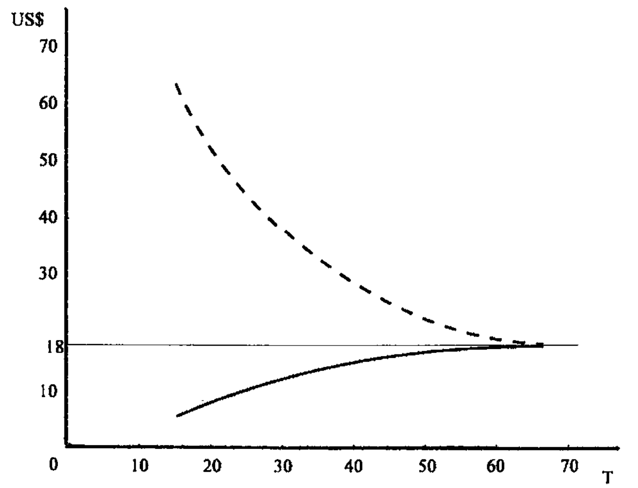

Fig.11.1: Reservation price paths for harvesting and thinning (for σ=7).

To examine the robustness of the results, we investigated the effects of varying the price-diameter relationship from thinning by varying the coefficient for *CT* in equation (11). The results, not reported in Table 11.1, suggest that increasing the coefficient for *CT* decreases expected NPV of bareland and increases expected thinning and harvest ages. The expected NPV of bareland decreases because the mean of the price distribution for every stumpage age is lower. Stumpage prices are lower because increasing the coefficient for *CT* "pivots" the price-diameter relationship downward. Also "pivoting" from increasing the coefficient increases the expected relative gains in prices from producing bigger trees. This increase in expected relative gains impacts both the commercial and pre-commercial aspects of thinning. The impact on the pre-commercial aspect is to encourage early thinning to produce bigger trees for final harvest. Conversely, the impact on the commercial aspect tends to postpone thinning, as landowners wait to realize higher expected thinning revenues. For our data, the latter effect dominates the former. For example with σ = 7, reducing the coefficient for *CT* from -0.3 to -0.6 and -0.9 resulted in new expected thinning ages of 24.9 and 27.8 years. Expected rotation ages also increased, to 47.6 and 49.7 years, respectively.

We also examined the quantitative impacts of changing the discount rate. The effects of raising r on NPV and rotation length are consistent with expectations. For example, with σ = 7, increasing r to 5% and 7% results in respective drops of NPV to US$395 and -US$107, and a reduction of the expected rotation age to respectively 41 years and 37 years. The effect of increasing the discount rate on the expected thinning age is more complex and depends upon the opposing impacts of pre-commercial and commercial aspects of thinning. The pre-commercial aspect becomes less important when the future is discounted more heavily, hence the incentive to thin at early ages is reduced. Also, for higher discount rates the incentive to thin for revenue will be increased.[19] That is, the landowner will attempt to move some of the harvest revenues forward to become thinning revenues. This tends to postpone thinning. For our data, the first effect dominates the latter. For discount rates of 5% and 7%, the expected thinning age decreases to 17.7 and to 15.8 years, respectively.

[19]Increasing the discount rate also shortens the rotation age, which would also tend to decrease thinning ages.

11.6 Conclusions

Although thinning of even-aged single species stands is standard silvicultural practice, there have been few formal treatments of thinning in the context of optimal forest management with fluctuating stumpage prices. In this chapter we derive new theoretical results from the inclusion of thinning in a flexible management model with stumpage prices that are independent random variables with stationary means. These results are supplemented by a numerical simulation using data on Dutch stumpage prices and a Dutch growth function that includes the pre-commercial aspect of thinning.

Some of the qualitative results obtained have direct analogues in optimal harvesting models without thinning. We found that reservation price paths for thinning and harvesting exist, and that the net present value of land and stumpage, and for relatively low discount rates most harvest reservation prices, increase with the spread of the stumpage price distribution.

Other qualitative results have no direct analogues when thinning is not permitted. We showed that, for every age of stumpage, the thinning reservation price is less than or equal to the harvest reservation price. Moreover, the reservation price paths converge, such that the probability of thinning reduces as the stand matures. The inclusion of thinning increases some reservation prices for harvesting, since the possibility of future thinnings increases both the opportunity cost of current harvesting and the benefits of current harvesting. As expected, the possibility of thinning increases the NPV of bareland. Finally, thinning reservation prices may either increase or decrease with an increase in the spread of the stumpage price distribution, depending on the magnitude of the commercial and pre-commercial aspects.

An important factor in explaining thinning behaviour is the relative size of the commercial and pre-commercial aspects of thinning. Typically, the pre-commercial aspect tends to accelerate thinning to concentrate as much aggregate growth on the remaining trees as possible, while the commercial aspect tends to a certain extent to postpone thinning such that thinned volume is greater. Shifting the spread of the stumpage price distribution, the interest rate or the price diameter relationship creates opposing incentives in the commercial and pre-commercial aspects, with the actual change in thinning behaviour becoming an empirical question.

The simulated numerical results provide an initial evaluation of the significance of the theoretical results. For our Dutch data we found that the results analogous to previous results in flexible management models without thinning continue to be important; that is, a reservation price strategy remains optimal and the increases in the net present value of land and stumpage are substantial and vary with the spread of the stumpage price distribution.

The numerical results for thinning are also of interest. Expected thinning age decreases sharply while expected harvest age increases under flexible management compared to Faustmann management due to an increase in the pre-commercial incentives to thin, from an increase in net present value of older stands. The harvest and thinning reservation prices converge with age, thus the probability of thinning approaches zero as the stand ages, which is consistent with standard practice. Although the gains in net present value of land and stumpage are relatively modest (about 7%), they may be large enough for many landowners to optimally choose thinning age. These gains would be larger if the landowner would thin more than once or also choose thinning intensity. The simulated numerical results also allow comparison of the imposing impacts of pre-commercial and commercial aspects of thinning that arise from varying the spread of the stumpage price distribution, the price-diameter relationship and the discount rate.

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Appendix 1:

Autocorrelation and partial autocorrelation function for pine prices 1975-1991

```
Autocorrelation function:

     Auto- Stand.
Lag  Corr.   Err. -1    -.75 -.5 -.25   0   .25  .5   .75   1   Box-Ljung  Prob

  1   .449   .228              .        o*********                  3.863   .04
  2  -.056   .220              .       *o        .                  3.929   .14
  3   .156   .212               .       o***     .                  4.467   .21
  4   .347   .204               .       o*******.                   7.350   .11
  5   .119   .195               .       o**      .                  7.724   .17
  6  -.181   .186                .  ****o      .                    8.669   .19
  7  -.228   .177                . *****o      .                   10.333   .17
  8  -.004   .167                .      *      .                   10.333   .24
  9   .064   .156                 .     o*    .                    10.500   .31
 10  -.211   .144                 . ****o     .                    12.630   .24
 11  -.331   .132                **.****o    .                     18.943   .06
 12  -.124   .118                  .  **o    .                     20.048   .06
 13  -.124   .102                   . **o   .                      21.518   .06
 14  -.243   .083                  **.**o  .                       30.021   .00

Plot Symbols:      Autocorrelations *     Two Standard Error Limits .

Partial Autocorrelation function:

   Pr-Aut- Stand.
Lag  Corr.   Err. -1    -.75 -.5 -.25   0   .25  .5   .75   1

  1   .449   .250              .        o********.
  2  -.323   .250              .  ******o        .
  3   .467   .250              .        o********.
  4  -.022   .250              .        *        .
  5  -.017   .250              .        *        .
  6  -.230   .250              .   *****o        .
  7  -.163   .250              .     ***o        .
  8   .122   .250              .        o**      .
  9  -.034   .250              .       *o        .
 10  -.139   .250              .     ***o        .
 11  -.056   .250              .       *o        .
 12  -.076   .250              .      **o        .
 13  -.265   .250              .   *****o        .
 14   .081   .250              .        o**      .
```

