A COMPUTABLE DYNAMIC BIOECONOMIC MODEL OF THE OPTIMAL UTILISATION AND MANAGEMENT OF SOUTH AFRICA'S RENEWABLE MARINE RESOURCES: A CASE STUDY OF THE HAKE FISHERY

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CPUE	Catch per Unit Effort
EEZ	Exclusive Economic Zone
ICSEAF	International Commission for the Southeast Atlantic Fisheries
ITQ	Individual Transferable Quota
MEY	Maximum Economic Yield
MSY	Maximum Sustainable Yield
SADSTIA	South African Deep-Sea Trawling Industry Association
SECIFA	South East Coast Inshore Fishing Association
SFAC	Sea Fisheries Advisory Committee
SFRI	Sea Fisheries Research Institute
TAC	Total Allowable Catch
TURF	Traditional Use Right in Fishing
UN	United Nations
UNCLOS	United Nations Conference on the Law of the Sea

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December 1997

DECLARATION

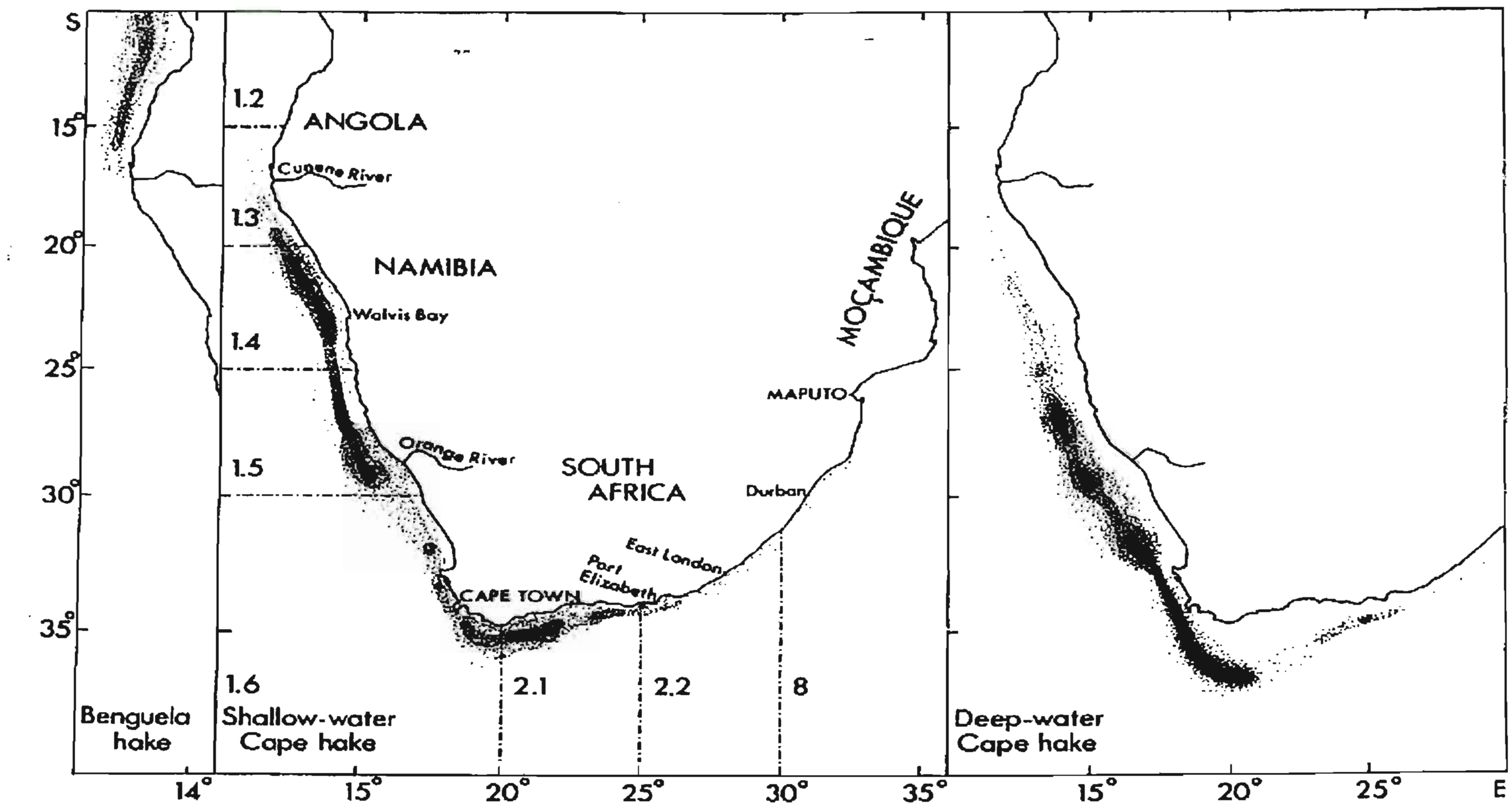
In accordance with the regulations of the University of Natal, I certify that the contents of this thesis are my own original work unless specifically indicated to the contrary in the text.

I further declare that this thesis has not been presented to any other university.

Signed: ______________

Date: 8 December 1997

MAP OF THE WEST AND SOUTH COAST HAKE FISHERIES



Source: Payne (1989, 137).

It is not from the benevolence of the butcher, the brewer, or the baker, that we expect our dinner, but from their regard to their own interest .... Every individual ... intends only his own gain, and he is in this, as in many other cases, led by an invisible hand to promote an end which was no part of his intention. Nor is it always worse for society that it was no part of it. By pursuing his own interest he frequently promotes that of society more effectually than when he really intends to promote it.

Adam Smith, *The Wealth of Nations* (1776, Book I, chapter 2, 2 and Book IV, chapter 2, 9)

Ruin is the destination toward which all men rush, each pursuing his own best interest in a society that believes in the freedom of the commons. Freedom in a commons brings ruin to all.

Garrett Hardin, *The Tragedy of the Commons* (1968, 162)

CHAPTER 1: INTRODUCTION

1.1 The Economics of Extractive Resource Use

Economics, as a discipline, is concerned with establishing optimal rates of use and patterns of allocation of scarce (or limited) resources as the basis for maximising social welfare. In this regard, historically, economists have tended to focus primarily on the use and management of non-renewable or exhaustible natural resources.[1] This focus is principally due to the fact that, by definition, non-renewable resources are relatively more scarce than renewable resources which, in turn, serves to escalate the threat of resource exhaustion and thereby jeopardise the sustainability of economic growth and development. However, more recently, the concern of economists has shifted to incorporate renewable resources. In this regard, Berkes and Farvar (1989, 3), for example, noted: 'increasingly, there is general agreement that extractive [that is, non-renewable as well as renewable] resources should be used sustainably'.[2] Continuing in this vein, Peterson and Fisher (1977, 681) argued:

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[1] The term 'resources' is applied by economists in a wide-ranging capacity to include 'tangible' and 'intangible' resources. 'Tangible' resources are typically identified by economists as consisting of 'land, labour and capital' (where 'land' incorporates all naturally occurring resources). 'Intangible' resources consist of productive resources such as entrepreneurship, human capital and technology. In this paper, the term 'resources' should generally be taken to mean 'natural resources'. Hence, 'resource economics' is, by definition, concerned with the optimal use and management of natural resources.

[2] The concept of sustainability has proved difficult to define and operationalise, and a large and diverse literature has emerged in recent years concerning the notion of 'sustainable development' (Reid, 1995, xiii-xx; Turner, 1995, 3). Indeed, many definitions (often incompatible with each other) have been suggested and debated, thereby exposing a range of approaches – including economic, ecological and socio-cultural approaches (Munasinghe, 1993) – linked to different world views (Pearce *et al*. 1989; Peace and Turner, 1990). However, a widely used definition is that offered by the World Commission on Environment and Development (1987): 'Sustainable development is development which meets the needs of the present without compromising the ability of future generations to meet their own needs.' That is to say, sustainable development is future oriented in that it seeks to ensure that future generations are at least as well off, on a welfare basis, as current generations. Sustainability is, therefore, a dynamic or intergenerational concept (Turner, 1995). See Reid (1995) or Turner (1995) for comprehensive reviews of the concept of sustainable development.

> Stimulated by the energy and environmental 'crises' of recent years, and perhaps by the widely debated Club of Rome study on the limits to growth ... a vigorous literature has evolved on [the management and optimal use of] natural resources and the environment.

However, the concern of economists with natural resource scarcity is not recent. Indeed, economists' concern with the use and management of natural resources dates to the emergence of economics as an identifiable discipline (Randall, 1987 18). As evidence of this, Peterson and Fisher (1977, 681) argued that the 'dismal science' has its roots in Malthus' apocalyptic scenario of inevitable human misery resulting from limited agricultural land and population growth: 'if the only check on population is misery, then population growth will continue until the population is miserable' (Peterson and Fisher, 1977, 681).[3] Similarly, Heilbroner (1973, 76) noted:

> For what the [Malthus] essay on population said was that there was a tendency in nature for population to outstrip all possible means of subsistence. Far from ascending to an ever higher level, society was caught in a hopeless trap in which the human reproductive urge would inevitably shove humanity to the sheer brink of the precipice of existence. Instead of being headed for Utopia, the human lot was forever condemned to a losing struggle between ravenous and multiplying mouths and the eternally insufficient stock of Nature's cupboard, however diligently that cupboard might be searched.

Anyone who was not sufficiently depressed by Malthus had only to turn to Ricardo (1817), who considered economic progress to be impenetrably constrained by the fixed supply of land.[4] A similar picture of gloom was painted by Jevons (1865) in *The Coal Question*. In that book, Jevons focused principally on the issue of whether Britain could maintain her industrial supremacy.[5] But, as noted by Gordon (1973, 108), Jevons' 'argument was meant to apply to all industrial civilizations' and, furthermore, 'Coal supply plays the same role of

[3] Thomas Carlyle is credited with first describing economics as the 'dismal science' after reading Malthus' (1798) (then anonymous) treatise *An Essay on the Principle of Population as It Affects the Future Improvement of Society* (Heilbroner, 1972, 76).

[4] The 'limits' concept is also present in Mill's (1857) 'stationary state' in which the stock of people and the stock of capital assets are constant.

[5] It ought to be noted that the English had long been concerned with the issue of the depletion of extractive resources. For example, laws were passed during the reign of Elizabeth I (1558-1603) which limited forging and furnace operations in districts where timber had become sufficiently scarce. Indeed, by the early seventeenth century, British industrial expansion was 'confronted with nothing less than a "national crisis" ... as a result of the severe depletion of timber supplies' (Rosenberg, 1973, 112).

ultimate constraint in Jevons' scenario as land supply does in the Ricardo model.' That is to say, Jevons' concern lay as much with the general problem of the constraints placed upon economic expansion by the scarcity of resources, as identified by earlier economists, as it did with the specific problem of the scarcity of coal.

From these early concerns, a substantial literature dealing with the issue of extractive resource use and management has developed.[6] The first comprehensive theory of extractive resources is credited to Gray (1914), who developed a microeconomic theory of the effects of taxation and price levels on extraction rates in mines.[7] However, this early theory on the management of non-renewable resources, and the extensions made to it by a number of writers (among them, Paish, 1938; Carlisle, 1954; Scott, 1955a; Brobst *et al*, 1973; Fisher and Krutilla, 1975; Helliwell, 1976), suffered from a fundamental flaw: the theory was couched in a static framework, whereas the problem of resource management is, by nature, one of dynamic optimisation. As Peterson and Fisher (1977, 693) noted:

> The decision on when and how hard to squeeze the orange depends on conditions in the future as well as the present. We shall assume that a resource manager wishes to determine the time path of extraction that maximises the present value of an exhaustible resource. This clearly requires dynamic optimisation.

Similarly, Dasgupta and Heal (1978) have emphasised that dynamic, or intertemporal, efficiency refers to an entire path of resource use through time. Therefore, any assessment as to whether or not a current pattern of resource

[6] At the broadest level, this literature can be divided into two branches, that dealing with non-renewable extractive resources, and that dealing with renewable extractive resources. Following the convention established by Ciriacy-Wantrup (1952, 35-8), extractive resources are defined as renewable or non-renewable, depending on whether they exhibit 'economically significant rates of regeneration'. It should be noted that Koopmans (1974) and Pindyck (1978) made the further distinction between exhaustible and non-renewable resources by noting that while both exhaustible and non-renewable resources do not exhibit growth or regeneration, new reserves of the latter can be acquired through exploratory effort and discovery.

[7] Gray (1914) developed what is now known as the 'user cost approach': the sacrifice of future use caused by consuming a unit of an exhaustible resource today. The cost of using an exhaustible resource was therefore argued to be made up of the sum of its extraction costs and this 'user cost' element.

use is efficient requires not only looking at current rates of use but 'peering into the distant future' as well (Feige and Blau, 1980, 117). Thus, it is argued that natural resource management hinges on the principle of dynamic, rather than static, efficiency.

The seminal paper dealing with optimal dynamic resource use is that provided by Hotelling (1931):

> The theory of natural resource economics ... [is] concern[ed] with the intertemporal allocation of renewable and non-renewable resources ... [and] typically applies dynamic control methods of analysis to problems of intertemporal resource usage. This has its roots in ... the seminal paper by Harold Hotelling (1931). (Cropper and Oats, 1992, 677-8)

Hotelling (1931) set out the requirements for dynamic efficiency in the extraction of non-renewable resources by showing that, under competitive conditions, the rent or royalty on a natural resource (the price net of extraction costs for the marginal unit) would increase over time at a percentage rate equal to the resource owner's discount rate. Thus, for a hypothetical resource with costless extraction, the 'Hotelling rule' states that optimal extraction, that is, dynamic efficiency, is achieved by the real price of the resource rising at a rate equal to the rate of interest (Feige and Blau, 1980, 119; Martinez-Alier, 1987, 164-71; Pearce, 1991, 317).[8] Since then, 'Hotelling's rule' has served as the platform for the vast theoretical literature that has developed *vis-à-vis* the optimal use and management of non-renewable resources (Peterson and Fisher, 1977).[9]

[8] See Dasgupta and Heal (1979) for a more detailed consideration of Hotelling's rule. Also see Conrad and Clark (1987) for a discussion and example of the application of the Hotelling model.

[9] Gordon (1967) and Cummings (1969) have also made important contributions to the theory of dynamic optimisation of non-renewable extractive resources. Less technical treatments of the theory of non-renewable resource extraction have been provided by Herfindahl (1955), Herfindahl and Kneese (1974) and Solow (1974). Much of the modern literature is devoted to investigating the implications of relaxing the assumptions of Hotelling's original work, and includes considerations of the impact of industrial structure, including monopolisation (Peterson, 1976) and cartelisation (Hynilicza and Pindyck, 1976; Schmalensee, 1976) of industries, on optimal extraction rates; the impact of exploration on non-renewable resource extraction (Lorie and Savage, 1955; Adelman, 1970 & 1972; Uhler, 1975) which, as noted above, provides for the important distinction, as made by Koopmans (1974) and Pindyck (1978), between non-renewable and exhaustible resources; the impact of variable (rather than zero) costs (due to, for example, declining ore grades) on optimal extraction rates (Solow and Wan, 1976); the role played by uncertainty in influencing decision making *vis-à-vis* optimal extraction rates of non-renewable resources (Grayson, 1960; Kaufman, 1963; Weinstein and Zeckhauser, 1975; Arrow and Chang,

While concerns with resource management historically lie with optimal use of non-renewable extractive resources (reinforced by, *inter alia*, the 'limits to growth' debate of the 1970s, which revolved around the possibility of the exhaustion of non-renewable resources),[10] more recently, emphasis in the literature has shifted to consider the optimal use and management of renewable extractive resources. Indeed, the 1980s witnessed a definite shift in emphasis from the exploitation of non-renewable to renewable resources. For example, Chiras (1992, 10-11) noted:

> ... economies depend chiefly on our ability to draw sustenance from a finite natural world – that is, from nonrenewable resources such as oil, natural gas, and minerals. Given trends in population growth, nonrenewable resource supplies, and demand, our continued existence will depend upon our ability to shift from finite nonrenewable resources to renewable resources ... [and] the destruction of these [renewable] resources is far more dangerous than the quick depletion of nonrenewable resources – oil and minerals – now under way.