Appendix 2:

A. *Proof that harvest reservation prices are greater than thinning reservation prices, i.e., $A(k|x_{k-1}) > \alpha(k|x_{k-1})$.*

Proof by contradiction. If $\alpha(k|x_{k-1}) > A(k|x_{k-1})$, then there exist two prices such that harvesting is optimal at the lower price and thinning is optimal at the higher price. Formally, from equation (1) we know that if $\alpha(k|x_{k-1}) > A(k|x_{k-1})$, there exists b > 0 and $\hat{p}$, such that

$$\hat{p}(k)Y(k|x_{k-1}) + V(0) > \hat{p}(k)Z(k|x_{k-1}) + \beta V(k+1|x_{k-1}), \text{ and} \quad (A.1)$$

$$[\hat{p}(k)+b]Y(k|x_{k-1}) + V(0) < [\hat{p}(k)+b]Z(k|x_{k-1}) + \beta V(k+1|x_{k-1}). \quad (A.2)$$

Rearranging (A.1) and (A.2) provides:

$$\hat{p}(k)[Y(k|x_{k-1}) - Z(k|x_{k-1})] > \beta V(k+1|x_{k-1}) - V(0), \text{ and} \quad (A.3)$$

$$[\hat{p}(k)+b][Y(k|x_{k-1}) - Z(k|x_{k-1})] < \beta V(k+1|x_{k-1}) - V(0), \quad (A.4)$$

which implies

$$\hat{p}(k)[Y(k|x_{k-1})-Z(k|x_{k-1})] > [\hat{p}(k)+b][Y(k|x_{k-1})-Z(k|x_{k-1})]. \quad (A.5)$$

However, since both b and $[Y(k|x_{k-1})-Z(k|x_{k-1})] > 0$, then $\hat{p}(k)[Y(k|x_{k-1})-Z(k|x_{k-1})] > [\hat{p}(k)+b][Y(k|x_{k-1})-Z(k|x_{k-1})]$, and (A.5) is false. Therefore $\alpha(k|x_{k-1}) \leq A(k|x_{k-1})$.

B. Proof that for positive discount rate that the simulation algorithm converges to V*(0).

Using standard comparative statics methodology (Samuelson 1947, Silberberg 1990) we show that the iterative simulation procedure converges to $V^*(0)$. To model the iterative process, first-order conditions (5), (6) and (7) can be rewritten as:

$$\alpha(k|x_{k-1})Z(k|x_{k-1}) - \beta V(k+1|x_{k-1},0) + \beta V(k+1|x_{k-1},1) = 0, \quad \text{(5')}$$

$$A(k|x_{k-1})[Y(k|x_{k-1}) - Z(k|x_{k-1})] - \beta V(k+1|x_{k-1},1) + [V^*(0)+b_s] = 0 \quad \text{(6')}$$

$$[V*(0)+b_{s+1}] + - C - \sum_{i=1}^{N} B^i \sum_{j=1}^{2^{i-1}} \Phi(x_{i-1}^j) *$$
$$[\int_{A(i|x_{i-1}^j)}^{\infty} f(p)\,[p\,Y(i|x_{i-1}^j) + [V*(0)+b_s]]\,dp + \int_{\alpha(i|x_{i-1}^j)}^{A(i|x_{i-1}^j)} f(p)\,(p\,Z(i|x_{i-1}^j))\,dp], \quad \text{(7')}$$

where $[V^*(0) + b_s]$ is the net present value of bareland after s iterations, and the starting net present value of bareland for the $s+1$ iteration.

The iterative procedure converges to $V^*(0)$ for any tolerance, if the $\lim_{s\to\infty} b_s = 0$. We show for positive discount rates that $0 < db_{s+1}/db_s < 1$, which implies that $|b_{s+1}| < |b_s|$, and the $\lim_{s\to\infty} b_s = 0$.

To show the result using comparative statics methodology, we designate the $A(\ )$'s, $\alpha(\ )$'s and b_{s+1} as the endogenous variables and using the Implicit Function Theorem to define the exogenous variables as implicit functions of the exogenous parameters. Next, we substitute the implicit functions into (5'), (6') and (7') and totally differentiate the system with respect to b_s. Then we solve for db_{s+1}/db_s. The easiest way to perform these steps is to use Cramer's Rule.

Differentiating (5'), (6') and (7') with respect to $A()$'s, $\alpha()$'s and b_{s+1} provides:

$$J = \begin{matrix} \delta_1 & 0 & 0 & 0 & \dots & 0 & 0 & 0 & 0 & \dots & 0 \\ 0 & \delta_2^1 & 0 & 0 & \dots & 0 & 0 & 0 & 0 & \dots & 0 \\ 0 & 0 & \delta_2^2 & 0 & \dots & 0 & 0 & 0 & 0 & \dots & 0 \\ 0 & 0 & 0 & \delta_3^1 & \dots & 0 & 0 & 0 & 0 & \dots & 0 \\ . & . & . & . & \dots & . & . & . & . & \dots & . \\ . & . & . & . & \dots & . & . & . & . & \dots & . \\ . & . & . & . & \dots & . & . & . & . & \dots & . \\ 0 & 0 & 0 & 0 & \dots & \gamma_1 & 0 & 0 & 0 & \dots & 0 \\ 0 & 0 & 0 & 0 & \dots & 0 & \gamma_2^1 & 0 & 0 & \dots & 0 \\ 0 & 0 & 0 & 0 & \dots & 0 & 0 & \gamma_2^2 & 0 & \dots & 0 \\ 0 & 0 & 0 & 0 & \dots & 0 & 0 & 0 & \gamma_3^1 & \dots & 0 \\ . & . & . & . & \dots & . & . & . & . & \dots & . \\ . & . & . & . & \dots & . & . & . & . & \dots & . \\ . & . & . & . & \dots & . & . & . & . & \dots & . \\ 0 & 0 & 0 & 0 & \dots & 0 & 0 & 0 & 0 & \dots & 1 \end{matrix} \qquad \textbf{(B.1)}$$

where $\delta_k^j = Z(k|x_{k-1}^j)$ and $\gamma_k^j = Y(k|x_{k-1}^j) - Z(k|x_{k-1}^j)$, $j = 1, 2 \dots 2^{k-1}$, i.e. δ_k^j is the quantity thinned in year k given the j^{th} thinning record for age k-1, and γ_k^j is the difference between the quantity thinned amount thinned in year k given the j^{th} thinning record for age k-1.

The determinant of J equals the product of the elements of the primary diagonal, i.e.

$$|J| = \prod_{k=1}^{T} \prod_{j=1}^{2^{k-1}} Z(k|x_k^j)\,[Y(k|x_k^j) - Z(k|x_k^j)] > 0. \qquad \textbf{(B.2)}$$

(Since the $|J|$ is non-zero the implicit functions defined earlier using the Implicit Function Theorem are valid.)

Partially, differentiating (5'), (6') and (7') with respect to b_s provides:

$$\begin{aligned}
\frac{\partial (5')}{\partial b_s} &= 0 \\
\frac{\partial (6')}{\partial b_s} &= 1 \quad \textit{and} \\
\frac{\partial (7')}{\partial b_s} &= -\sum_{i=1}^{N} B^i \sum_{j=1}^{2^{i-1}} \Phi(x_{i-1}^j) * [\int_{A(i|x_{i-1}^j)}^{\infty} f(p)\,(p\,Y(i|x_{i-1}^j)\,dp
\end{aligned} \tag{B.3}$$

Substituting the negatives of the partial derivatives of (B.3) into the last column of J provides:

$$J = \begin{matrix}
\delta_1 & 0 & 0 & 0 & \cdots & 0 & 0 & 0 & 0 & \cdots & 0 \\
0 & \delta_2^1 & 0 & 0 & \cdots & 0 & 0 & 0 & 0 & \cdots & 0 \\
0 & 0 & \delta_2^2 & 0 & \cdots & 0 & 0 & 0 & 0 & \cdots & 0 \\
0 & 0 & 0 & \delta_3^1 & \cdots & 0 & 0 & 0 & 0 & \cdots & 0 \\
. & . & . & . & \cdots & . & . & . & . & \cdots & . \\
. & . & . & . & \cdots & . & . & . & . & \cdots & . \\
. & . & . & . & \cdots & . & . & . & . & \cdots & . \\
0 & 0 & 0 & 0 & \cdots & \gamma_1 & 0 & 0 & 0 & \cdots & -1 \\
0 & 0 & 0 & 0 & \cdots & 0 & \gamma_2^1 & 0 & 0 & \cdots & -1 \\
0 & 0 & 0 & 0 & \cdots & 0 & 0 & \gamma_2^2 & 0 & \cdots & -1 \\
0 & 0 & 0 & 0 & \cdots & 0 & 0 & 0 & \gamma_3^1 & \cdots & -1 \\
. & . & . & . & \cdots & . & . & . & . & \cdots & . \\
. & . & . & . & \cdots & . & . & . & . & \cdots & . \\
. & . & . & . & \cdots & . & . & . & . & \cdots & . \\
0 & 0 & 0 & 0 & \cdots & 0 & 0 & 0 & 0 & \cdots & -\frac{\partial (7')}{\partial b_s}
\end{matrix} \tag{B.4}$$

Solving for db_{s+1}/d_s provides,

$$\frac{db_{s+1}}{db_s} = \frac{|J_b|}{|J|} = -\frac{\partial(7')}{\partial b_s} = \tag{B.5}$$

$$\sum_{i=1}^{N} B^i \sum_{j=1}^{2^{i-1}} \Phi(x_{i-1}^j) * [\int_{A(i|x_{i-1}^j)}^{\infty} f(p)(p\,Y(i|x_{i-1}^j)\,dp$$

Note that the right-hand side of (B.5) is bounded by 0 and 1. (If the interest rate equals 0, then $\beta = 1$ and the right-hand side of (B.5) equals 1. As the interest rate approaches ∞, then β approaches 0 and right-hand side of (B.5) approaches 0). Since for positive interest rates, $0 < db_{s+1}/db_s < 1$, then for positive interest rates, $|b_{s+1}| < |b_s|$, which implies that the $\lim_{s \to \infty} b_s = 0$, and for any arbitrary starting value the iterative procedure converges to $V^*(0)$ as s approaches ∞.

CHAPTER 12

DISCUSSION, SUMMARY AND CONCLUSIONS

Economics of renewable resources provides a broad field for research, and in this thesis I have tried to look at a variety of issues. The field allows researchers to become engaged in a wide range of topics, each with its own peculiarities, problems and interesting features. Depending on the issue, models should try to capture, for example, non-market values (as with whales and elephants) or the transformation of different state variables (as with primary and secondary forests). Models can be constructed to maximize social welfare, profits or a political preference function. In addition to the variety in topics, the research field invites analysts to employ many different techniques to study the various aspects of resource economics. There is certainly scope for more elaborate future work, especially with respect to the institutional and public choice aspects of resource economics. Also, rigorous incorporation of non-market values, uncertainty and "ecological thresholds" deserves more attention.

Since the separate chapters that constitute the core of this thesis include summaries of the main results and end with concluding paragraphs, this chapter can be brief. Some of the main findings will be summarized in a more general tone than before, and where possible I will try to distil recommendations for future research.

In chapters 1 and 2 of this thesis I provided an introduction to the economics of natural resource management, based on a literature survey. The introduction starts with recognizing that two conceptually distinct themes are central in this field: sustainability and efficiency. Sustainability relates to intertemporal equity, and may be extended to incorporate ethical considerations concerning other species than humans. Although sustainable use is perhaps the most important issue of resource- and environmental management (potentially much more relevant than efficiency, which is not much more than a "no-waste criterion"), the literature suffers from lack of uniformity in its interpretation. The question what sustainability exactly amounts to appears to depend for an important part on the analysts (subjective) opinion about the current generation's responsibility for future generations and their (subjective) estimation of future substitution possibilities between different factors of production and technological change. Obviously, there are "objective" limits to these subjective beliefs in the form of thermodynamic laws.

Efficiency, on the other hand, simply defines intertemporal exploitation paths that maximize the present value of welfare or profits, depending on the context chosen. [Possibly, requirements with respect to sustainability can be incorporated into standard efficiency models by adding an additional (subjective) constraint, or by including an additional argument in the objective function that somehow measures future welfare.] In this thesis I have focused on efficient solutions to various resource management problems.

Exploiting a stock of an exhaustible or renewable resource implies that possibilities for future exploitation are affected. A key result in resource economics due to Hotelling (1931) is that resource owners should be indifferent to current and future exploitation: an intertemporal exploitation path is defined as efficient if it is not possible to increase welfare (profits) by shifting exploitation from one period to another. The standard Hotelling result for an exhaustible resource, valid under a set of quite stringent assumptions, is that along an efficient exploitation path the rate of growth of resource prices should be equal to the discount rate. When stock effects exist, such as exploitation costs or non-use values dependent on the size of the stock *in situ*, or when exploiting firms have market power, the standard rule should be adjusted so that exploitation is more conservative, and a more gently-sloped price path located at a "higher level" is the result.

Renewable resource economics combines economic reasoning with biological models of growth and reproduction. Typically, single-aged forestry models have focused on optimum rotation ages in the tradition of Faustmann (1859), and in fishery models the search is often for efficient steady states in the tradition of Clark (1976). Both types of models are briefly discussed in chapter 2. At the optimum rotation age for a single-aged stand of trees, the land owner is indifferent to harvesting the trees or allowing an extra year of growth. Similarly, a stock of fish is at its efficient size when the resource owner is indifferent in harvesting an additional unit of fish or leaving them to swim and grow. In models of renewable resource economics, harvesting is typically interpreted as disinvestment, whereas refraining from harvesting and allowing the stock to grow and reproduce is considered investing in the stock.

The approach-dynamics in fishery problems have been studied less extensively, probably due to the mathematical complexities involved, but it has been demonstrated in the literature that a temporary reduction of a population below its equilibrium population may be efficient under the condition of irreversible investments in harvest capacity. When an interior solution of the maximization problem exists, eventually the equilibrium population is reached (with the so-called bang-bang models) or very closely approximated (with non-linear models). However, there is no guarantee that (positive) equilibrium stocks exist and it may be "efficient" to eradicate the entire population, even for a single rational resource owner (for some authors this is sufficient evidence of the folly of striving for efficient outcomes). The intuition is that maintaining positive populations (especially of slowly growing species) may be no competitive investment for a rational owner.

Apart from deliberate extinction, which might occur when investment opportunities elsewhere in the economy are favourable, fisheries may be threatened by lack of well-defined property rights. Under the condition that resources are available to everyone, no single firm will be willing to invest in its survival because the benefits of this investment will almost certainly accrue to somebody else. Under this condition, harvesting takes place at full force and depending on the characteristics of the species and the available technology, inadvertent extinction may be the result. Implementation of the exclusive 200-mile fishing zones has enabled national and supranational policymakers to formulate policies aimed at preventing this phenomenon. Various policy instruments are discussed in chapter 2.