Pearce (1993, 73-4) identified three main causes of this switch in concern amongst economists. First, it became apparent that 'renewability' did not mean that resources would necessarily renew themselves. Renewability depended critically on the management regime in place; over-exploitation of renewable resources – such as tropical forests or freshwater fish stocks – could result in the collapse or extinction of these resources, thus rendering them 'non-renewable'. Second, the vast literature that developed, in particular, over the course of the

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1978; Hoel, 1978b; Devarajan and Fisher, 1981); and the impact of recycling on optimal rates of extraction on non-renewable resources (d'Arge and Kogiku, 1973; Schulze, 1974; Weinstein and Zeckhauser, 1974). A number of recent contributions have analysed the effects of initial investment costs – set-up costs – on the Hotelling path (Campbell, 1980; Pearce, 1991). It should also be noted that Hotelling considered the implications of relaxing some of the assumptions of his original model in investigating, amongst other things, the impact of monopoly, costs that rise as extraction increases (the Ricardian case), the influence of fixed investment costs and the effect of a severance tax (that is, a tax on depletion) on optimal extraction rates of non-renewable resources (Pearce, 1991, 326-8). Fuller reviews of these early developments in the theory of non-renewable resource use are provided elsewhere (Weitzman, 1975; Peterson and Fisher, 1977; Pearce, 1991).

[10] For example, in the influential work *Limits to Growth* (Meadows *et al*, 1972, 64-7) it was estimated that the remaining 'life' of aluminium, given existing estimates of reserves, was 31 years, and that world petroleum and gold reserves would last 20 and 9 years respectively, from the year of estimation (1972). Furthermore, the 'energy crisis' of the 1970s tended to give the discussion of resource exhaustion a politically relevant basis, even though the crisis was one of price and not the physical availability of resources (Pearce, 1993).

1970s, had focused on selected resources, especially fisheries and forests. As scientific knowledge grew, however, it was recognised that ecosystems in general behave as renewable resources. This view has its roots in Boulding's (1966) essay on 'spaceship earth'. Whereas the conventional 'textbook' treatment depicted the economy as a linear system (resources flowing to consumption and investment), Boulding drew on two laws of thermodynamics to establish two propositions. The first law of thermodynamics, the conservation of matter or energy, states that the economic system cannot destroy anything: whatever enters the economic system must reappear in the same or a transformed state elsewhere in the system. Thus, all resource extraction, production and consumption results in waste products ('residuals') equal in matter to the resources flowing into these sectors. The second law of thermodynamics, the entropy law, states that energy cannot be recycled, while many materials uses are so dissipative that the transformed product cannot be recycled. Thus, there is no possibility of waste products being 'fully' recycled to re-enter the resource flow. All this adds up to the economy being a circular system rather than a linear one, with resources flowing through the economic system and then on to the environment as a 'sink' for residuals or being partly recycled to become resources again.[11] Third, as the developing country perspective emerged, so it readily became apparent that it is renewable resources – such as water, trees, fish, crop residues, grass cover and soils – that matter most for the immediate welfare of the world's poor.[12]

The substantial origins of renewable resource economics can be traced back to the seminal article of Gordon (1954), in which an economic model of a renewable resource, the fishery, was developed using orthodox, microeconomic analysis in an effort to explain low incomes among fishers in Canada's marine

[11] Thus the carbon cycle, for example, operates as a balanced system in which carbon is emitted from living things and is absorbed by carbon 'sinks', notably oceans and forests (Pearse, 1993, 74).
[12] As an aside, it is interesting to note that it estimated that renewable energy sources, such as solar power, could meet 5 percent of the world's energy needs in 20 years and 50 percent by 2050 (*Business Week*, 1997).

fisheries. In setting out the problem of renewable resource management in economic terms, Gordon provided the principles which have served as the centrepiece for all subsequent developments in the theory of renewable resource economics (Clark, 1976; Pearce, 1991).[13] Overwhelmingly, renewable resources tend to be open-access resources (*res nulliae*), that is, property to which there is unrestricted access, such as fisheries, forests, wildlife stocks and 'communal' pastoral land (Morey, 1980; Berkes and Farvar, 1989; Pearce, 1991). Gordon (1954) argued that, if left unregulated, open-access renewable resources will be over-exploited. In Gordon's (1954, 141) own words: 'the uncontrolled equilibrium means a higher expenditure of effort, higher fish landings, and a lower continuing fish population than the optimum equilibrium'.[14] Indeed, Gordon (1954, 131) went on to argue that, unless regulated, the fishery will be driven to a point at which the total costs and total revenue of harvesting are equal. In this case, the economic rent or yield (revenue less economic costs) of renewable resource extraction is zero. Thus, Munro and Scott (1985, 623) noted:

> If a [renewable] resource is commercially valuable and is open to unrestricted exploitation, the resource will certainly be subject to excessive depletion from society's point of view. Since the resource is open to all and owned by none, [there is] no incentive to conserve the resource. A [person] who refrains from harvesting the resource is likely to find, not that he has helped conserve the resource, but rather that he has simply enhanced the harvest opportunities of his competitors.

The case is made more dramatically by Hardin (1968) in his celebrated paper *The Tragedy of the Commons*: 'Ruin is the destination toward which all men rush, each pursuing his own best interest in a society that believes in the freedom of the commons. Freedom in a commons brings ruin to all.' This is the

[13] While Hotelling's (1931) work has served as the platform for the substantial literature that has emerged since the 1930s on the optimal use and management of non-renewable resources (Cropper and Oats, 1992, 677-8), it has been argued that: 'Gordon's result's ... may be considered the second fundamental theorem of resource economics, complementing Hotelling's theorem for individually owned resource stocks' (Clark, 1976, 7).

[14] It should be noted that Gordon's argument with regard to higher fish catches falls away once the analysis is extended into a dynamic setting because, as recognised by Gordon, the ongoing lower biomass ultimately leads to lower catches.

so-called Class I problem of open-access renewable resources (Munro and Scott, 1985, 631; Johnston, 1992, 6).[15]

From this, it follows that it is necessary for authorities to regulate or control the use of renewable resources in order to prevent (unsustainable) over-exploitation or 'mining' of these resources. Historically, authorities have responded by adopting biological goals, such as maximum sustainable yield (or some derivative thereof), as the basis for sustainable exploitation of renewable resources. As Clark (1973b) noted: 'The most commonly encountered proposal for managing a biological resource is to maximise the sustained yield.' In this respect, Gordon (1954) provided a further important principle regarding the 'optimal' exploitation of renewable resources by arguing that biological principles are an incomplete basis for renewable resource use. The reason for this is straightforward: biological principles fail to maximise economic rent – the difference between sustainable revenue (catch) and the costs of harvesting – where maximisation of economic rent is taken as the basis for social welfare maximisation. As Gordon (1954, 129) noted:

> We can define the optimum degree of utilization of any particular fishing ground [renewable resource] as that which maximises the net economic yield, the difference between total cost, on the one hand, and total receipts (or total value production), on the other.

Thus, Gordon (1954) adopted the criterion of maximisation of economic rents as the appropriate basis for fisheries management, in contributing towards maximum social welfare.[16] In so doing, Gordon provided the justification for the replacement of biological principles with the bioeconomic (biological coupled with

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[15] The Class I problem of rent dissipation due to over-fishing under open-access conditions is to be distinguished from the Class II problem of open-access, as discussed later in this chapter, which refers to the situation where resources are preserved, but resource rents are dissipated through over-investment by fishers in boats, equipment and gear.

[16] The terms 'economic rent' and 'economic yield' are frequently used interchangeably; this study retains this practice.

economic) principle of maximum sustainable rent, as the basis for optimal utilisation of renewable resources.[17]

While the origins of renewable resource economics are found in Gordon's model, in retrospect, the model was deficient in a number of ways (Butlin, 1975). The model was developed within a static framework, and was based on the highly restrictive assumptions of autonomous, linear resource prices and harvesting costs. In addition, the model, as developed by Gordon, did not allow for the biological and economic uncertainty inherent in fishing and, more generally, renewable resource use. Moreover, the model failed to consider the implications of multi-species harvesting, species interaction and the possibility of multi-harvesting technologies. Since the mid-1950s, however, a vast literature has developed in an effort to compensate for the various deficiencies of the Gordon model.

Arguably the most successful extension of Gordon's work has involved shifting the model into a dynamic setting. The criterion of maximisation of economic rents adopted by Gordon (1954) is partial: it ignores the principle of optimisation across time. As early as 1955, Scott recognised the principal of dynamic efficiency, arguing that the optimal management of fisheries hinged upon the principle of maximising the net present value of all future net returns of the fishery (Morey, 1980, 840). It should be noted, however, that Gordon was acutely aware of the need for a dynamic approach to resource economics. For example, in a later work, Gordon (1956, 65-72) argued:

> The conservation problem is essentially one which requires a dynamic formulation .... The economic justification of conservation is the same as that of any capital investment – by postponing utilization we hope to increase the quantity available for use at a future date. In the fishing industry we may allow our fish to grow and to reproduce so that the stock at a future date will be greater than it would be if we attempted to catch as much as possible at the present time .... In theoretical terms this means that the optimum degree of exploitation of a fishery

[17] Economists typically identify three criteria in the management of renewable resources: economic efficiency, biological sustainability and social equity. While Gordon (1954) describes the bioeconomic principle of the maximisation of rent as 'optimal', rent maximisation ignores the issue of social equity. This shortcoming is addressed at a later stage in this study.

> must be defined as a time function of some sort. That is to say, it is necessary to arrive at an optimum which is a catch per unit of time, and one must reach this objective through consideration of the interaction between the rate of catch, the dynamics of fish populations, and the economic time-preference schedule of the community or the interest rate on invested capital. This is a very complicated problem and I suspect that we will have to look to the mathematical economists for assistance in clarifying it.

Indeed, it was not until the 1970s, with the development of sufficiently sophisticated mathematical tools (in particular, optimal control theory) that the formidable task of casting Gordon's renewable resource model in a dynamic setting was accomplished by, *inter alia*, Plourde (1970 & 1971) and Quirk and Smith (1970).[18] Since then, an extensive and thorough treatment of the dynamic optimisation problem has been provided by Clark (1976) and Munro (1981).

In turn, this 'basic dynamic model' has been extended to address a number of the additional limitations of Gordon's initial work (Munro and Scott, 1985). The most significant extensions of the model include the relaxation of the restrictive assumptions of autonomy in prices and costs (Clark and Munro, 1982; Munro and Scott, 1985) and linearity in prices and costs (Copes, 1970 & 1972; Clark and Munro, 1975 & 1983; Clark, 1976; Munro and Scott, 1985). The model has also been extended to incorporate multi-species harvesting and multi-harvesting technologies (Quirk and Smith, 1970; Clark, 1976; Hupert, 1979; Mendelssohn, 1978 & 1980; May *et al*, 1979; Bishop and Samples, 1980; Munro, 1982; Sobel, 1982; Clark, 1985; Lipton and Strand, 1989; Smale, 1993); and a considerable, but incomplete, literature dealing with the issue of uncertainty has developed since the early 1970s (Mann, 1970; Jacquette, 1972 & 1974; Thompson *et al*, 1973; Reed, 1974 & 1979; Beddington and May, 1977; Gleit, 1978; Mendelssohn, 1978 & 1980; Lewis, 1981; Spulber, 1982a; Charles, 1983;

[18] Also see Brown (1974), Neher (1974), Long (1977), Peterson and Fisher (1977), Smith (1977), Dasgupta and Heal (1979), Levhari *et al* (1981), Clark and Munro (1981) and Dasgupta (1982) for examples of alternative treatments of the capital-theoretic approach to renewable resource economics.

Andersen and Sutinen, 1984; Clark and Kirkwood, 1986; Getz and Haight, 1989; Hannesson and Steinshamn, 1990).[19]

Three points should be noted regarding the above developments in the theory of renewable resource economics. First, the bulk of the advances have been made within the field of fisheries economics (Clark, 1976; Peterson and Fisher, 1977). However, this is primarily due to the origins of the theory, and does not imply that the theory is not applicable to other renewable resources. On the contrary, advances made in the theory of fisheries economics are readily and easily applied to other renewable resources (Clark, 1976 & 1985; Getz and Haight, 1989; Pearce and Turner, 1990; Turner, 1995; Klemperer, 1996).[20] Indeed, this point was made by Gordon (1954, 124) himself:

> Although the theory presented in the following pages is worked out in terms of the fishing industry, it is, I believe, applicable to all general cases where [renewable] natural resources are owned in common and exploited under conditions of individualistic competition.

Second, while considerable advances have been made in the theory of renewable resource economics since its formulation in the mid-1950s, Gordon's principal results remain valid, and continue to serve as guiding principles in the theory of renewable resource economics. That is, unless regulated, open-access renewable resources will be over-exploited; and biological principles are a partial, and therefore incomplete, basis for the management of renewable resources. Third, and of seminal importance, in spite of the validity of Gordon's results, these principles have not spilled over into practice; this argument is developed in Section 1.2 below.

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[19] This review is by no means complete; a number of writers have made further contributions to the theory. For example, Clark and Munro (1980) and Schworm (1983) have developed a multi-sectoral model of the fishery; and a number of writers, including Lewis and Schmalensee (1977), Getz (1979), Spulber (1982) and Flaaten (1983), have considered the issue of optimal harvest techniques. Some of these extensions are discussed more fully in Chapter 3. Fuller reviews of developments in the theory of renewable resource use and management have been provided by, amongst others, Butlin (1975), Clark (1976 & 1985), Cushing (1977), Hannesson (1978), Mirman and Spulber (1982), Cunningham *et al* (1985), Munro and Scott (1985) and Wilen (1985).

[20] For example, Klemperer (1996) applied many of the developments discussed above to forestry resources.

1.2 Open-Access Renewable Resource use and Management: A Review of the Evidence

While substantial advances have been made in the theory of renewable resource economics over the past four decades, these developments have generally not spilled over into practice. Indeed, evidence of the over-exploitation of renewable resources abounds.[21] Greene (1991), for example, has argued that growth in the world's population and economy, increased and widespread industrialisation and the development of international trade have occurred on such a scale that severe environmental damage and unsustainable exploitation of the earth's resources are taking place on a global scale. More specific reference to the over-exploitation of renewable resources has been made by Chiras (1992, 10):

> Today's challenge ... is to refrain from destroying forests, grasslands and other [renewable] resources that can provide a continuous supply of food and materials, and to rebuild (restore) what has already been lost .... Unfortunately, heavy timber cutting in tropical and temperate forests, overhunting and overfishing are destroying renewable resources at a rapid rate and, in the process, undercutting the resource base upon which humanity depends.