However, as demonstrated in the recent past, conflicts over resource stocks at the boundary of these exclusive zones will probably be important in the near future (think of the troubles between Spain and Canada in 1995). Also, the terms of fishing contracts in fishing zones of many developing countries will be subject of renegotiation [see Lingsma (1996) for a popular, mostly non-economic, discussion], so it is still too early to conclude that problems related to property rights are a subject of the past. Finally, it should be noted that even under a system of well-defined property rights and with fast growing species that could easily compete with investments elsewhere in the economy, interest groups may put pressure on governments that cause deviations from "socially optimal" harvesting and threaten populations. This is elaborated upon in chapter 8.

Chapters 3 and 4 of this thesis are about tropical deforestation. In section 2.6.2 we have examined the popular misconception that trade in tropical timber products is a major direct cause of deforestation. It is easily verified that little support for this claim is found: some quick and dirty calculations show that less than one percent of tropical deforestation can be directly attributed to the timber trade. On average, commercial logging for domestic and export markets amounts (directly) to about 10% of tropical deforestation. As is well documented, the major cause of deforestation is agricultural conversion. The nature of this conversion process can take many different forms (e.g., clearing land for subsistence agriculture or for large-scale cattle raising for export markets), but the results are often identical: potential loss of biodiversity, nature values and regulatory functions of the forest ecosystem, displacement of indigenous people and squandering of scarce assets. On the other hand, it should be recognized that there is still much uncertainty about the size of the stock of tropical forest that maximizes the present value of welfare for the national (or global) community [for an exception at national level, see Ehui and Hertel (1989)], hence how much forest should be conserved from an economic viewpoint is unclear.

In the first part of chapter 3, the relation between (the threat of) agricultural conversion on the one hand and logging and forest conservation on the other hand has been worked out in more detail. The reverse relation, i.e., the impact of the logging sector on agricultural conversion is well studied and described [see e.g. Pearce and Brown (1994)]. Of crucial importance in this chapter is the difference between mature, undisturbed primary forests and secondary forests that have been subject to selective logging and hence display net growth. As mentioned above, agricultural conversion by (shifting) cultivators on tropical forest concessions (a process called encroachment) is generally considered a primary cause of deforestation in many areas. Therefore, in much of the literature on tropical deforestation, authors propose measures to reduce the intensity or scale of encroachment. Examples of such policies could include investments in education, agricultural reform and reduction of population growth.

Without minimizing the potential values of such programs, there is another side to encroachment, that we have elaborated upon in chapter 3. Empirical support exists for the hypothesis that encroachment is confined to accessible logged-over (secondary) forest areas. Therefore we argue that encroachment has similar effects as a kind of property tax on owning secondary forests: it induces rational forest owners to cut back on their possession of this type of forests. With damage due to encroachment positively related to the size of the secondary forest area (i.e., damage due to agricultural conversion increases as the area of accessible forest increases), concessionaires will respond to the threat of agricultural conversion by reducing the rate at which primary forests are harvested. The net effect of encroachment on conservation benefits is therefore theoretically ambiguous: losses in nature values from enhanced conversion of secondary forests may be balanced by a larger area of (more highly valued) primary forest. However, the necessary conditions for this "braking- or inhibiting effect" to occur (e.g., loggers should consider successive rounds of logging, hence concession rights should be defined for a long enough period) are often violated in practice [e.g. Repetto and Gillis (1988)]. Future research in this field could include modelling the effect of encroachment on optimum encroachment prevention investments undertaken by logging firms, and the net effect on deforestation that results. Also, empirical research could be directed at the underlying causes of regional migration and agricultural conversion. Currently, it is hardly understood whether push factors (land scarcity in the traditional agricultural areas), pull factors (higher expected income in forest areas) or considerations relating to risk aversion [think of pooling the income of a geographically dispersed family (through remittances) with uncorrelated fluctuations in harvest levels in different production areas, as opposed to pooling income with your neighbour who is subject to similar climatic conditions] dominate migration decisions. Obviously, these different underlying causes of migration would call for very different policy measures to combat deforestation.

In the last part of chapter 3 we examine the often heard claim that excessive deforestation is caused by loggers applying a discount rate that is too high (measured with the social discount rate as a benchmark). Such discount rates can be caused by risk premiums or high opportunity costs of capital. We demonstrate that with restrictions on supply, for instance in the context of a tropical timber concession contract, the general conclusion about adverse effects of discounting on resource conservation no longer holds in a model that distinguishes between primary and secondary forests. If the concession period for logging firms is exogenously determined and fixed, and when we recognize that depleting a stock of primary forest automatically implies building a stock of secondary forest, then the effect of high discount rates on the stock of primary forest is ambiguous. Thus we obtain the paradoxical result that for a certain range of values for the discount rate, an increase in the discount rate actually postpones depletion of primary forests. The intuition behind this result is that high discount rates reduce the marginal benefits of converting primary forest into a productive asset, which slows down the optimal extraction rate of the stock of virgin forest.

In chapter 4 we explore the robustness of the standard result that international transfers promote conservation of tropical forests. For this purpose, we relax the conventional (but restrictive) assumptions of deterministic prices and a risk-neutral forest owner. The standard result, lucidly described by Barbier and Rauscher (1994), is based on the observation that governments have to balance non-use values of forest protection and consumption of certain commodities, which are partly paid by selling timber. With constant prices, international transfers enable developing countries to increase consumption. Under standard assumptions, this results in a reduction of the marginal benefit of consumption which, in equilibrium, corresponds with a lower marginal utility of forest conservation. The way to achieve a reduction in marginal utility of conservation is through expansion of the forest area. Expansion of the forest area is possible in the Barbier and Rauscher model because they assume a logistic growth function for the forest stock (in contrast to Ehui and Hertel, mentioned above, who model deforestation as an irreversible process). However, with fluctuating timber prices and risk-averse resource owners (of the DARA type), we demonstrate that international capital transfers to developing countries to reduce tropical deforestation are a less effective instrument to combat deforestation than previously thought, or perhaps not an effective instrument at all. The reason is that because of increased wealth, governments become less prudent in their harvesting decisions. This effect weakens the above-mentioned effect. In addition, under conditions of (mean preserving) stochastic prices, harvesting is less attractive for risk-averse resource owners than extraction with constant prices. This implies that the steady-state of the forest stock under stochastic prices exceeds the steady state stock with deterministic prices. This implies that the marginal utility of additional conservation [achieved through provision of (costly) transfers] is lower with stochastic prices. Combining these effects leads us to the conclusion that the attractiveness of international transfers as an instrument to combat deforestation with stochastic prices is reduced.

Chapters 5 and 6 are concerned with (the economic rationale of) strict species conservation, and reveal results that may be hard for die-hard conservationists to accept. In chapter 6, we address whether a total ban on commercial harvesting of minke whales makes economic sense, and in section 5 we explore the impact of the ivory trade ban on elephant conservation. Results with respect to the latter are inconclusive. Banning the trade in ivory affects investment and disinvestments decisions of host countries, and we have theoretically shown that this could result in either an increase or decrease in optimum size of elephant populations. Determinants of the shifts in population size are the discount rate applied by the government, agricultural damage (or opportunity costs) of having a stock of elephants and the contribution of elephants to tourism benefits. While the optimum population size with trade in ivory is a declining function of the discount rate (for low rates, the efficient population should be large and vice versa, which is easily verified with standard fisheries models described in chapter 2 of this thesis), the efficient population in the absence of trade reduces to a simple static model of balancing marginal recreation benefits and marginal agricultural damages.

Using Kenyan data, we infer that the apparent success of the trade ban in protecting elephants may not be permanent, because the critical discount rate (i.e., the rate that makes the optimum population with and without trade of equal size) is pretty close to what can be considered an approximation of the social discount rate. The critical rate is probably (somewhat) lower than the discount rate currently applied by governments when deciding on investing in elephant conservation, but this condition is unlikely to sustain indefinitely. When per capita income in African countries rises it is generally believed that the (social) discount rate falls, and eventually the critical rate may exceed the discount rate. Then, elephant conservationists should try to lobby to have the trade ban lifted if they want to have more elephants around. Some caveats of the model (especially the absence of poaching is a crude simplification) are discussed in chapter 5. Future research in this field should try to incorporate poaching into formal models.

The purpose of chapter 6 was to analyze whether the current (commercial) harvest moratorium for minke whales was efficient. Although we do not claim that this is per se the most interesting aspect of the annually returning whaling discussion (equity issues may be more relevant for policymakers, especially the design of mechanisms to transfer public good benefits from non-whaling countries to countries that would like to resume whaling) we think that it sheds some light on society's preferences. In addition, comparing marginal benefits and marginal costs (of harvesting whales, in this particular case) is not much less than a professional aberration of resource economists, and probably most economists in other fields alike.

(Minke) whales have been called mixed goods because they combine features of private goods (meat and blubber) with features of public goods (existence values and so on). We demonstrate, not surprisingly, that the existence of public non-use values increases the size of the efficient stock beyond the optimal level in the absence of such values. Solving a dynamic optimization model, we conclude that the current stock of minke whales is probably suboptimally small. An uncertain factor is the marginal value of preservation. For most scenarios analysed we conclude that the current stock is too small, but it is realized that this result is obtained under the strong assumption of constant marginal value of preservation. In other words, when society cares less about the 100,000th whale preserved than about number 500, it could well be the case that the current stock of whales is excessively large and should be subjected to harvesting. More work in the field of valuation is needed to address this issue.

As mentioned above, a major objective of chapter 6 was to verify the economic rationale of the whaling moratorium. For the dynamic model, this implies numerically solving the optimal approach paths (the so-called stable arm) to the steady states of the various scenarios. We conclude that the (temporary) moratorium is an efficient policy if maximizing the present value of welfare is the objective of decision makers, and if the restrictive assumption of constant marginal utility of conservation holds.

An interesting result of chapter 6 is that most of the conclusions obtained for the dynamic model are reversed when the management problem is approximated by a series of static models, as proposed by Kuronuma and Tisdell (1993). With the static model, the current benefits of harvesting and conservation are compared *at the margin*. This model incorrectly suggests that the ban on commercial harvesting may be inefficient. We conclude that one should be careful to infer policy recommendations from static models for problems that are inherently dynamic.

The step from whale management to fisheries management is easy to make for an applied economist. For a long time, the major impediment to rational resource use was similar for the whaling and fishing sectors: open access to the resource implied that no rational firm was willing to invest in stock survival. In the post-Second World War era this resulted in massive depletion of fish and whale stocks alike. In chapter 7 we have investigated whether the responses of the European Union to overcome property rights failure have had a positive effect on fishery management or not. More specifically, we have tried to establish if policy implementation that followed the declaration of exclusive fishing zones reversed (to a certain extent) the well-described process of rent dissipation.

In chapter 7 we have presented a simple time-series analysis of German fish prices to evaluate rent dissipation. The method employed in this chapter consisted of estimating a model on the basis of a time series of prices for the "open access period" and to estimate studentized residuals for the following "common property period". The results for all six commercial species selected indicated that studentized residuals were significant for a series of observations. We have interpreted this as an indication that scarcity rent has returned as a component of prices, which would imply that establishing exclusive zones and successive development of fishery policies had a positive impact on management of fish stocks in Europe.

As argued by Dupont (1991), measuring rent dissipation requires information about prices and (marginal) harvesting costs over time, where rent is defined as the difference between the two. In chapter 7 we have taken a short cut by omitting an explicit analysis of the latter part. Instead, we have tried to verbally address the complexities of cost development over time. We feel strengthened by the empirical observation that in the late 1980s and early 1990s the profitability of fishing increased. This implies that (marginal) costs have not kept pace with rising prices, which is consistent with our conclusions. Yet, we agree that the European Fisheries policies give rise to much more work for economists, and we consider this chapter a modest first step. Future work could include a comparison of rent return between different member states with different national policies. To make this a fruitful exercise, the harvest side of the problem should be modelled explicitly. More in general, it seems that multi-species harvesting and simultaneous modelling of exploitation and pollution problems provide interesting departure points for analysis.

The conclusion that fisheries policies have contributed to more efficient management of fish resources is perhaps encouraging, but there is widespread evidence that current harvesting is still far from optimal. Current stocks of many commercial species are below their economic (and biological) optimums. However, reform of the sector to ensure an increase in future income (e.g. Schmidt 1993) will most probably meet resistance from the fishermen.

We have tried to come up with a way to cut quotas without reducing utility of fishers, as described in chapter 9. By establishing a futures market of tradeable fishing rights, uncertainty for fishers buying, selling, leasing or simply owning these rights can be greatly reduced. This would make a risk-averse fisher better off, which offers scope for a reduction in the size of his quota to bring him back to his original "utility level". Yet, translating these academic results into a workable format may create problems, as the relevance of concepts such as "utility level" may be questioned outside the realm of theoretical economics. (At least, I must admit that I do not feel tempted to explain the concept of a utility-neutral quota cut to an audience of the fishers concerned.)

Related to the previous remark is the following: An aspect that has been somewhat overlooked in theoretical modelling of resource management, in my opinion, is the public choice side of decision making. It seems that every resource economists is able to crank out theoretical models that maximize welfare for society, but very few have actually attempted to model government decision making in detail. Yet, there is much complaining about suboptimal fisheries management (and excessive deforestation, to which the same remark applies). These decision makers just do not seem to listen! It seems to me that a lot of the theoretical work (including most of the work in this thesis) may be of little or no value to policymakers. Not only because some important variables are omitted in these models for convenience, but also because the postulated objective function for the social planner ("*maximize present value of welfare*") may be unrealistic. At least, public choice theories have clearly indicated that there may be more to policy-making than simply doing what is best for the people at large.

In chapter 8 we have tried to explicitly incorporate some of the concerns of public choice theorists into a simple model. We have distinguished three different groups in society (consumers, labourers and vessel owners) with different objectives. For example, consumers strive for low fish prices, labourers in the sector want to earn high wages and vessel owners lobby for large profits. Depending on the quantity of pressure they are able to generate (which is not explicitly modelled in chapter 8, but will be included in future research), they can influence government policies with respect to fish resources. In our model, decision making is modelled as a kind of "political market", where participants pay with pressure in order to purchase preferred policies. It is recognized that this picture is much more complete when the pressure generation process is included in the model and endogenous. In the current specification we merely develop a theoretical model and consider the effects of possible (exogenous) distributions of power or political force among the social groups on "optimal" stock size pursued by political planners. It turns out that the political stock may be either larger or smaller than the (standard) efficient stock, which may explain *over-harvesting*. Also, we have found that the optimal solution may be difficult to attain, and that specific combinations of policy instruments are required to end up in a steady state.

Chapters 10 and 11 are based on forestry models with uncertainty. In the latter chapter, uncertainty arises from stochastic stumpage prices with a known mean and standard deviation. We build on path-breaking research in this field by Lohmander (1987) and Brazee and Mendelsohn (1988) by including thinning in the now standard analysis of reservation prices. Thinning is a standard silvicultural treatment in large parts of the world, and it is strange that so far nobody has included this aspect before. [Although I should add that writing this chapter was a painstaking and sometimes tedious job, and that the marginal benefits in terms of scientific insight are relatively few. Perhaps we can trust (forest) economists to recognize these high-marginal-cost/low-marginal-benefit study areas in an early phase and avoid them where possible]. Central to the issue of strategic thinning is the balancing of commercial and pre-commercial aspects: thinning yields immediate revenues and increases the quality of the residual stand. Among other things, we find that the reservation price paths for thinning and harvesting gradually approach each other (obviously with the former always below the latter) and that inclusion of thinning significantly increases profits for a strategically behaving forest owner.