In similar vein, Berkes and Farvar (1989, 6) wrote:

> The notion advanced in the *State of the World* – that the general economic decline of Africa is largely attributable to the non-sustainable use [over-exploitation] of [renewable resources such as] forest, grazing land, water and soil resources – is now widely accepted.

More detailed evidence on the over-exploitation of renewable resources, including forests, water, wildlife, soils and pastures is readily available. For example, Chiras (1992, 6) noted: 'Rain forests are a rich source of diverse wild species, new medicines and useful products. Once covering an area the size of the United States [of America], tropical rain forests have been reduced by at least a third, perhaps as much as a half.' In considering the use of soil, Chiras (1992, 221) noted:

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[21] Recent examples of this evidence can be found in, *inter alia*, Morey (1980), Miller (1988), Silver (1990), Council on Environmental Quality (1990), Quarrie (1992) and VanDeVeer and Pierce (1994).

> Each year, most countries lose topsoil to wind and water erosion in amounts that far exceed soil regeneration. In addition, millions of acres of farmland and rangeland are destroyed annually because of desertification caused by overgrazing and intensive agriculture. Worldwide, an area the size of Belgium (about 15 million acres) turns to desert each year. According to the UN [United Nations] Environment Program, if current trends continue, one-third of the world's cropland will become desert by the end of the century.

However, it is arguable that the most dramatic examples of over-exploitation of renewable resources have occurred in commercial marine fisheries. As Clark (1985, 6) has noted:

> ... anyone familiar with the world-wide history of fisheries development since the 1950s has no need of academic verification to realise that Gordon's predictions have now been borne out over and over again. Depletion of major fish stocks and the impoverishment of fishing fleets and processing companies have become common phenomena worldwide.

In support of this argument, Clark (1985, 6) provided a number of examples of the collapse (or near collapse) of commercial marine fisheries over the course of this century (Table 1.1).[22] However, a more comprehensive picture of the over-exploitation of commercial marine fisheries has been provided by Loayza (1992, 6) who noted that between two-thirds and four-fifths of fish stocks in major fishing areas of the Atlantic and Pacific oceans are in a state of full exploitation, over-exploitation or depletion. In similar vein, Knauss (1994) has estimated that in the United States of America, approximately 45 percent of fish stocks are over-exploited, while Rosenberg *et al* (1993) have estimated that 59 percent of stocks found in European Union waters are over-exploited.

[22] As is the case with other renewable resources, a vast literature documenting the over-exploitation of fish stocks exists. Recent examples include, *inter alia*, Clark (1985), Johnston (1992), Loayza (1992 & 1994) and VanDeVeer and Pierce (1994).

Table 1.1: A Sample of Depleted Commercial Marine Fisheries

Stock	Peak Catch (Year)	Catch (1981)
Antarctic blue whale	29 000 units (1931)	0 units
Antarctic fin whale	27 000 units (1938)	0 units
Hokkaido herring	850 000 tons (1913)	0 tons
Peruvian anchoveta	12 300 000 tons (1970)	300 000 tons
Southwest African pilchard	1 400 000 tons (1968)	0 tons
North Sea herring	1 500 000 tons (1962)	Negligible
California sardine	640 000 tons (1936)	0 tons
Georges Bank herring	370 000 tons (1968)	0 tons
Japanese sardine	2 300 000 tons (1939)	17 000 tons (1973)

Source: Clark (1985, 6).

1.3 Over-Exploitation of Renewable Resources: Reasons, Implications and Responses

It can be argued that the over-exploitation of renewable resources can be ascribed to three main factors. First, contrary to the prescriptions of Gordon (1954) and others (Scott, 1955; Crutchfield, 1956; Turvey 1957 & 1964; Crutchfield and Zellner, 1963), in many instances renewable resources have gone unprotected (Chiras, 1992; Council on Environmental Quality, 1990; Greene, 1991). For example, renewable marine resources essentially remained 'open-access' until the early 1970s; and it was only with the Third United Nation's Convention on the Law of the Sea (UNCLOS) (1973-82) that the notion of an exclusive economic zone, which gave jurisdiction to coastal states over sea and seabed resources located within 200 nautical miles of coastlines, became widely adopted (Cushing, 1977; Smith and Greene, 1991, 38-48). Even so, transboundary migration, for example, has served to undermine the effectiveness of regulations that have been introduced since the 1970s (Smith and Greene, 1991, 48-53). Moreover, the 'high seas' remain open-access, and there are numerous examples of the over-exploitation of these resources (Cleroux, 1995; Le May, 1995e; Usher, 1995). Of even greater concern are examples of government policies having exacerbated the over-exploitation of renewable resources. For example, in examining the use of forest resources, Repetto (in Chiras, 1992, 119) concluded that:

> ... tax and trade regimes, land tenure laws, agricultural resettlement programs and administration of timber concessions with loggers are but a few of the policies that aggravate deforestation ... [and] these policies ... contribute significantly to the wasting of forest resources.

Second, where authorities have attempted to manage renewable resources, they have tended to adopt biological principles, such as maximum sustainable yield, as opposed to the bioeconomic principle of rent maximisation, as prescribed by Gordon (1954), as the basis for managing the use of renewable resources. Clark (1976, 1-3), for example, noted:

> The management of renewable resources, where it has been practised at all, has generally been based on the concept of maximum sustainable yield .... This is perhaps the simplest possible management objective that accounts for the fact that a biological resource stock cannot be exploited too heavily without an ultimate loss of productivity. Few biological [renewable] resource stocks, historically speaking, have been managed on the [welfare maximising] present-value criterion.[23]

Third, where authorities have attempted to manage the use of renewable resources, they have tended to adopt 'direct' controls, such as gear and equipment restrictions, time and place restrictions (including closed seasons and closed areas), harvesting restrictions, quality controls and licence limitations. However, the success of this so-called 'command-and-control' approach in achieving optimal exploitation of renewable resources has been limited. Indeed,

[23] As recently as the beginning of this decade, Smith and Greene (1991, 49) argued: 'The efficient management of a fishery should give the greatest catch year after year (the maximum sustainable yield) while maintaining an equilibrium stock.' See also Cunningham (1980).

while there are examples of the successful application of direct controls,[24] these cases are heavily outweighed by the number of failures.[25]

Accordingly, the last fifteen years has witnessed a shift in emphasis within the literature, as well as among resource managers in a number of countries, towards the use of 'indirect' or market-based controls (Morey, 1980; Munro, 1982; Wilson, 1982; Cunningham, 1983; Rosenman, 1986; Hannesson, 1987; Arnason, 1994a & 1994b; Hannesson. 1994; Loayza, 1994; Neher, 1994; Wilen, 1994). The advocates of these indirect or market-based controls argue that the problems of renewable resource use are typically due to the absence of well-defined property rights. Thus it is concluded that the problems of renewable resource management are solved via either the creation of private property rights; or the 'bureaucratic simulation' (through the use of taxes or subsidies) of the outcome that might be expected from a private property rights system (Wilson, 1982, 418). However, contrary to these developments in theory, and the gamut of evidence pointing to the inefficiency and ineffectiveness of direct controls, where regulation of renewable resources has taken place, authorities have tended to adhere to a command-and-control approach (Fuggle and Rabie, 1992).

In summary, starting with the work of Gordon (1954), substantial advances have been made in the theory of renewable resource economics. However, these developments have generally not spilled over into practice. As a result, examples of over-exploitation and inefficient and ineffective management of renewable resources abound. This outcome is of considerable concern given the recent, but emphatic, shift in emphasis that has taken place within resource economics, that is, the shift toward the optimal use and management of renewable resources, as opposed to non-renewable resources, as the basis for achieving sustainable economic growth and development. This argument, which

[24] See, for example, Buxton (1993a & 1993b), Tunesi and Diviacco (1993) and Holland (1995).
[25] See, for example, Johnson and Liebecap (1982), Cunningham (1983), Munro and Scott (1985), DuPont (1991), Bennett and Attwood (1993), Lewis (1993a), Garratt (1993) and John (1994).

forms the basis for the current study, is further developed below, with particular reference to the South African case.

1.4 Extractive Resource Use, Sustainable Growth and Development: Applying the Theory of Renewable Resource Economics in South Africa

1.4.1 An Overview of the Arguments and Evidence

Extractive resources, especially non-renewable resources, are of significant importance to the South African economy. Indeed, the modern South African economy was founded upon the exploitation of non-renewable resources, especially diamonds, gold and, to a lesser extent, coal (Edgecombe and Guest, 1988; Natrass, 1990). As Natrass (1990, 129) noted:

> The mining sector has historically been the mainspring of South African modern economic development. The foundations of the present day economy were laid with the exploitation of the mineral discoveries of firstly, diamonds [in the 1860s] in Kimberley, and then gold [in the 1880s] on the Witwatersrand. Even today, more than a century later, the mining sector plays a crucial role in the continuing development of the South African economy.

More specific evidence of the importance of the mining industry to the South African economy is readily available. To start with, over the first two decades of this century mining and quarrying accounted for as much as one-quarter of the gross domestic product (Central Statistical Services, 1986 & 1991), and between 1910 and 1990, mining and quarrying contributed an average 15 percent per annum to South Africa's gross domestic product.[26] The extraction of non-renewable resources has also served as an important source of employment, with the mining industry directly employing more than 10 percent of the South African labour force between 1950 and 1990 (Central Statistical Services, 1986 & 1991). The sector has also accounted for the bulk of South Africa's foreign

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[26] It was not until 1965 that manufacturing exceeded the contribution of the agriculture and mining sectors to South Africa's gross domestic product (Lumby, 1990, 1).

exchange earnings over the course of this century: between 1910 and 1995 exports of metals and minerals, including gold. accounted for between one-half and three-quarters of the total value of South Africa's goods exports (Office of Census and Statistics, 1950 & 1957; Department of Statistics, 1966 & 1972; Central Statistical Services, 1986 & 1997).[27]

In addition. the 'long term and indirect effects' that is, backward and forward linkages, of the exploitation of mineral resources have served as the basis for industrialisation in South Africa (Lumby, 1990, 8).[28] Here it is worth quoting Natrass (1990, 162-164) at length:

> Although the exploitation of mineral resources has without doubt been the foundation upon which modern South Africa has been built, it is the process of industrialisation which has generated the structure that now rests on those foundations .... [However] South African manufacturing has always been closely tied to the fortunes of the mining industry and even today the links between the two sectors remain strong. In the initial phases of industrialisation, the products produced were closely allied to the needs of mines and the subsequent development of manufacturing relied heavily on the economic resources that had been accumulated by the mining sector .... The foreign exchange earned from the sale of minerals financed the extensive importation of capital goods and essential intermediate inputs that were needed by the growing industrial sector. The new sector also drew on the pool of skilled labour and the financial and business know-how that had developed as the mining industry expanded. Mining houses started to diversify their activities and moved into the industrial field, both directly and indirectly, allowing capital accumulated in the mining enterprises to be used to expand South African industrial capacity.

In short, the exploitation of non-renewable resources has served as a prime-mover in facilitating the economic growth and development of the modern South African economy.

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[27] It is worthwhile noting that the export of gold accounted for approximately 45 percent of South Africa's goods exports between 1910 and 1995 (Office of Census and Statistics, 1950 & 1957; Department of Statistics, 1966 & 1972; Central Statistical Services, 1986 & 1997).

[28] In commenting on the origins of the modern South African economy, Lumby (1990, 8-9) cautioned that the role played by the mining sector in facilitating industrialisation should not be overemphasised: 'In reality, it was only through its long-term and indirect effects that the discovery and exploration of South Africa's mineral deposits provided the "original stimulus" to local manufacture, while the "greatest stimulus" to local industry before the Second World War came from the combined influence of the First World War, tariff protection after 1925 and, above all, the unprecedented expansion in gold mining after South Africa's abolition of the gold standard and devaluation in December 1932.'

However, much in line with the shift in emphasis that has taken place in other countries toward the use of renewable resources as the basis for sustainable economic growth and development, concern has been expressed at the degree of dependence of the South African economy upon non-renewable resources. The basis for this concern has been well-explained by Kantey (1992, 11):

> Economic activity which is based solely on the extraction of non-renewable resources (such as gold and coal in South Africa) is, by definition, unsustainable in the long term. Declining levels of aggregate employment on South African mines – a result of lower yields and of lower mineral prices – are one symptom of this unsustainability.

More specific evidence is readily available. For example, the index of physical volume of mining production declined steadily from a peak of 117.6 index points in 1970 to 99.3 index points in 1994 (Central Statistical Services, 1986 & 1997). Somewhat more dramatically, the production of gold (by far South Africa's most important non-renewable resource) declined from a peak of 1 002 tons in 1980 (equivalent to three-quarters of world output in that year) to 579 tons in 1994 (equivalent to one-quarter of world output) (Central Statistical Services, 1986 & 1997). Furthermore, while employment levels in mining and quarrying increased steadily from 306 554 people in 1920 to 763 319 in 1987, this figure has since fallen sharply to 561 655 people in 1993 (a level last seen in 1958) (Central Statistical Services, 1986 & 1997). The declining importance of the non-renewable resource sector as an employer of labour is underlined by the fact that while mining and quarrying employed 8.08 percent of the economically active population in 1970 (the peak of the physical volume of mining production index), this figure had fallen by almost one-third to 5.62 percent by 1991 (Central Statistical Services, 1986 & 1997). In addition, in the foreign sector, exports of gold, which have historically accounted for approximately half (but in some years as much as three-quarters) of South Africa's exports by value, had fallen to 28.06 percent by 1993 (Central Statistical Services, 1986 & 1997).

In brief, South Africa's historical reliance on non-renewable resources, which has served as the basis for the growth and development of the modern economy, is unsustainable. Continuing in this vein, Kantey (1992, 11) has argued:

> Moving South Africa onto a sustainable growth path would thus require gradual but definite shifts in patterns of resource utilisation. There must be an active shift of the economic base towards renewable resources while, at the same time, making better use of non-renewable resources.

That is to say, moving the South African economy onto a sustainable growth path requires shifting the economic base away from a reliance upon non-renewable resources and towards the use of renewable resources. However, in line with the central principles of renewable resource economics outlined above, the success of such a strategy hinges critically upon the ability of authorities to prevent unsustainable use of renewable resources by establishing and securing optimal (that is, dynamic rent maximising) rates of extraction of renewable resources via the adoption of effective and efficient management tools. To date, this has not taken place in the use and management of renewable resources in South Africa.