Chapter 10 is in many ways different from the stochastic price problem just described. It does not build in a thorough but straightforward manner on previous work, but in my opinion is more like a digression from conventional mainstream economics. This chapter is about land allocation on Vancouver Island, a heavily debated subject with a host of different interest groups involved. Preferences with respect to what the land should provide are often conflicting and stated in imprecise form. Moreover, a lot of the technical information that is needed to support modelling, for example, yield functions but also the recreational and preservation values attached to different forest types, is lacking. We have employed fuzzy logic to deal with both types of uncertainty. For this purpose, we have combined objective information and our best professional judgement to specify central values for imprecise parameters (or realizations that provide total satisfaction for objectives), spreads and membership functions [for the basis of fuzzy logic as we employed it, see Zimmerman (1991)].

Given the nature of the process that is to be modelled, our conclusion is that the fuzzy approaches can be judged a distinct improvement over the traditional approach of constrained maximization of net present value. The resulting land use allocation with the fuzzy models points much more in the direction of a "compromise" solution (i.e., with integrated forest management), as was explicitly sought for in the decision-making process, than the conventional model. As a side benefit, from varying the realizations of the imprecise coefficients by moving the level of the α-cut, we have obtained information about what kind of data are crucial to achieve robust solutions. Robust solutions are land use allocations that need no drastic shifts from one land use option to another in light of small changes in parameter values. Rather surprisingly, it turns out that information with respect to timber benefits are a weaker link in this respect than the data on non-market values we have employed.

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SAMENVATTING

In dit proefschrift neem ik een aantal onderwerpen die betrekking hebben op de economische theorie van beheer van natuurlijke hulpbronnen onder de loep. Voorbeelden van natuurlijke hulpbronnen zijn visvoorraden, bossen en olie. Het onderzoeksveld is breed en biedt keuze uit een scala aan onderwerpen, ieder met zijn eigen problemen, eigenaardigheden en interessante aspecten. Aangezien de afzonderlijke hoofdstukken van dit proefschrift afgesloten worden met een concluderende en samenvattende sectie, zal ik deze samenvatting kort houden.

De hoofdstukken 1 en 2 betreffen een inleidend literatuuronderzoek. In hoofdstuk 1 wordt het onderzoeksveld afgebakend en worden enkele basisbegrippen besproken. Twee centrale thema's zijn "duurzaamheid" en "efficientie". Duurzaamheid heeft betrekking op gelijke toegang tot natuurlijke hulpbronnen door verschillende generaties. Efficiëntie betekent het (intertemporeel) maximaliseren van een doelfunctie. Het maximaliseren van "de winst" of "het nut voor de samenleving" zijn veel gebruikte voorbeelden van dergelijke functies. Na de *Earth summit* in Rio de Janeiro in 1992 zou "duurzaamheid" wellicht het belangrijkste thema van de twee dienen te zijn, maar er bestaat in de literatuur veel verschil van inzicht over de exacte interpretatie van dit begrip. De interpretatie is subjectief en onder andere afhankelijk van de inschatting van de toekomstige substitutie-mogelijkheden tussen de verschillende productiefactoren (bijvoorbeeld tussen natuurlijk en fysiek kapitaal), en van de mogelijkheden die in de toekomst door technologische vooruitgang geboden zullen worden.

De interpretatie van efficiëntie is helder, en de *mainstream* van de neoklassieke economen is traditioneel gericht op dit thema. (Van meer recente aard zijn modellen waarin de beide thema's gecombineerd worden.) De meeste hoofdstukken in dit proefschrift gaan in de eerste plaats over het efficiënt gebruik van (vernieuwbare) natuurlijke hulpbronnen.

Het exploiteren van een natuurlijke hulpbron heeft invloed op de mogelijkheden voor toekomstig gebruik van deze voorraad. In geval van mijnbouw, bijvoorbeeld, gaat de huidige extractie van ertsen ten koste van grondstofwinning in de toekomst. Dit impliceert dat bij het delven van een eenheid erts, in aanvulling op de pure extractiekosten, rekening moet worden gehouden met de zogenaamde *opportunity costs*. In dit verband wordt met opportunity costs bedoeld de mogelijke baten die de eenheid erts in de toekomst had kunnen opleveren. Deze "extra" kosten verlagen de optimale hoeveelheid die geëxploiteerd dient te worden. Een belangrijke regel voor efficiënt gebruik van een niet-vernieuwbare hulpbron, zoals olie of steenkool, is dat onder bepaalde voorwaarden de groeivoet van de zogenaamde *rent* van de hulpbron (gedefinieerd als het verschil tussen de prijs en de marginale exploitatiekosten) gelijk moet zijn aan de interestvoet. Als deze regel opgaat zijn eigenaren van een hulpbron indifferent tussen huidige en toekomstige extractie: het rendement van het aanhouden van een voorraad van een bepaalde hulpbron is even hoog als het rendement van exploiteren en het verdiende geld vervolgens elders investeren of op de bank zetten. Dit is de bekende Hotelling regel.

Voor efficiënt gebruik van hulpbronnen die zichzelf binnen een redelijke termijn kunnen vernieuwen, zoals een voorraad vis of een bos, moet de Hotelling regel aangepast worden. In de economie van dit soort hulpbronnen worden economische principes gecombineerd met biologische groeimodellen. Met behulp van bosbouwmodellen kan dan bijvoorbeeld de optimale omlooptijd van een gelijkjarig bos berekend worden. Met behulp van visserijmodellen wordt de optimale visstand in een bepaald gebied berekend. Het afzien van oogsten, bijvoorbeeld door een hoeveelheid bomen nog een jaar te laten groeien of door de jaarlijkse vangst van een bepaalde vissoort te verlagen, kan geïnterpreteerd worden als "investeren" in de hulpbron. Omgekeerd geldt dat oogsten gelijk staat aan desinvesteren. In hoofdstuk 2 van dit proefschrift wordt door middel van een literatuurstudie ingegaan op deze materie en op verwante zaken als eigendomsrechten (immers, wie wil investeren in een hulpbron als niet duidelijk is of de toekomstige baten toevallen aan hemzelf?) en de rol van overheidsbeleid.

Hoofdstuk 3 en 4 zijn gewijd aan tropische ontbossing. Het *directe* verband tussen de commerciële kap van tropisch hardhout en ontbossing is uiterst zwak. Een veel belangrijkere factor is het "omzetten" van bossen in landbouwgrond (ongeveer 80% van alle ontbossing wordt veroorzaakt omdat ruimte voor landbouw, inclusief veehouderij, gemaakt moet worden). Zoals veel bosbouwers en milieubeschermers benadrukken bestaat er echter wel een sterk *indirect* verband tussen commerciële, selectieve houtkap en oprukkende landbouw. De bosbouwsector zorg namelijk voor het aanleggen van een infrastructuur in gebieden waar (selectief) gekapt is. Dit maakt het bedrijven van landbouw in deze gebieden aantrekkelijker.

Een veel gehoorde beleidsaanbeveling is dat boeren uit de bossen geweerd moeten worden, bij voorkeur door een algemeen ontwikkelingsbeleid gericht op het tegengaan van te snelle bevolkingsgroei, het bevorderen van alfabetisering en het hervormen van de landbouwsector. Ongetwijfeld wordt hiervoor met de beste bedoelingen gepleit. Dit proefschrift toont echter aan dat dit beleid niet altijd bevorderlijk is voor natuurbehoud.