As has been the experience in other countries, widespread and severe over-exploitation of renewable resources – including soils, forests and plants, wildlife, water and air – has taken place in South Africa.[29] For example, Verster *et al* (1992, 181) noted: 'soil is one of the most fundamental of the natural resources .... Unfortunately there is considerable evidence and general agreement that South African soils are deteriorating due to poor management practices.' With regard to pastures, Cowling and Olivier (1992, 221) observed: 'About 70 percent of South Africa is set aside as natural grazing. Most of this natural grazing land is seriously overstocked and as much as 60 percent of the veld is currently in poor condition.' Indigenous forest stocks have also been over-exploited:

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[29] Evidence of the over-exploitation of renewable resources in South Africa is provided by, *inter alia*, Bothma and Glavovic (1992), Cowling and Olivier (1992), Lusher and Ramsden (1992), O'Keeffe *et al* (1992), Petrie *et al* (1992), Rabie and Fuggle (1992), Rabie and Day (1992), Verster *et al* (1992) and International Development Research Centre *et al* (1994).

> South Africa is poorly endowed with natural forests, and the indigenous forest area has been further curtailed by over exploitation over the past 100 years .... South Africa is experiencing limited deforestation because most of the indigenous forests were cleared out over the past 100 years. (International Development Research Centre *et al.* 1994, 103)

In considering freshwater systems, O'Keeffe *et al* (1992, 293) noted:

> Water quality is deteriorating in a number of ways, and will continue to deteriorate as more pressure is put on the scarce supplies. Rivers are literally running dry as impoundment and abstraction increase, and as the buffering capacities of catchments are reduced by urbanisation and devegetation.

However, much in line with experiences in other countries, it is commonly acknowledged that the highest incidence (and most extreme examples) of over-exploitation of renewable resources in South Africa have occurred in the country's small, but economically important, commercial marine fisheries.[30] In this regard, examples of over-exploitation of marine resources include the collapse of the pilchard, chub mackerel and Cape horse mackerel stocks during the 1960s and 1970s; the severe over-exploitation of hake stocks – the backbone of South Africa's commercial marine fisheries – over the course of the late 1960s and 1970s; and the over-exploitation and, in instances, extinction, of resources in the inshore fisheries (which include, *inter alia*, the commercially valuable abalone and rock lobster resources) and linefisheries throughout this century (Beckley and Van der Elst, 1993; Stuttaford, 1994 & 1995; Van D. Boonstra, 1992, 1993, 1994 & 1995).[31]

The over-exploitation of renewable resources in South Africa can be ascribed to three main causes. First, it has been widely acknowledged that South Africa's renewable resources, including soil, water and vegetation, have been over-exploited due to an absence of management:

> Apartheid policies robbed the people of South Africa of the integrity to exercise effective control in the use of natural resources. The policies of successive previous governments ensured that there was a skewed distribution of access to natural resources that favoured the white minority. Under apartheid, economic

[30] The evidence in this regard is reviewed more fully in Chapter 2 of this study.
[31] There is also evidence to suggest that South Africa's freshwater fish stocks have been severely over-exploited (Bothma and Glavovic, 1992; Cardy, 1997), although this is beyond the scope of the current study.

> development was not guided by concern for sustainability of the natural resource base. but went full steam ahead without regard for the consequences for either human health or environmental health (International Development Research Centre *et al.* 1994, 8).

In this vein, it has been noted by the South African government in its *Reconstruction and Development Programme* that: 'South Africa's apartheid policies, combined with the unregulated activities of local and transnational corporations, contributed to the degradation of environmental resources, including soil, water and vegetation.' (African National Congress, 1994, 38; Hart, 1992). That is to say, the prescriptions of Gordon (1954) and others have failed to spill over into practice.[32] In support of this point, a comprehensive survey of the literature on renewable marine resources produced only one attempt (Andrew and Butterworth, 1987), and partial at that, to apply the theory of renewable resource economics in the South African context. Second, where regulation of resources has taken place, South African authorities have tended to rely upon biological principles, such as maximum sustainable yield, as the basis for regulating the use of renewable resources (Bothma and Glavovic, 1992; Cowling and Olivier, 1992; Field and Glazewski, 1992; O'Keeffe *et al*, 1992; Verster *et al*, 1992). Third, despite developments in theory, and the large body of evidence regarding the inefficiency and ineffectiveness of direct controls, authorities have tended to adhere to a command-and-control approach in the regulation of renewable resources (Fuggle and Rabie, 1992). Not surprisingly, and in accordance with experience in other countries, direct controls have met with limited success in South Africa. For example, despite the adoption of a plethora of direct controls from the 1950s onward, a number of the country's commercial marine fisheries have been heavily over-exploited over the last four decades (Field and Glazewski, 1992). As a result, the South African government has gone so far as to describe controls governing the use of resources in South Africa as suffering from 'discrepancies, anomalies and ineffectiveness' (African National Congress, 1994, 40).[33]

[32] Including Scott (1955), Crutchfield (1956), Turvey (1957 & 1964) and Crutchfield and Zellner (1963).
[33] Also see International Development Research Centre *et al* (1994).

1.4.2 AIMS AND SCOPE OF THE STUDY

Against the foregoing background, this study has two main aims:

(i) Based on the principles of resource economics developed over the last four decades, this study seeks to develop and apply a bioeconomic model of optimal (that is, dynamic rent maximising) resource use in the South African context.

(ii) Given the results of the modelling process, and by drawing on the substantial body of literature and evidence that has emerged in respect to the management of renewable resources, this study then seeks to provide some guidance as to the appropriate form resource management should take if the goal of dynamic rent maximisation in renewable resource use is to be achieved.

This study is undertaken with reference to South Africa's renewable marine resources and focuses on the country's economically important hake fishery. There are three main reasons for focusing on marine resources and, more specifically, the hake resource. First, South Africa's renewable marine resources are of considerable socio-economic importance (discussed in Chapter 2). Second, in spite of a plethora of controls and regulations, there is considerable evidence to suggest that South Africa's renewable marine resources have been severely over-exploited. Third, while it is important that similar studies be conducted with regard to other renewable resources, this study is confined to an examination of South Africa's hake fishery. This focus is attributable to the fact that the individual features of resources, and sectors of the industry associated with these resources, necessitate 'resource-specific' investigations. This is in line with the view presented in South Africa's draft marine fisheries policy (African National Congress, 1994a, 4), where it is argued that management of fisheries resources should take place on a case-by-case basis. For this reason, the modelling and estimation of optimal resource utilisation strategies and devising of appropriate management policies for all commercial marine fisheries is, necessarily, a considerable task and well beyond the scope of the current study. This study focuses on the hake fishery as it is by far the most economically important renewable marine resource in South Africa, accounting for approximately one-quarter of the commercial catch by mass and one-third by value.

It should be noted that this study is undertaken within the neo-classical framework, an approach that is in line with the main thrust of the theory of renewable resource economics. While it is true that the theory of renewable resource economics has been developed within alternative frameworks – with a particular emphasis being placed on the framework offered by institutional (and organisational) economics – these advances have not been well established.[34] Rather, the bulk of the theory of renewable resource economics remains located within the neo-classical paradigm, as per Gordon (1954) and others. Nevertheless, the insights offered by these alternative frameworks are not ignored in this study. This is particularly true of the latter part of this study which is devoted to providing a set of policy guidelines based on the substantial body of literature and evidence that has emerged out of a host of investigative and theoretical frameworks, with reference to the management of renewable resources.

The remainder of this study is made up of six chapters. Chapter 2 provides an overview of the socio-economic importance of South Africa's renewable marine resources, and examines the evidence relating to the over-exploitation and mismanagement of these resources. Based on the theory of renewable resource economics, Chapter 3 is devoted to providing a theoretical perspective on the issues examined in Chapter 2. This is done through the development of a dynamic bioeconomic model of optimal renewable resource use. It should be noted that while the model developed in Chapter 3 is applicable to a wide range of renewable resources, the nature of modelling demands a sharp focus – that is, resource-specific studies – and, for the reasons given above, this study focuses on the hake resource. However, in order to apply the dynamic

[34] For discussions of alternatives to the neo-classical paradigm see, for example, Panayotou (1982), Berkes (1989), Tvedten and Hersoug (1992), Clay and McGoodwin.

bioeconomic model to the hake resource, it is first necessary to provide some background to the major biological and economic features of the hake fishery to inform the modelling process. This task is undertaken in Chapter 4. A detailed discussion of the modelling procedure and results follows in Chapter 5. This involves two main tasks: first, the dynamic bioeconomic model is reduced to a form in which it is easily estimated, and values of the various biological and economic parameters required in the modelling process are established; and second, having established estimates of optimal use rates, this study turns to a consideration of the implications of bioeconomic resource management for the fishery and, more generally, social welfare.

The dynamic bioeconomic model developed in Chapter 3 and applied in Chapter 5 is based on a number of restrictive assumptions, and Chapter 6 is concerned with the implications of relaxing these assumptions. More specifically, Chapter 6 is devoted to extending the dynamic bioeconomic model into a variable biological and economic parameter setting and examining the impact of variable parameter values on the optimal outcome. Chapter 6 also considers the implications for economic rent (that is, social welfare) of adopting bioeconomic principles as the basis for resource management.[35] Having established optimal outcomes in the hake fishery, Chapter 7 turns to consider the issue of resource management. In particular, this chapter is concerned with two main issues. First, more generally, the chapter seeks to show that direct controls are ineffective and inefficient in the management of renewable resources and that optimal resource management hinges upon the use of market-based or indirect instruments. Second, more specifically, in arguing the advantages of market-based instruments over direct controls, the chapter goes on to explore the problems encountered in applying

[35] As an aside, it should be noted that it is recognised at the outset that while some of the assumptions adopted in the initial stages of the modelling process are later relaxed, the model developed in this study is by no means complete: stochastic aspects are neglected, as are the problems of making decisions based on uncertainty or incomplete information. The modelling procedure also ignores issues of multi-species interaction and multi-harvesting technologies. (These issues are explored more fully in Chapter 3.) None the less, it is argued that the model represents an important first step in the application of the diverse, and often complex, theory of renewable resource use in a South African context.

market-based controls to the hake fishery. In so doing, Chapter 7 seeks to establish a feasible management system for achieving economically efficient, biologically sustainable and socially equitable use of South Africa's hake resource through the adoption of efficient and effective management tools. This objective is in line with the goals of South Africa's draft fisheries policy (African National Congress, 1994a, 4): 'The underlying principle of stock management should be optimal sustainable utilisation, that is stocks should be harvested at levels which optimise the [net] benefits to society without placing them at undue risk'. Chapter 8 is devoted to concluding remarks.

CHAPTER 2: THE SOUTH AFRICAN FISHING INDUSTRY

2.1 INTRODUCTION

Oceanographic conditions along the South African coastline provide a favourable habitat for a wide variety of renewable marine resources, which constitute the basis for the rich and diverse fishing grounds located along the west and east coasts. These fishing grounds, in turn, serve as the backbone of the country's fishing industry, which enjoyed particularly rapid growth over the period 1940-70 due to both supply-side and demand-side forces. Since the mid-1970s, however, the fishing industry has stagnated. There are at least two reasons for this. First, the rapid increase in harvest rates that occurred after 1940, coupled with a regulatory approach that failed to afford adequate protection for stocks, ensured that by the mid-1970s, the bulk of the country's major commercial fish stocks – mainly hake, pilchard and anchovy – had become over-exploited. Second, although a comprehensive set of regulations was in place by the end of the 1970s, these controls have generally failed to reverse significantly the depleted state of fish stocks. Despite these regulations, other stocks, for example west coast rock lobster, have become over-exploited since the mid-1970s.

Given the socio-economic importance of the fishing industry, the failure of regulations to either reverse or prevent over-exploitation raises serious concern, and points to the urgent need to adopt more appropriate regulatory controls. Here, however, a further issue is raised, namely the inappropriate attitude of South Africa's fisheries managers to fisheries regulation and policy formulation. More to the point, irrespective of the effectiveness of controls, South Africa's fisheries managers have based regulations almost exclusively on biological principles. A central argument of this study is that biological principles are an incomplete basis for fisheries management.

This chapter provides a brief overview of the economic history and socio-economic importance of the South African fishing industry. The influence of regulatory controls on South Africa's fisheries is also explored, and explanations for the failure of these controls to reverse or prevent over-exploitation are suggested. The implications of the failure of regulatory controls to reverse or prevent the depletion of stocks are also examined. These concerns, in turn, provide the basis for questioning the appropriateness of formulating fisheries policy primarily, if not exclusively, on the basis of biological principles.

2.2 A Background to the South African Fishing Industry

Oceanographic conditions along the South African coastline provide a favourable habitat for a wide variety of living marine resources. Along the west coast, in particular, upwelling of cold, nutrient-rich water together with a wide continental shelf provides for an environment that supports the extremely rich fishing grounds of the south-east Atlantic (Shillington, 1986, 22-25).[1] Coupled with the influence of a number of supply-side and demand-side forces, these highly productive fishing grounds served as the basis for the rapid growth that took place in the South African commercial fisheries over the period 1940-70.

On the supply side, the relatively large (and unexploited) commercial fish stocks, located primarily on the west coast, provided an ideal base for growth in the fishing industry; and growth in output was spurred by improvements in techniques for finding, harvesting, processing and transporting fish that

[1] Upwelling is the process by which cold marine water from 100-meter depths rises to the surface and is driven far from the coast by horizontal movements. These waters are rich in nitrate and phosphate nutrients, and the areas of upwelling often support massive blooms of phytoplankton. These blooms start the biological food chain with zooplankton feeding on the phytoplankton, and pelagic filter-feeding fish preying on the zooplankton (Shillington, 1986, 24). Although only 0.1 percent of the earth's oceans consist of upwelling systems, these systems contribute approximately 50 percent of the world's fish catch (World Resources Institute, 1994, 185).

occurred from the late 1930s onwards. On the demand side, it was initially the search for food, brought about by the onset of the Second World War, that gave the fishing industry the necessary boost. In particular, it was the inception of the pilchard fishery in the early 1940s, which saw the pelagic catch increase from just 7 300 tons in 1943 to 195 000 tons in 1958, that gave the industry this primary boost (Stuttaford, 1995, 20).[2] These early developments in the South African fishing industry are well summarised by Payne and Crawford (1989, 50):

> In the 1940s, World War II brought a demand for canned fish and provided an impetus for purse-seine fishing. Nevertheless, the total harvest from southern Africa's seas remained relatively small until 1950. Then, technological advances, which allowed fishermen to operate farther from shore than previously, to plot their courses with greater certainty, to locate unseen fish accurately, to communicate with each other and with 'spotter' planes regarding the presence or absence of fish, to handle the fishing gear more easily, to bring the catch aboard and to offload it in a minimum of time, even to process fish at sea, all led to rapid increases in the quantity of fish caught.

Later, demand was bolstered by the opening up of new domestic (inland) and export (chiefly the United Kingdom and United States of America) markets. Concurrently, population growth, coupled with on-going increases in *per capita* incomes, served to provide further momentum on the demand side. Moreover, growing demand spurred some fishers on to seek out new fish stocks; and these stocks provided an additional boost on the supply side. The net result of these forces was that total commercial landings increased from 71 000 tons to 422 480 tons between 1938 and 1958 (Stuttaford, 1995, 20); and although over-fishing resulted in a collapse of the pilchard fishery in 1963 (see Section 2.4.1), a switch to other pelagic stocks (anchovy and, to a lesser extent, chub mackerel and redeye), coupled with advances in technology, ensured that the pelagic fishery continued to grow through the 1960s and 1970s (albeit at a considerably slower pace).

By 1978, the pelagic catch stood at just under 400 000 tons (an increase of 20 000 percent over the 1938 level) (Grindley, 1969; Stuttaford, 1995, 20). Furthermore, the exceptional growth in the pelagic catch was supported by

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[2] Pelagic fish are fish species (such as anchovy, pilchard, redeye and mackerel) occurring in schools near the surface of the sea which are usually caught with purse seine nets.

more modest (but less erratic) growth in the demersal fishery.[3] Between 1938 and 1978, the demersal catch grew from 23 630 tons to 186 990 tons: a seven-fold increase in catch over the four decades. As shown in Table 2.1, the net result of these various forces and events was an increase in total commercial landings from 71 000 tons to 635 900 tons between 1938 and 1978: an almost ten-fold increase in just four decades.

Since the late 1970s, however, growth in the commercial catch has stagnated, with total catch fluctuating between 500 000 and 700 000 tons per annum (with the exception of the years 1987 and 1988 when the catch reached 919 173 tons and 898 561 tons respectively).[4] In spite of this stagnation in the total commercial catch, the earlier substantial increase in the commercial catch documented above has resulted in South Africa emerging as an important global player in the supply of fish: the country is formally defined as 'one of a group of 22 medium-sized fishing countries' (Van D. Boonstra, 1993, 1), and accounts for roughly 0.5 percent – and in some years over 1 percent – of the global catch. Table 2.1 provides a summary of the growth in South Africa's commercial catch from 1938 to the present, as well as the contribution of South Africa's commercial catch to the total world catch over the same period.[5]

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[3] Demersal fish are fish species (such as hake, kingklip, snoek and sole) occurring in the deeper water layers near the bottom of the sea which are usually caught with trawl nets.

[4] It is worth noting that the rapid growth in South Africa's commercial catch through the 1940s, 1950s and 1960s, followed by stagnation in growth rates in the 1970s and 1980s, closely mirrors trends in the world catch (World Bank, 1992, 11).

[5] Reference here is exclusively to marine catches, because although South Africa does have a freshwater fishing industry, the industry is very small, contributing just 2 300 tons to the total catch in 1990 (World Resources Institute, 1994, 352).

Table 2.1: South Africa's Commercial Catch and Contribution to World Catch (1938-94)[6]

Year	Tons	Percentage of World Total
1938[7]	71 000	0.35
1948	181 790	0.91
1958	422 480	1.08
1968	549 510	0.90
1978	635 900	0.91
1983	587 823	0.76
1985	595 838	0.69
1986	622 959	0.67
1987	919 173	0.98
1988	898 561	0.91
1989	652 636	0.66
1990	545 717	0.56
1991	589 979	0.61
1992	691 936	0.70
1993	596 069	0.59

Source: Adapted from World Bank (1992, 12); Stuttaford (1995).

The commercial catch is dominated by two large fishery sectors: the demersal (or trawl) fishery and the pelagic (or purse seine) fishery. Collectively, these fisheries account for roughly 90 percent of the reported annual catch.[8] The trawl fishery is the more stable (evidenced by a coefficient of variation on total catch that is one-tenth the size of that recorded by the pelagic fishery over the ten-year period 1985-94),[9] and is dominated by hake which typically accounts

[6] Total world catch includes the marine and freshwater fisheries as well as aquaculture; marine catches account for about 80 percent of the total world catch (World Resources, 1994, 332).

[7] Accurate figures on the size of the catch prior to 1938 are not available. However, based on various sources (Gillchrist, 1914; Scott, 1950; Stoops, 1953; Gertenbach, 1973), it is reasonable to suggest that the total South African catch was of the order of 35 000-45 000 tons in 1928, from where it grew steadily to reach 71 000 tons in 1938.

[8] Between 1975 and 1995 the demersal and pelagic fisheries, collectively, never contributed less that 88 percent of the total catch and have accounted for as much as 95 percent of the catch on occasion (Van D. Boonstra, 1993, 1).

[9] See Table 2.2.

for around 65 percent of demersal landings; but horse mackerel, chub mackerel, snoek, monkfish and ribbonfish are important by-catch species (Van D. Boonstra, 1993, 39). The purse seine fishery is dominated by anchovy, pilchard and redeye which, over the period 1985-94, accounted for 73.8 percent, 11.7 percent and 11.4 percent of pelagic landings respectively.[10] Other important fisheries include the small but valuable crustacean fisheries (mussels, oysters, abalone and rock lobster) and the tuna, squid-jigging and handline fisheries, which collectively contributed an average 4.9 percent to the total commercial catch over the period 1985-94. Table 2.2 provides a breakdown of the nominal catch by sector for the ten-year period 1985-94.

2.3 An Overview of the Economic Importance of the South African Fishing Industry

The rapid growth in commercial fishing recorded since the start of the Second World War has resulted in the South African fishing industry emerging as an important contributor to economic activity. The industry makes a significant contribution to gross domestic product, and is a substantial generator of employment, income and wealth. The industry is also responsible for a large portion of private sector investment and is a material earner of foreign exchange. Moreover, the industry has strong spill-over effects, providing opportunities in related industries such as boat-building and maintenance. Finally, but most obviously, the catch taken by the fishing industry serves as an important source of nutrition for humans and animals. These points are detailed below.

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[10] However, due to a collapse in biomass, the anchovy catch has recently fallen dramatically, reaching 41 000 tons in 1996 and set at zero for 1997 (*Cape Business News*, 1997a).

Table 2.2: Nominal Catch (Tons) by Sector of the South African Commercial Sea Fisheries (1985-94)

	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	Standard Deviation	Mean	Coefficient of Variation (%)[1]
Trawl:	186 040	197 043	210 193	200 515	215 725	240 506	217 863	212 046	211 885	187 748	16 049	207 956	67.72
Hake	139 889	146 473	145 636	138 488	133 850	134 821	135 220	134 788	141 202	144 045	4 776	139 441	3.43
Other	46 151	50 570	64 557	62 027	81 875	105 685	82 643	77 258	70 683	43 703	19 254	68 515	28.10
Purse seine:	377 464	392 827	674 092	671 415	403 375	259 343	247 067	451 987	357 040	315 095	149 834	414 971	36.11
Pilchard	32 986	35 553	40 644	37 505	34 564	56 740	51 948	53 407	50 717	93 438	17 990	48 750	36.90
Anchovy	272 642	299 596	593 278	565 055	294 101	150 100	150 560	347 456	235 830	155 553	159 400	306 417	52.02
Redeye	39 871	56 917	34 471	61 251	44 360	44 710	33 484	47 440	56 331	54 147	9 670	47 298	20.44
Other	31 965	761	5 699	7 604	30 350	7 793	11 075	3 684	14 162	11 962	10 596	12 506	84.73
Rock lobster:	4 210	4 870	5 924	4 739	4 604	4 544	3 899	3 483	3 194	3 265	845	4 273	19.78
West	3 728	3 796	4 015	3 834	4 000	3 491	2 996	2 480	2 176	2 198	744	3 273	22.75
South/East	482	1 074	1 909	905	604	1 053	903	1 003	1 018	1 067	378	1002	37.69
Line/Small net:	18 336	19 728	21 221	16 764	22 979	24 717	17 665	21 336	21 857	23 496	2 631	20 810	12.64
Other :[2]	9 784	8 491	7 743	5 128	5 953	15 517	3 485	3 084	2 093	14 321	4 593	7 560	60.76
Total	595 834	622 959	919 173	898 561	652 636	545 717	489 979	691 936	596 069	543 925	145 292	655 679	22.16

Source: Adapted from Van D. Boonstra (1992, 1993, 1994 & 1995); Stuttaford (1995).

[1] The coefficient of variation is calculated as the percentage ratio of the standard deviation to the mean.
[2] Includes, *inter alia*, prawn, langoustine, abalone, mussels, oysters, guano and seaweeds.

The wholesale value of commercially exploited marine resources amounts to approximately 0.5 percent of South Africa's gross domestic product (Figure 2.1 provides a breakdown of the wholesale value for the major sectors of the commercial sea fishery).[11] Furthermore, fishing is a relatively labour-intensive activity. The South African Deep-Sea Trawling Industry Association (SADSTIA) estimated that, in 1994, the industry provided direct employment for about 35 000 people in the formal sector, with an estimated 8 000 people employed in harvesting and 27 000 people employed in processing and distribution (SADSTIA, 1994, 8). The fishing industry has strong spill-over effects, and provides substantial indirect employment through related activities such as boat-building and maintenance. Accurate figures on the extent of spill-over effects are not generally available.[12] However, Marais (1997) estimates that the fishing industry provides indirect employment for 100 000 people which is at least indicative of the magnitude of the fishing industry's multiplier effect.

The fishing industry is also responsible for considerable investment: the licensed fishing fleet consists of approximately 4 000 vessels (Van D. Boonstra, 1993, 7-8).[13] The bulk of these vessels (about 70 percent) fish out of ports and harbours located along the west coast of South Africa. Of the remainder, 25 percent of the vessels are located on the south and east Cape coasts and 5 percent on the KwaZulu-Natal coast. A very small portion of the fleet (less than 0.3 percent) fishes out of Walvis Bay (Namibia) (Van D. Boonstra, 1993, 8). The fleet supports an important post-harvest sector that is involved in value-adding activities such as handling, processing, packaging,

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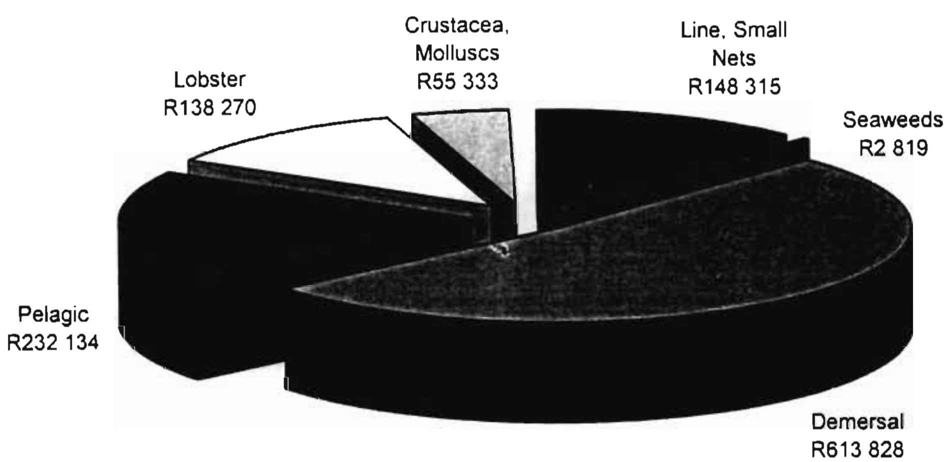
[11] It should be noted that the contribution made by the fishing industry to the South African economy has been heightened since the late 1970s when the industry contributed, on average, one-quarter of a percent to gross domestic product. This increase in the contribution to gross domestic product is primarily attributable to the replacement of foreign fishing fleets with domestic fleets over the course of the 1970s. For instance, foreign trawlers accounted for over 60 percent of the hake catch (commercially the most important species) in South African waters in 1972. However, by 1976, the figure had dropped to 45 percent and by 1978 had become insignificant (Stuttaford, 1995, 11).

[12] Pers. comm. (Bross, SADSTIA).

[13] This is substantially less than the 7 816 vessels that were employed when the fleet size peaked in 1969. However, increases in fleet capacity have more than compensated for reductions in the number of vessels. See, for example, Newman *et al* (1979, 4).

storing and distributing fish and fish products. Accurate figures are not available on the value of the investments in fleet and plant, but figures drawn from the 1975 and 1979 fishery censuses would suggest that the investments made in the fishing industry account for about 0.25 percent of total fixed investment in South Africa (Central Statistical Services, 1993, 10.5; South African Reserve Bank, 1980, S96).

Figure 2.1: Wholesale Value by Sector (R'000s) of the South African Commercial Sea Fisheries (1993)



Source: Adapted from Stuttaford (1995, 28).

The fishing industry is also an important creator of wealth. For example, virtually all of the companies which are listed on the Johannesburg Stock Exchange that are involved in the South African fishing industry (the exceptions being ICS Holdings Limited, the diversified parent company of Sea Harvest Limited, and the KwaZulu-Natal based Natal Ocean Trawlers Limited) have Q-Ratios that are well above the average for the top 132 listed industrial companies.[14]

[14] Tobin's Q-Ratio measures the ratio of a firm's market value to the value of the firm's debt and, as such, is a measure of wealth creation. A ratio greater than one indicates creation of wealth for shareholders; and the higher the ratio the greater the degree of wealth creation. The mean ratio for the top 132 industrial companies was 1.24 for 1994 (Financial Mail, 1995, 114-120). In contrast, the average ratios for the listed companies involved in the South African fishing industry over the three years 1992-94 were: Sea Harvest Corporation Limited (2.68), Irvin and Johnson Limited (1.83), Foodcorp Limited (1.44) and Oceana Fishing Limited

The fishing industry is also an important earner of foreign exchange, accounting for just under 1 percent of South Africa's total export earnings (equal to R815 million in 1994). More significantly, exports of fish and fish products consistently exceed imports – in 1994 by a factor of eight – to the extent that fish and fish products account for just over 5 percent of South Africa's trade surplus (Monthly Abstract of Trade Statistics, 1994, 248-99) The most important export products are frozen fish and fish fillets (primarily hake), rock lobster and molluscs. Collectively, these products account for about 80 percent of the fishing industry's exports. The most important import products are frozen fish (especially horse mackerel) and crustaceans (chiefly shrimps and prawns), which collectively account for roughly 65 percent of imports of fish and fish products (Monthly Abstract of Trade Statistics, 1994, 248-99).[15] It should be noted that Stuttaford argues that South Africa's trade figures for fish exports should be viewed with caution because categories for some important export products (such as abalone) do not exist.[16] Stuttaford suggests that import figures are largely correct, due to the fact that Customs and Excise derive revenue from imports and so are far more concerned with the accuracy of these figures. None the less, the broad thrust of the argument still holds, namely that exports have grown substantially (particularly over the last five years), with the result that exports now outstrip imports by a considerable margin. Figure 2.2 summarises the trend in exports and imports of fish and fish products for the five years 1990-94.[17]

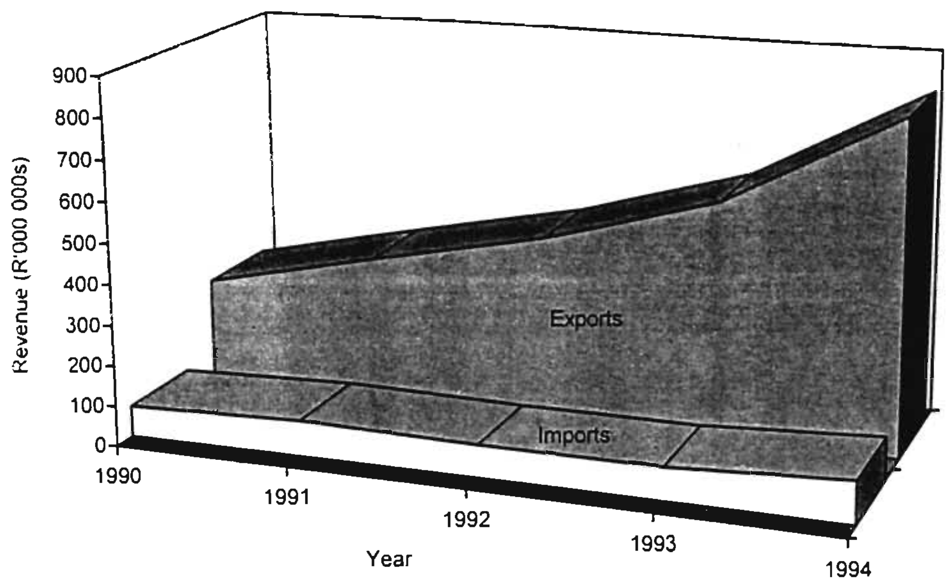
(1.40). The ratio for ICS Holdings Limited (1.12) is below the three year average for the top 132 industrial companies, and that for Natal Ocean Trawling Limited is not available, but is presumably substantially less than one as the company was provisionally liquidated at the end of 1994 (Jones, 1995).