In hoofdstuk 3 wordt de relatie tussen de (dreiging van) oprukkende boeren en het kaptempo van bosbouwers onderzocht. Omdat, zoals vermeld, conversie van bossen in landbouwgrond met name voorkomt in selectief gekapt bos, kunnen bosbouwers met kapconcessies voor meerdere jaren de schade door oprukkende landbouwers beïnvloeden door hun kapbeslissingen ("*in welk tempo zet ik mijn ongestoorde, primaire bos om in makkelijk toegankelijk, secundair bos?*") te veranderen. Het verminderen van schade zal overwogen worden indien het bosbouwers is toegestaan na verloop van tijd terug te keren naar het opengelegde gebied voor aanvullende kap. Met een model waar schade aan het bosbestand van een houtbedrijf positief gerelateerd is aan de omvang van het secundaire bos laten we zien dat de dreiging van oprukkende boeren twee effecten heeft. Aan de ene kant zullen bosbouwers de kap in toegankelijke bossen intensiveren om de brandende boeren vóór te zijn. Een andere reactie is het vertragen van de omzetting van ongestoorde (en ontoegankelijke) primaire bossen in secundaire bossen. We concluderen dat de dreiging van landbouwers leidt tot minder secundair en meer primair bos. Afhankelijk van de maatschappelijke waardering voor deze verschillende bossystemen kan dit uit oogpunt van natuurbehoud een verbetering of een verslechtering inhouden.

Het huidige tempo van ontbossing is volgens velen te hoog. Een mogelijke verklaring is dat kapbedrijven een te hoge rentevoet hanteren bij het beslissen over de spreiding van kapactiviteiten over de tijd. Immers, veelal wordt verondersteld dat hogere discontovoeten (implicerend dat relatief meer belang wordt gehecht aan huidige consumptie dan aan toekomstige consumptie) ondubbelzinnig negatief uitpakken voor natuurbescherming. Een tweede doelstelling van hoofdstuk 3 is te onderzoeken of dit correct is in de context van een model waarin onderscheid gemaakt wordt tussen primaire en secundaire bossen en waarin een winstmaximaliserend bedrijf geconfronteerd wordt met een overheid die bepaalde eisen stelt. We laten zien dat hoge rentevoeten niet noodzakelijkerwijs versnelde kap uitlokken. Het selectief kappen van primair bos betekent namelijk automatisch dat secundair bos gecreëerd wordt. Bij optimaal bosbeheer wordt de winst door kap in beide typen bos in de afweging betrokken. We laten zien dat hoge discontovoeten de baten van het omzetten van primair bos in secundair bos verlagen. Daarmee wordt het tempo vertraagd waarin deze primaire bossen dienen te worden gekapt om de winst te maximaliseren.

In hoofdstuk 4 onderzoeken we of het verschaffen van ontwikkelingsgeld aan ontwikkelingslanden een efficiënt instrument is om tropische ontbossing af te remmen. Op basis van een model waar de overheid van een Derde Wereldland de baten van bosbeheer maximaliseert, in dit geval opbrengsten uit verkoopbaar hout en niet-gebruikswaarden gekoppeld aan bosbescherming, is in het verleden geconcludeerd dat ontwikkelingshulp leidt tot extra bosbescherming. De redenering luidt als volgt: extra geld door middel van internationale transfers leidt tot meer consumptie in het Derde Wereldland, zodat de marginale baten van consumptie zullen dalen. Om het evenwicht te herstellen moeten de marginale baten van bosbescherming ook dalen, hetgeen alleen bereikt kan worden door het bosareaal uit te breiden. We hebben dit model uitgebreid en realistischer gemaakt door een risico-mijdende overheid en onzekerheid met betrekking tot toekomstige houtprijzen te veronderstellen. Uiteraard leidt deze uitbreiding normaliter tot afremmen van de kapinspanning. In aanvulling op dit effect hebben internationale donaties in het uitgebreide model een tweede effect: de overheid wordt door het extra geld minder risico-mijdend en laat zich in mindere mate door de onzekere prijzen afremmen om bos te kappen. De conclusie is dat de effectiviteit van internationale transfers als instrument om bij te dragen tot bosbescherming in het verleden is overschat.

In hoofdstuk 5 en 6 behandelen we enkele economische achtergronden van het beschermen van bepaalde diersoorten. In hoofdstuk 5 staat het verbod op de handel in ivoor centraal. Met de bedoeling om olifanten te beschermen is hiertoe, na enkele decennia van grootschalige olifantenslacht, besloten aan het einde van de jaren '80. Tot op heden heeft dit beleid bijgedragen aan herstel van olifantenpopulaties. Met een eenvoudig economisch model hebben we onderzocht of een handelsverbod altijd dit effect zal blijven hebben. Dit is waarschijnlijk niet het geval. Voor een overheid levert het beheren van een populatie levende olifanten verschillende baten op: het trekt toeristen aan en, na eventuele opheffing van het verbod, zijn olifanten een bron van ivoor en andere nuttige producten. Aan de andere kant leveren olifanten schade op aan landbouwgewassen en mogelijkerwijs ook aan natuurparken. Zolang het handelsverbod gehandhaafd blijft zal een overheid proberen de baten van de bescherming van een *extra* olifant (in dit geval dus inkomsten uit toerisme) gelijk te stellen aan de kosten die deze olifant met zich meebrengt. Dit wordt bereikt door regelmatig olifantenpopulaties uit te dunnen, ook al mag het aldus verkregen ivoor niet verkocht worden. Dergelijke operaties zijn in enkele landen al aan de gang. Indien het handelsverbod wordt opgeheven zal de overheid olifanten ook beschouwen als een vernieuwbare bron van ivoor. Het bejagen van olifanten voor ivoor levert dan directe baten op en het laten leven van een olifant wordt een soort van investering. Met behulp van data voor Afrika in het algemeen en Kenia in het bijzonder laten we zien dat de optimale populatie olifanten zoals die met een *handelsverbod* door een overheid wordt nagestreefd niet noodzakelijkerwijs groter is dan de optimale populatie met *handel* in ivoor. Een belangrijke factor die de optimale hoeveelheid olifanten in de situatie met handel bepaald is de hoogte van de discontovoet die de overheid gebruikt. Een hoge discontovoet leidt tot lage olifantenpopulaties wanneer handel in ivoor is toegestaan, en omgekeerd. De discontovoet waarbij de hoeveelheid olifanten met handel in ivoor de optimale populatie met een handelsverbod overtreft is in de nabijheid van de "sociale discontovoet". Als de discontovoet zoals gehanteerd door Afrikaanse overheden lager wordt dan deze *break even discount rate*, dan zijn olifanten gebaat bij handel in ivoor. Aangezien de discontovoet waarschijnlijk niet constant is (veelal wordt verondersteld dat de discontovoet een afnemende functie van het inkomen is), concluderen we dat olifantenbeschermers in de toekomst wellicht zullen moeten pleiten voor opheffing van het handelsverbod. In het model hebben we geen rekening gehouden met stroperij. Dit beschouwen we een van de noodzakelijke uitbreidingen voor de toekomst.

In hoofdstuk 6 bekijken we de economische achtergrond van het verbod op de commerciële walvisvaart. We berekenen de optimale hoeveelheid dwergvinvissen in het Noordoostelijk deel van de Atlantische oceaan met behulp van een model waarin we rekening houden met de niet-gebruikswaarden van levende walvissen (de populariteit van organisaties als Greenpeace geeft aan dat veel mensen "nut" ontlenen aan levende walvissen). In tegenstelling tot ander onderzoek concluderen we dat de huidige populatie dwergvinvissen te laag is. Bovendien blijkt dat het optimaal is om volledig van walvisvangst af te zien tot de populatie gegroeid is tot de optimale omvang. Het moratorium is dus economisch te verdedigen. In het hoofdstuk demonstreren we tot slot dat een simpel statisch model, dat door sommige onderzoekers wordt gebruikt om dit soort problematiek te benaderen, ongeschikt is. Het model leidt tot beleidsaanbevelingen die diametraal tegenover de beleidsimplicaties van een dynamische specificatie staan.