15 Figures detailing the fishing industry's expenditure on imports of machinery and equipment are not available.

16 Pers. comm. (Stuttaford, Marine Information Close Corporation).

17 The industry also earns foreign exchange through the provision of maintenance and repair facilities. For example, it is estimated that Japanese and Taiwanese fishing vessels calling at South African ports for maintenance and repairs, contribute more than R340 million (1995 prices) in foreign exchange annually (D'Angelo, 1996).

Figure 2.2: Exports and Imports of Fish and Fish Products (1990-94)



Source: Adapted from Monthly Abstract of Trade Statistics (1992, 1993 & 1994).

In addition to commercial fishing, South Africa's fish stocks also provide the basis for a large recreational fishing sector. The warm east coast Agulhas current system supports a wide diversity of species, and this sustains the substantial recreational fisheries located primarily along the KwaZulu-Natal and Eastern Cape coasts.[18] Considering first the recreational shore fishery, approximately 750 000 people participate in recreational angling each year and spend about R750 million (1995 prices) in the process. The catch is estimated to be about 17 000 tons per annum. Deep-sea recreational anglers spend approximately R23 million pursuing their sport and the capital investment in boats and equipment is conservatively estimated at about R1.4 billion (although a large portion of the investment component is imported equipment) (Row, 1996).[19] Moreover, the recreational fishery provides

[18] The warm east coast Agulhas current system is low in nutrients and oxygen and does not support as much fish life as does the west coast Benguela current system. However, there is a greater diversity of species in the warm east coat waters (Shillington, 1986, 25) which provides the basis for the recreational fisheries which are located along the east coast.

[19] It is accepted that it is extremely difficult to establish the size of recreational fisheries (Guastella and Nellmapius, 1993); even where relatively good reporting systems exist, as in the KwaZulu-Natal fishery. Accordingly, estimates of the size of the recreational fishery are very loose. Rorke (1996), for example, estimates the size of capital investment in deep-sea recreational angling to be at least three times greater than the figures cited above, and estimates expenditure by those anglers to be well in excess of R100 million per annum.

support for South Africa's important – and rapidly expanding – tourism industry. It is estimated that the coastal zone accounts for about 35 percent of tourism and, as such, contributes just over 1 percent to gross domestic product.[20] Moreover, the tourism industry is highly labour-intensive, with coastal tourism directly employing as many as 150 000 people,[21] and this section of the tourism industry earns R2.5 billion in foreign exchange annually (*Business Day*. 1995). A large, but unquantified, informal fishing sector also exists.

In addition to the above, fish are an important source of nutrition for humans and animals (Smith and Wick, 1983, 25; Frankel, 1988, 162). For example, *per capita* consumption of fish (on a global scale) grew steadily from 6.7 kilograms per annum in 1950 to over 13 kilograms per annum by 1990 (World Bank, 1992, 12). Consumption of fish in South Africa currently stands somewhere between 5 and 7 kilograms per person per annum (Payne and Crawford, 1989, 50; Stuttaford, 1995, 104-5), but may rise if *per capita* incomes grow and there is an increased awareness of the perceived health benefits to the individual of eating more fish. Domestic demand is also likely to receive a further boost from population growth. Moreover, the industry is an important supplier of feedstock for poultry and other animals: approximately 55 percent of the total catch is processed into fish meal (Stuttaford, 1995, 28). It is expected that international demand for fish will continue to grow at such a rate that demand will outstrip supply by some 30 million tons by the end of this decade (World Bank, 1991, 4).[22] Thus it is likely that growth in domestic demand, coupled with excess demand in international markets, will result in an ongoing escalation in the economic importance of the South African fishing industry.

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[20] Pers. comm. (Bester, Econometrix Limited).

[21] Pers. comm. (Kohler, Deloitte and Touche Consulting Group Limited).

[22] The World Bank expects the total catch to remain static at current levels – about 100 million tons per annum – over the next five years, but expects demand to increase by 30 million tons to around 130 million tons per annum over the same period (World Bank, 1991, 4-5).

The trend toward increased economic importance is likely to be augmented by growth in value-adding activities: a large portion of the South African catch is processed as low value-added non-food products (such as fish meal and fish body oil). For instance, in the pelagic fishery, approximately 90 percent of the catch (equal to 55 percent of the total commercial catch by mass) is processed as fish meal and fish body oil. Fish meal and body oil has a wholesale value approximately twice as great as the landed value of the catch; the remaining 10 percent of the pelagic catch is processed into canned fish products, mainly for human consumption, which has a wholesale value sixteen times as great as the landed value of the fish (Crawford, 1981, 6; Stuttaford, 1995, 28).

In summary, the South African fishing industry is of substantial socio-economic importance: the industry contributes to gross domestic product and is an important employer and investor. Furthermore, the fishing industry accounts for a significant portion of the country's foreign exchange earnings. Moreover, the harvesting sector supports a large post-harvest sector which is involved in value-adding activities such as handling, processing, packaging, storage and distribution. In addition, the industry has spill-over effects into other activities such as boat-building and maintenance. The fish stock also serves as the basis for the commercially important recreational fishery which is responsible for substantial investment, and provides support for South Africa's large (and growing) tourism industry. Furthermore, the industry is an important supplier of nutrition for both humans and animals. Finally, the socio-economic importance of the South African fishing industry is likely to increase as the demand for fish and fish products is spurred by domestic and international growth in population and *per capita* incomes, as well as shifts in consumption patterns towards fish and fish products, and shifts in production patterns towards value-added products. Accordingly, it is vital that South African fisheries are managed on an appropriate basis. To date, however, it is questionable whether this has been the case. South African fisheries have been subject to a number of problems, relating to the management, regulation and operation of the fisheries, which have threatened – and continue to threaten – the future of the fishing industry. These problems are explored in the following section.

2.4 Fisheries Problems and Policies

There are at least three main sets of problems that can be identified in connection with the management, regulation and operation of South African fisheries. First, historically, South African fisheries have been subject to biological over-exploitation, that is to say, stocks have been over-fished to the extent that the yield (growth) of the biomass is below the biologically determined maximum sustainable yield.[23] Second, and in the face of over-exploitation, numerous efforts have been made to reduce the rate at which stocks are harvested. However, in many cases, these efforts have failed to reverse the effects of over-exploitation, with a large proportion of stocks remaining in a severely depleted state. A number of factors have contributed to this failure. Specifically, regulations have promoted higher levels of concentration of ownership within and between the harvesting and processing sectors. In turn, higher levels of concentration have provided incentives for illegal activity, and this has exacerbated problems of over-exploitation. Stock management has been further undermined by an inadequate policing system which has been unable to prevent illegal activity, such as poaching. Third, and of greatest significance, irrespective of the relative success or failure of regulations, the management of South Africa's commercial fisheries has been based primarily upon biological principles. The central theme of this study is that biological principles are only a partial (and therefore incomplete) basis for the management of fisheries. These points are discussed in greater detail below.

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[23] The concepts of maximum sustainable yield and biological over-exploitation are explored more fully in Chapter 3.

2.4.1 Over-Exploitation and the Regulatory Response

During the early part of this century, fishing activity in South Africa was largely unregulated. Before the end of the Second World War, the most common management approach adopted by the country's fisheries managers was one of open access (Kantey, 1992, 33-4).[24] This management approach was based on the traditional concept of fisheries as a resource which all people had the right to harvest, and relied on the market forces of supply and demand to regulate the rate of exploitation of fish stocks. However, the period 1940 to 1975 witnessed a number of developments which ultimately led to the replacement of the *laissez faire* approach with an interventionist approach based on direct controls.

As already noted, the late 1930s saw the onset of a period of rapid growth in South Africa's commercial catch, which was fuelled by the inception of the highly lucrative pilchard fishery, and spurred by steady growth in the trawl catch. The immediate result was an unprecedented influx of participants into the fishing industry, with most of the new entrants being attracted to the less capital-intensive purse seine sector. Gertenbach (1973, 2081), for example, wrote of teachers and doctors moving into a booming fishing industry by acquiring boats during the 1940s and early 1950s. This influx of participants led to a substantial increase in the rate of exploitation of fish stocks, especially pilchard. In turn, the rapid increase in the rate of exploitation of fish

[24] The earliest attempts to protect South Africa's living marine resources from over-exploitation date back to the 1700s: *Placaaten* of 20 October 1709 and 29 September/1 October 1792 served to prohibit seal hunting on Dassen Island and control whaling in Table Bay and False Bay respectively (Rabie and Fuggle, 1992, 14). The latter part of the nineteenth and early part of the twentieth century were witness to substantial growth in legislation aimed at natural resource conservation. The newly established legislature of Natal, following the example of the Cape legislature, passed game laws at regular intervals – including Cape Acts 7 of 1883, 29 of 1890 and 43 of 1899, and Natal Laws 8 of 1868, 13 of 1880 and 21 of 1884 – aimed at the conservation of fish stocks (Rabie and Fuggle, 1992, 14). After Union in 1910, the management of fisheries initially remained the responsibility of individual provinces. However, the Sea Fisheries Act No. 10 of 1940 brought the regulation of fisheries under the control of the Union government. Being aimed principally at marketing, the Act had little conservation emphasis, although the Act did allow for protection of stocks by way of closed seasons, closed areas and minimum length and mesh size. However, these regulations were initially implemented to an extremely limited extent (Scott, 1950). See Stander (1995) for a more comprehensive review of the history of fisheries regulation in South Africa.

stocks raised concerns within government about over-fishing. In this regard, Stoops (1953, 246) noted:

> The spectacular earnings of the West Coast fisherman have led to the entry of so many new craft into the industry that, as previously mentioned, consideration is being given to restrictions of numbers in the interests of conservation.

These fears were further fuelled by the 'high profile' collapse of the commercially important Canadian pilchard fishery after the total catch fell from 791 000 tons in the 1936/37 fishing season to just 3 220 tons in the 1952/53 season (Stoops, 1953, 243). Ultimately, this amalgam of concerns and evidence led to the introduction of various regulations in the South African purse seine fishery during the first half of the 1950s. Regulations aimed at limiting entry (licences for vessels and restrictions on the numbers and capacity of processing plants), as well as effort and catch (limited fleet hold capacity, limited quotas, minimum mesh size and closed seasons).[25] However, these regulations came too late to prevent the collapse of the pilchard stock, and between 1958 and 1968 the catch fell by over 50 percent from 194 620 tons to just 94 000 tons (Stuttaford, 1995, 20). A similar collapse occurred in the Cape horse mackerel fishery, where the catch fell by 70 percent between 1958 and 1968 (Stuttaford, 1995, 20). However, fears of over-exploitation in the pelagic fishery were allayed by the almost immediate replacement of the pilchard and horse mackerel catches with anchovy, redeye and chub mackerel (see Figure 2.3). Furthermore, in the trawl fishery, the catch increased steadily between 1958 and 1968 from 103 260 tons to 138 550 tons, and by 1978 had reached 186 990 tons (Stuttaford, 1995, 20).

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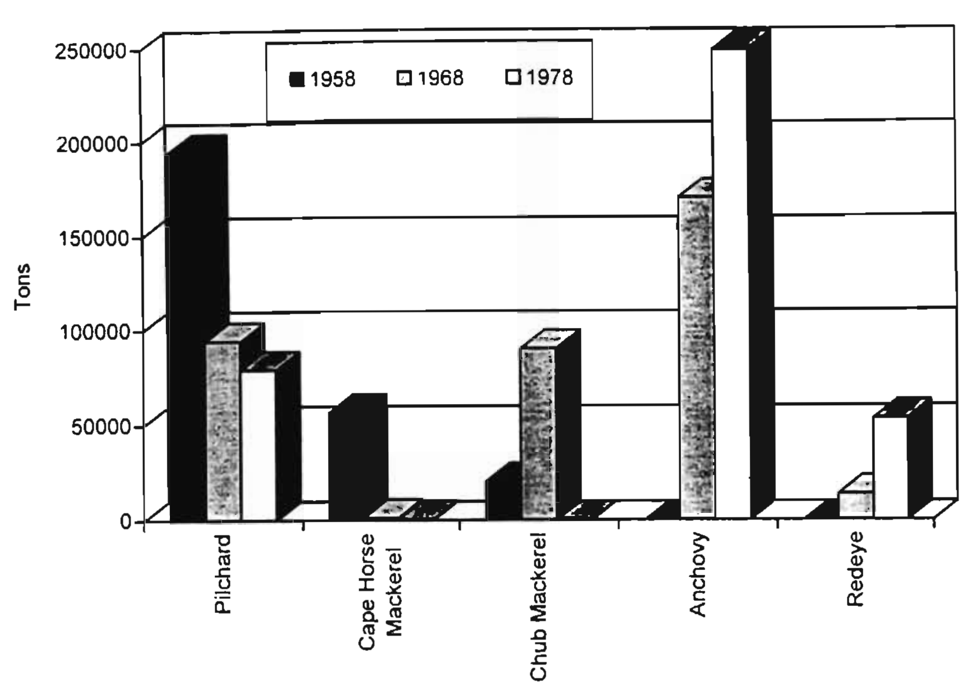
[25] A well-documented history of the early evolution of regulations in the purse seine fishery is provided in Gertenbach (1973). See also Newman *et al* (1979) for a concise summary of the early history of restrictions.

Figure 2.3: Nominal Catch (Tons) in PelagicFisheries (1958-78)



Source: Adapted from Stuttaford (1995, 20).

Notwithstanding the increase in certain stocks, two further developments took place through the 1960s and 1970s which ensured that, by the end of that period, the bulk of South Africa's commercial fisheries had been over-exploited. First, continued technological advancement – which included the introduction of the hydraulic winch and echo sounder in the early 1960s, and the power block, sonar and fish pump later that decade – facilitated higher rates of exploitation of stocks. Second, foreign fishing fleets became significant harvesters of South African stocks through the 1960s and early 1970s. During the 1950s, the bulk of the catch in the south-eastern Atlantic was taken by the fleets of the three southern African nations (South Africa, Namibia and Angola) (Grindley, 1969, 72). By the early 1970s, however, foreign fleets had become heavy exploiters of domestic stocks. For example, during 1970-75, foreign fleets accounted for over 50 percent of the hake catch in South African waters (Stuttaford, 1995, 11).[26] The entrance of

[26] It should be noted that South African vessels are also responsible for having over-exploited stocks in the waters of other countries. For example, until Namibia's independence in 1990, inshore fishing and processing was dominated by a cartel of South African-owned firms which left behind a legacy of over-fishing, in particular, of pelagic stocks (Payne, 1989, 140; Payne

foreign fleets had two main effects: (i) the catch available to South African fleets was reduced, and this served to undermined the feasibility of the domestic fishing industry; and (ii) there is also evidence to suggest that distant water fleets are less concerned with the management of fish stocks than domestic fleets which, in turn, suggests that the presence of foreign fleets in South African waters resulted in accelerated rates of exploitation (Payne and Punt, 1995, 17-21).[27]

The combined effect of the above three forces – rapid growth in the rate of exploitation, which resulted in a collapse of stocks including the pilchard and mackerel stocks; continued exploitation of other stocks at a higher rate facilitated by technological improvements; and heightened participation by foreign fleets in the South African fishing industry – was that by the mid-1970s, South Africa's major commercial fish stocks were generally considered to be in a state of over-exploitation. This was evidenced, in the first instance, by falling catch per unit effort.

Catch per unit effort is typically considered to be a first guide to the state of stocks, where catch per unit effort is measured as the total catch divided by the level of fishing effort required to take the catch. All else constant, it is assumed that increases in effort produce proportionate increases in catch, and that a declining catch per unit effort is evidence of stock depletion.[28] Based on this reasoning, it can be shown that the bulk of South Africa's

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and Punt, 1995, 18; Economist Intelligence Unit, 1996, 22). South African fishers have also been identified as responsible for depleting stocks in Moçambican waters (Rostami, 1990; Seery, 1995). Furthermore, South African fishers have been responsible for over-exploitation of stocks in international waters. For example, in 1965, the Vema Seamount (a submerged isolated pinnacle, located 1 000 kilometres north-west of Cape Town) was found to be rich in rock lobster, but was exploited by vessels of several nations, including South Africa, to such an extent that by 1978 divers could not find a single lobster (Grindley, 1969, 89; Shillington, 1986, 23).

27 Similar evidence exists for other countries. For example, the highly publicised recent dispute between the Canadian government and Spanish trawlers over the rights to stocks located on the Grand Banks fishery provides an excellent case in point (Cleroux,1995; Usher, 1995); and similar evidence is available for fishing by foreign fleets in Chilean waters (Le May, 1995e).

28 Measures of catch per unit effort vary substantially through fisheries. For example, in the hake trawl fisheries, catch per unit effort is measured in tons per day in the west coast fishery, but tons per hour in the south coast fishery; and catch per unit effort for longlining is measured in kilograms per 1 000 hooks.

commercial fish stocks had been driven into an over-exploited state by the mid-1970s. As an example of over-exploitation, catch per unit effort figures for the most important components of the trawl fishery, the west and south coast hake fisheries, are available from 1955 and 1969 respectively; and the evidence is unambiguous. In both fisheries, catch per unit effort fell substantially from early highs. For instance, by the mid-1970s, catch per unit effort in the west coast hake fishery stood at a quarter of the 1955 level; and a similar collapse in catch per unit effort was recorded in the south coast hake fishery between 1969 and 1975 (Punt, 1994, 161-2).[29] To be more specific, while the total catch in both the west and south coast hake fisheries climbed over these respective periods – in the west coast fishery, the catch more than doubled between 1955 and 1972 from 115 000 tons to 244 000 tons; and in the south coast fishery, the catch increased from 41 000 tons to 100 000 tons between 1969 and 1974 – the increase in effort required to take these higher catches was disproportionately greater than the increase in the catch (Leslie, 1994). Figure 2.4 provides a summary of catch per unit effort in the hake fisheries from 1955 onwards.[30]

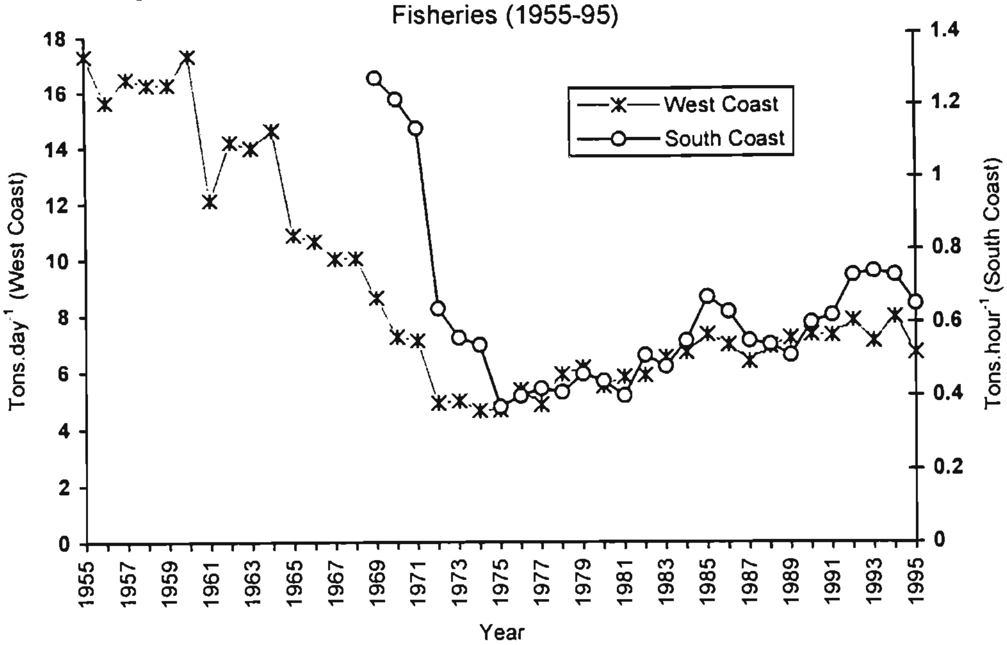
Evidence of over-exploitation of pelagic stocks is also available. For example, catch per unit effort in the purse seine fishery declined by almost 60 percent between 1964 and 1972. Furthermore, although catch rates recovered marginally in the period 1972-76, they remained well below the levels recorded in the mid-1960s. Figure 2.5 provides a summary of catch per unit effort in the pelagic fishery for the period 1964-76.[31]

[29] The total catch in both the west and south coast hake fisheries climbed over these respective periods. In the west coast fishery, the catch more than doubled between 1955 and 1972 from 115 000 tons to 244 000 tons; and in the south coast fishery, the catch increased from 41 000 tons to 100 000 tons between 1969 and 1974. The implication is that a massive increase in effort was required to boost the total catch in the face of a falling biomass.

[30] Catch per unit effort data are available from 1955 onwards for the west coast fishery, and from 1969 onwards for the south coast fishery (Punt, 1994, 162). More recently, serious concern has been expressed regarding the accuracy of catch per unit effort data in the hake fisheries (Bergh and Barkai, 1996). This point is explored in greater depth in Chapter 4.

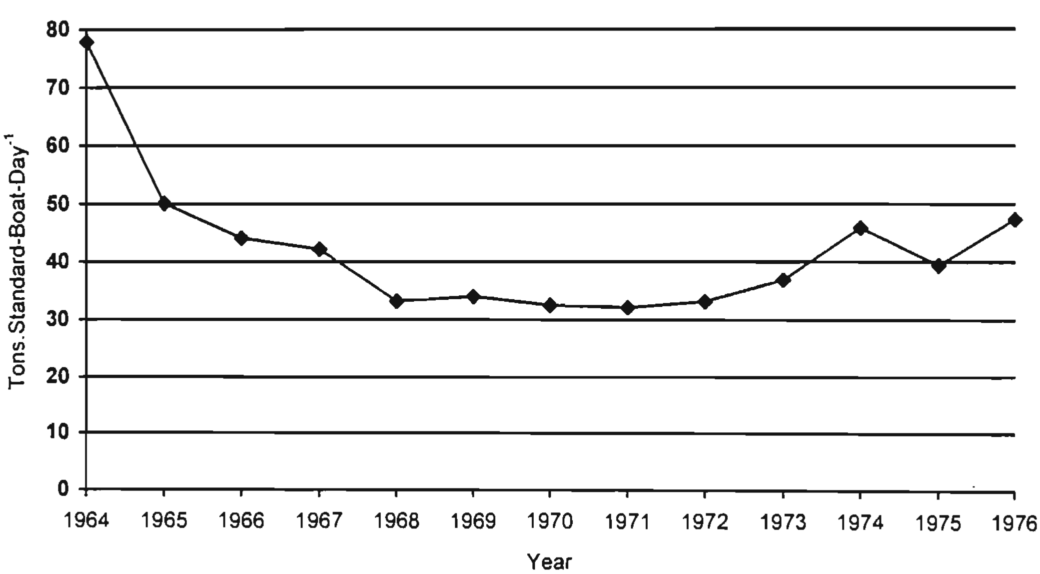
[31] Statistics are available from 1956 onwards, but are only reasonably comprehensive from 1964. For reasons given later, catch per unit effort fell into disuse as a guide to stock levels in the pelagic fishery after the mid-1970s. Accordingly, accurate catch per unit effort figures are only available for the years 1956-76.

Figure 2.4: Catch Per Unit Effort in the West and South Coast Hake Fisheries (1955-95)



Source: Payne and Punt (1995, 19).

Figure 2.5: Catch Per Unit Effort in the Purse Seine Fisheries (1964-76)



Source: Newman *et al* (1979, 23).

In short, falling catch per unit effort in the trawl and purse seine fisheries suggests that, by the mid-1970s, both demersal and pelagic stocks could be considered to have been in a depleted state due to over-fishing.[32] It should be noted, however, that the validity of catch per unit effort as a guide to the state of pelagic fisheries is questionable. The reason for this is that where stocks are relatively evenly spread across a fishing ground, as is generally the case with demersal stocks, it is reasonable to expect catch to increase proportionately with effort. In these circumstances, catch per unit effort can be taken as a good first guide to the state of stocks. However, this is not the case for species that have a restricted distribution or are encountered sporadically, such as pelagic species which occur in schools. More to the point, once a pelagic stock has been located, it requires the same amount of effort to take the available catch, irrespective of the state of the overall biomass. Thus, the only major variable in total fishing effort is the level of effort required to locate stocks; and whilst it is generally true that greater search times are required to locate depleted stocks, improvements in technology have substantially reduced the search effort needed to locate, for example, pelagic stocks.[33] In addition, the argument against the use of catch per unit effort as a guide to the state of pelagic stocks is strengthened by the fact that, particularly in South Africa's pelagic fishery, changes in boat size and improvements in technology have meant that, *ceteris paribus*, it has become increasingly easy to catch and transport fish. These advances in technology have served to distort measures of catch per unit effort.[34] Attempts have been made by, *inter alia*, Newman *et al* (1979, 6) and Crawford (1981) to overcome these distortions by generating catch per unit effort based on 'standard' effort, that is, effort adjusted for technological improvements.[35] Such adjustments, however, should be viewed with caution, as 'standardisation' of effort data is typically a complex and inexact task.[36]

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[32] This argument is given further support by the fact that no less than four commissions of inquiry reported on the state of the South African fishing industry between 1972 and 1986 (Rabie and Fuggle, 1992, 21).

[33] Pers. comm. (Leslie, Sea Fisheries Research Institute).

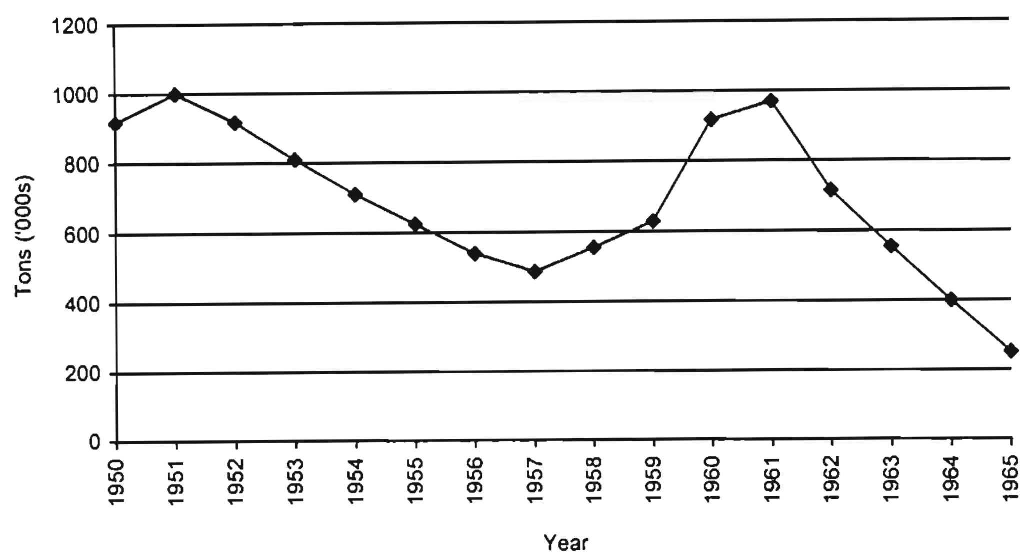
[34] Pers. comm. (Van D. Boonstra, Sea Fisheries Research Institute).

[35] Technology has played a less important role in influencing catch rates in the trawl fishery (Payne and Punt, 1995, 21).

[36] Pers. comm. (Leslie, Sea Fisheries Research Institute). See Squires (1987b) for a discussion of the numerous difficulties involved in standardising fishing effort.

In line with these arguments, it is suggested that, in the case of pelagic stocks, whilst catch per unit effort does provide a first guide to the state of stocks, a superior measure of the level of exploitation of these stocks is given by the size of the biomass. Reasonably accurate and comprehensive biomass data for all major species are available from 1984 onwards (see Figure 2.7). Earlier data, however, provide anecdotal evidence of a substantial decline in biomass in the pelagic fishery after 1950. For example, in the pilchard fishery, biomass fell by 74.8 percent between 1951 and 1965 (see Figure 2.6).

Figure 2.6: Trends in Biomass of South African Pilchard (1950-65)



Source: Lochner (1980, 17).

In short, falling catch per unit effort figures in the demersal and pelagic fisheries, as well as declining biomass in the pelagic fishery, suggested that between the mid-1950s and the mid-1970s, the bulk of South Africa's commercial fish stocks had been driven into a state of over-exploitation. Over the same period, however, South Africa's fisheries managers had gradually shifted away from *laissez faire* management by introducing a number of direct controls in an attempt to protect South Africa's commercial fish stocks from the effects of over-fishing. This shift away from *laissez faire* management toward the use of direct regulation, which was initiated by interventions in the pelagic fishery, was complete by the end of the 1970s as a dramatic collapse in the hake fishery, South Africa's most important commercial fishery (see Chapter 4), forced South Africa's fisheries managers to adopted direct controls in the only remaining unregulated commercially important fishery.[37] Thus, by the end of the 1970s, all of South Africa's major commercial fish stocks were controlled, with regulations for harvesting primarily taking the form of licences, quotas, closed seasons and minimum mesh sizes. Licences for processing were also introduced in an effort to restrict fishing activity.[38] Regulations introduced by South Africa's fisheries managers were bolstered by the declaration of a 200-nautical-mile exclusive economic zone (EEZ) in November 1977.[39] The EEZ denies foreign fleets the right of access to all stocks located within 200 nautical-miles of the country's coastline.