De hoofdstukken 7, 8 en 9 betreffen visserijeconomie. Zoals besproken in hoofdstuk 2 heeft exploitatie van de zee gedurende een lange tijd plaatsgevonden onder condities van *open access*. Dit betekent dat het niet mogelijk was om geïnteresseerde vissers te weren van bepaalde visgronden. Omdat niemand geweerd kon worden, voelde niemand zich verantwoordelijk voor een duurzaam beheer. Iedereen zal proberen op zo kort mogelijk termijn zo veel mogelijk geld te verdienen door vis te vangen voordat een andere visser daar aan toe komt. Onder *open access* verdwijnt de *rent* volledig: er wordt zoveel gevist dat de prijs uiteindelijk gelijk is aan de marginale vangstkosten. Aan het eind van de jaren '70 is aan deze toestand een einde gekomen door het instellen van exclusieve zones waarbinnen overheden het recht krijgen om buitenstaanders te weren en eigen beleid te voeren. We hebben in hoofdstuk 7 onderzocht of de overgang van *open access* naar een situatie waar overheidsbeleid gevoerd kan worden heeft geleid tot een beter beheer van visbestanden. Op basis van de economische theorie kan voorspeld worden dat overheden (net als bedrijven met gegarandeerde eigendomsrechten) rekening houden met levende vissen als investering. Dit betekent dat de rent van de hulpbron positief moet worden. Empirisch onderzoek met behulp van Duitse data wijst uit dat de rent inderdaad positief geworden is na instellen van Europees visserijbeleid.

De conclusie uit hoofdstuk 7 is zeker niet dat het huidige beleid optimaal is. Er werd slechts geconcludeerd dat vergeleken met vroeger de hulpbron nu efficiënter geëxploiteerd wordt. Ander onderzoek heeft aangetoond dat het huidige beleid verre van optimaal is. Een verklaring voor suboptimaal visserijbeheer (die verrassend vaak over het hoofd wordt gezien) is dat de standaard-veronderstelling dat overheden proberen "de welvaart voor de samenleving te maximaliseren", niet opgaat. De overheid wordt beïnvloedt door belangengroepen met bepaalde doelstellingen, die kunnen afwijken van wat sociaal wenselijk is. In hoofdstuk 8 laten we zien dat een belangengroepen-benadering, waarin we vissers, arbeiders en consumenten onderscheiden, leidt tot beheer dat afwijkt van hetgeen standaardmodellen voorschrijven. De observatie dat er onvoldoende vis rondzwemt in de Noordzee hoeft niet noodzakelijkerwijs te impliceren dat de overheid haar doelstellingen niet haalt door het verkeerd inzetten van bepaalde instrumenten. Het kan even goed wijzen op afwijkende doelstellingen als gevolg van lobbyende belangengroepen.

In hoofdstuk 8 wordt expliciet rekening gehouden met de "macht" van de visserijsector. Het is niet realistisch om te veronderstellen dat een overheid simpelweg een bepaald beleid kan voeren dat rechtstreeks tegen de belangen van bepaalde groepen ingaat. In hoofdstuk 9 laten we zien hoe, in het geval van de visserij, een overheid beperkende maatregelen (zoals het verkleinen van quota) kan doorvoeren zonder de belangen van de vissers al te zeer te schaden. Het instellen van een termijnmarkt voor verhandelbare visquota neemt het prijsrisico dat vissers lopen voor deze quota weg. Dit betekent dat een risico-mijdende visser beter af is dan voorheen. Dit biedt het perspectief om op hetzelfde moment het quotum te verlagen, zodat per saldo de visser niet beter of slechter af is. Financiële instrumenten kunnen dus (in theorie) leiden tot bescherming van natuurlijke hulpbronnen.

Het proefschrift bevat tot slot twee hoofdstukken over bosbouw waarin onzekerheid een belangrijke rol speelt. In hoofdstuk 10 laten we zien hoe een bepaald soort onzekerheid met betrekking tot de (veelal strijdige) voorkeuren van beleidmakers (bijvoorbeeld: "*werkgelegenheid is heel belangrijk, maar de hoeveelheid natuur mag niet veel kleiner worden*") geïncorporeerd kan worden in een landgebruiksmodel. Daarnaast behandelen we onzekerheid die samenhangt met gebrekkige kennis omtrent technische coëfficiënten in bosbouwmodellen. We gebruiken *fuzzy set* theorie, gebaseerd op *membership functions*, om onzekerheid te modelleren. Een cruciaal aspect van fuzzy logic is dat elementen gedeeltelijk tot een bepaalde set kunnen behoren. We vinden dat de resulterende landallocatie te verkiezen valt boven de uitkomsten van een recht-toe-recht-aan model.

In hoofdstuk 11 behandelen we een heel ander soort onzekerheid. Zoals boven vermeld fluctueren houtprijzen in de praktijk. In tegenstelling tot *fuzzy set* theorie (waarbij niet het plaatsvinden van een gebeurtenis onzeker is, maar de gebeurtenis zelf) is de kansverdeling van de stochastische prijs bekend. In de literatuur is een zoekmodel met reserveringsprijzen ontwikkeld om de baten van een strategisch, flexibel kapbeleid (kappen wanneer de prijs hoger is dan de reserveringsprijs, afzien van kappen indien de prijs lager is) te kunnen vergelijken met de opbrengsten van het meer rigide Faustmann model (zie ook hoofdstuk 2). Uit studies blijkt dat de winsten van boseigenaren met ongeveer 30% stijgen wanneer een reserveringsprijs-benadering gekozen wordt. In hoofdstuk 11 hebben we dit model uitgebreid door in aanvulling op strategische eindkap ook strategisch uitdunnen in ogenschouw te nemen. We vinden dat de reserveringsprijs voor uitdunnen altijd lager is dan de reserveringsprijs voor kappen en dat een strategisch kapbeleid ten aanzien van uitdunnen de winst van een boseigenaar substantieel verhoogd. De extra baten van flexibel uitdunnen zijn, in overeenstemming met de verwachtingen, minder groot dan de extra baten van het volgen van een strategisch eindkapbeleid.

CURRICULUM VITAE

Erwin Hendricus Bulte werd geboren op 26 april 1968 te Amsterdam. In 1986 behaalde hij het Atheneum B diploma aan het Pius X College te Beverwijk. Datzelfde jaar begon hij met de studie Bosbouw aan de Landbouwuniversiteit Wageningen (LUW). In 1989 begon hij daarnaast aan een studie Agrarische Economie. In 1992 behaalde hij het doctoraal diploma voor beide studies, met als afstudeervakken o.a. Tropisch Natuurbeheer, Ontwikkelingseconomie, Landbouwpolitiek en Milieu-economie. In september 1992 trad hij in dienst bij de vakgroep Staathuishoudkunde van de LUW als Assistent-in-Opleiding (AIO). Als AIO heeft hij het programma van het landelijke Netwerk Algemene en Kwantitatieve Economie (NAKE) doorlopen. Voor het doen van onderzoek heeft hij korte periodes doorgebracht aan the University of British Columbia, the University of Illinois en the Pennsylvania State University. Vanaf oktober 1996 is hij als universitair docent in dienst bij de vakgroep Ontwikkelingseconomie van de LUW.