The regulations listed above had two primary aims. First, they sought to ensure that all major fish stocks were harvested at a biologically sustainable rate and, where relevant, that over-fished stocks were allowed to rebuild. Second, they aimed to provide protection for the domestic fishing industry by excluding foreign fishing fleets. The success of these regulations, however, has been mixed. On the positive side, efforts to reduce fishing by foreign fleets were met with almost immediate success. For example, in the case of the hake fishery, the total catch by foreign trawlers in South African waters stood at over 60 percent of total catch in 1972, but had become insignificant by the end of the decade (Stuttaford, 1995, 11).

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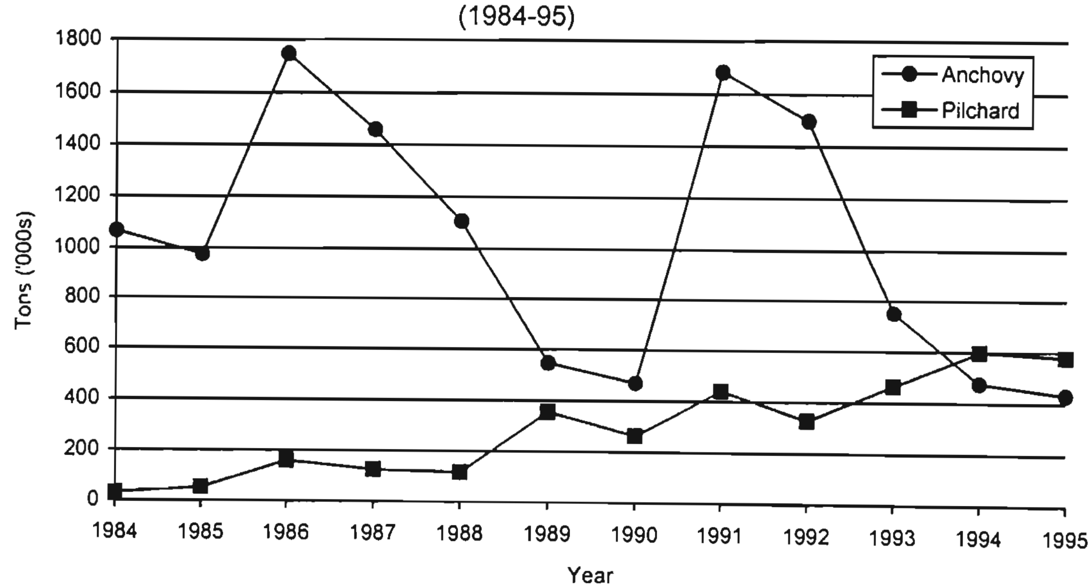
37 Although the inshore fishing grounds of False Bay – the scene of the earliest activities of the trawling industry – had been closed since 1928 (Stoops, 1953, 247); and regulations prescribing a minimum mesh size on the west coast were put in place as early as 1938 (Scott, 1950, 546).

38 South Africa's response to the problem of stock depletion was not atypical. Most countries were slow to respond to the problem of over-fishing: the first international control of a fishery (the Pacific halibut fishery) only took place in the 1930s, and most countries failed to introduce any form of regulation until the 1950s or 1960s (Cushing, 1977, 38-46).

39 A comprehensive review of the international law of the sea is provided in Field and Glazewski (1992).

The evidence with regard to the rebuilding and protecting of fish stocks is more mixed. Although reasonably accurate and comprehensive biomass figures for South Africa's major pelagic stocks are only available from 1984 onwards (Butterworth *et al*, 1993: Pelagic Working Group, 1996, 1), the available evidence suggests that South Africa's fisheries managers have, at best, been only partially successful in preventing or reversing the depletion of pelagic stocks. In the case of anchovy, the 1995 stock level was the lowest biomass estimate since the start of the time series in 1984 (see Figure 2.7) (Department of Environmental Affairs and Tourism, 1997, 5). Furthermore, more recent data (*Business Day*, 1997a) show that the anchovy biomass has continued to fall, resulting in calls for a complete ban on the direct catching of this fish species.[40] In contrast, the pilchard biomass has increased steadily from 32 000 tons in 1984 to 580 000 tons in 1995 – an eighteen-fold increase in biomass in just over ten years (see Figure 2.7). As further evidence of this recovery, the Department of Environmental Affairs and Tourism (1997, 5) recently noted that by-catches of juvenile pilchard have become so abundant in anchovy catches that the reduction fishery is unable to operate efficiently.

Figure 2.7: Trends in Biomass in the Purse Seine Fisheries (1984-95)



Source: Adapted from Pelagic Working Group (1996, 3).

[40] At the time of writing, the initial anchovy catch for 1997 had been set equal to zero (*Business News*, 1997a).

In the demersal fishery, catch per unit effort has increased from the low levels recorded in the 1970s. In the case of hake, catch per unit effort in the west coast fishery increased from around 4.5 tons per day in the mid-1970s to about 8 tons per day in 1992; and in the south coast fishery, catch per unit effort approximately doubled from 0.4 tons per hour to over 0.7 tons per hour over the same period (see Figure 2.4). However, these figures must be interpreted with caution, as the recovery of most stocks has generally been weak, partial and erratic. More specifically, although the trend in catch per unit effort has been upwards, the recovery has not been continuous, with catch rates dropping in a number of years. Furthermore, catch per unit effort still remains well below the highs recorded in the 1950s and 1960s. Moreover, there are indications that over-exploitation of other demersal stocks has continued. For example, in the kingklip trawl fishery, catch per unit effort fell steadily from over 20.0 kilograms per hour in 1983 to 7.5 kilograms per hour in 1989 (Butterworth *et al*, 1992, 993). A substantial decline in the catch per unit effort in the longline demersal fishery for kingklip was also recorded over a similar period: Japp (1993, 134) shows that catch per unit effort fell by two-thirds in both the south and west coast kingklip fisheries over the period 1984-90; and in 1991, the kingklip quota was abolished completely as a result of over-fishing by longliners (Kantey, 1992, 35).

There is also evidence of over-fishing in linefisheries. Penney (1993), for example, provides evidence of a substantial decline in sixteen major linefish stocks over the period 1910 to 1990. Furthermore, evidence of over-fishing in the KwaZulu-Natal offshore linefishery has been provided by Garratt (1993, 14):

> After the [S]econd World War the skiboat was introduced into Natal. The design and capabilities of this boat resulted in the rapid expansion of a sport and commercial fleet along the coast. The introduction of this craft, together with the development of the echosounder and more sophisticated navigation equipment, appear to have had a most profound impact on local stocks. Historical catch records show that there has been a drop in catch per unit effort of more than 90 percent since the 1950s and that there have been dramatic changes in species composition of the catches over the same period. Both are signs of overexploitation.

Thus, Pilford and Pampallis (1993, 118) commented that: 'there have been major changes in the [KwaZulu-Natal commercial] linefishery over the years with several species showing dramatic declines in catches'.[41] Evidence of severe stock depletion in the Cape commercial and recreational linefisheries is also available. Taylor (1993) shows that recreational angling along the Cape coast is (at least partly) responsible for the deterioration of fish stocks, while Bennett (1993) provides evidence of over-exploitation of stocks in both the recreational and commercial linefisheries along the Cape coast, where catch per unit effort for at least one major species is noted to have fallen by 90 percent between 1976 and 1991.

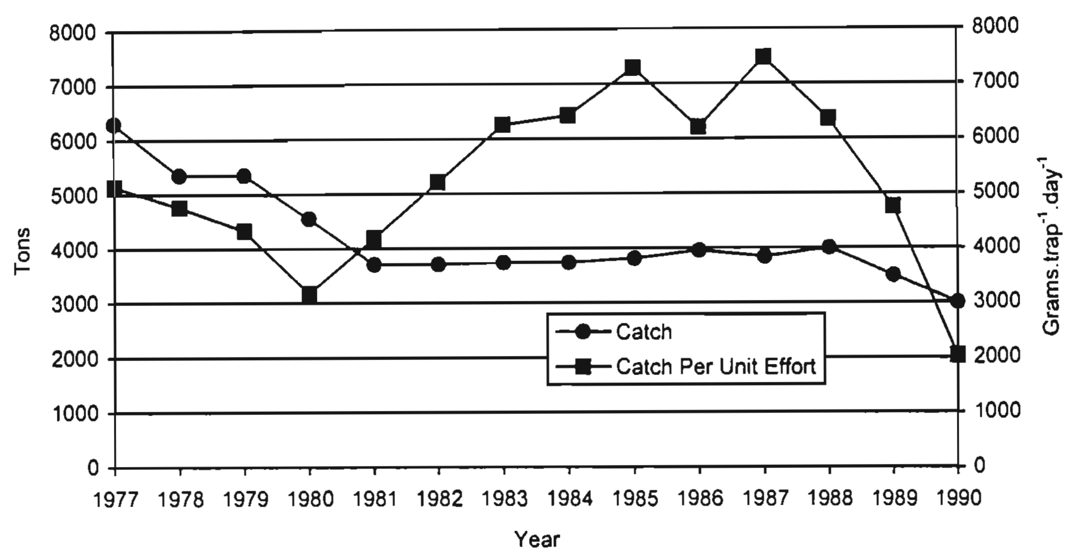
Finally, catch rates and total catch have fallen dramatically in inshore fisheries. For example, catch and catch per unit effort have fallen steadily since the inception of the commercially important rock lobster fishery, indicating that that resource has been over-fished (Kantey, 1992, 35). This is particularly true of the west coast rock lobster (*Jasus lalandii*) fishery, which typically dominates total lobster catches (see Table 2.2). In that fishery, catch and catch per unit effort have fallen dramatically since the mid-1970s. As Figure 2.8 reveals, catch in the west coast rock lobster fishery fell by over 50 percent between 1977 and 1990.[42] Furthermore, although catch per unit effort initially increased (albeit intermittently) from 5 140 grams per trap per day in 1977 to 7 470 grams in 1987, catch per unit effort fell dramatically after 1987, collapsing to just 2 030 grams per trap per day in 1990, a decline of 72.8 percent in three seasons. There is also evidence to suggest that over-fishing has taken place in the smaller deep-sea south coast rock lobster (*Palinurus gilchristi*) fishery over the course of the 1980s and 1990s. For example, the total allowable catch in the south coast rock lobster fishery fell

[41] See also Gowans (1997a & 1997d).

[42] Total catch in the west coast rock lobster fishery peaked at about 12 500 tons in the early 1930s, and remained relatively high until the mid-1960s. Thereafter, the west coast catch collapsed in two stages: once over the late 1970s, and again over the late 1980s. This trend has continued into the 1990s, with the total allowable catch in the west coast rock lobster fishery falling to 1 700 tons, one-eighth of its historic peak, in the 1996 fishing season (Department of Environmental Affairs and Tourism, 1997, 5).

from 475 tons per annum in the late 1980s to 412 tons in the 1996 fishing season (Department of Environmental Affairs and Tourism, 1997, 5). Over-fishing has also occurred in the lucrative abalone fishery. Evidence of this is provided by the decline in the total allowable catch from a peak of 2 800 tons in the 1965 fishing season to 560 tons in the 1996 season. In fact, exploitation of the resource has been so severe that there are fears that the species is on the verge of becoming 'commercially extinct' (Department of Environmental Affairs and Tourism, 1997, 6; MacGregor, 1997).

Figure 2.8: Trends in Catch and Catch Per Unit Effort in the West Coast Rock Lobster Fisheries (1977-90)



Source: Fielding (1996).

In sum, restrictions across both the commercial and recreational fisheries have provided for the recovery of some stocks. However, where recoveries have taken place, they have been weak, partial and erratic. Of substantially greater concern, however, is the fact that many stocks have become over-exploited in spite of regulations.[43] In turn, the ineffectiveness of regulations threatens the future of South Africa's commercially important fishing industry.

[43] This is not out of line with global experiences. Despite a plethora of regulatory mechanisms, 89.6 percent and 60.3 percent of fish stocks in major fishing areas in the Atlantic and Pacific oceans respectively are in a state of full exploitation, over-exploitation or depletion (Loayza, 1992, 6).

There are at least two main explanations for the failure of regulatory controls to prevent or reverse over-exploitation. First, regulations have resulted in high degrees of concentration in the harvesting and processing sectors which, in turn, has promoted illegal activity, such as poaching. The problem has been compounded by inadequate policing, with the result that many stocks have gone under- or unprotected. Second, and of greater concern, is the fact that South Africa's fisheries managers have relied virtually exclusively on scientific criteria as the basis for fisheries management and regulation, and have largely ignored the clear shift in thinking (and increasingly in practice) that has taken place in most other countries away from exclusively biological management of fisheries. These two points are discussed in Sections 2.4.2 and 2.4.3 below.

2.4.2 Regulatory Failures: Concentration, Poaching and Policing

High levels of concentration of ownership in the processing and harvesting sectors have promoted poaching and other illegal activity, such as bribing of officials and intimidation, by fishers, of officials and other fishers, which has eroded the effectiveness of regulations (Dudley and Cliff, 1993; Garratt, 1993; Kyle, 1993; Geldenhuys, 1994; Le May, 1994a & 1994c; Streak, 1994; Yeld, 1994; Leisegang, 1995; Ensor, 1996; Gowans, 1996; Marais, 1996; Schronen, 1996; Shepherd-Smith, 1996). The situation has been compounded by inadequate policing of fisheries and inadequate (or absent) protection for some species. Two factors are chiefly responsible for the high levels of concentration of ownership that have emerged in the fishing industry. First, in the period before 1940, most fishing boats were owned by private fishers and factories were small privately owned companies which were virtually family businesses (Stoops, 1953, 245; Gertenbach, 1973, 2082) However, lack of control over the ability of processors to obtain catch encouraged factories to acquire ownership of vessels and their licences, which led to increasingly higher levels of vertical integration in the fishing sector (Gertenbach, 1973, 2080; Frankel, 1988). Moreover, many of the smaller concerns, whose management, administration and financial status proved inadequate in the face of rapid expansion in the fishing industry, were absorbed by larger units; and mergers and acquisitions among various companies and their subsidiaries meant that by as early as the 1950s, the ownership of fishing and processing licences became lodged in the hands of a few large companies (Stoops, 1953, 245). Second, concentration of ownership was accentuated by apartheid structures which reserved the right of access to resources for a small minority, such that on many occasions, and essentially on the basis of colour, traditional fishing communities were either forced out of fishing altogether, or into the employ of (white-owned) fishing companies (Kantey, 1992, 36; Food and Allied Workers Union, 1997).

The result of these two forces is that large white-owned companies have come to account for the bulk of South Africa's commercial catch and virtually all post-harvest activities. To elaborate: in the 1994 fishing season, 93 percent of the total allowable catch was allocated to large companies or their subsidiaries, comprising Irvin and Johnson Limited (controlled by Anglovaal Industries Limited), Sea Harvest Corporation Limited (CG Smith Limited), Atlantic Fishing Limited (Premier Group Limited), Marine Products Limited (Foodcorp Limited), Oceana Fishing Group Limited (CG Smith Limited via Tiger Oats Limited), Lusitania Sea Products Limited, Viking Fishing Limited and Suiderland Limited. The remaining 7 percent of the 1994 total allowable catch was allocated to small-scale fishers. Furthermore, only 0.7 percent of the total allowable catch was allocated to black fishers, although these fishers, as a group, owned 7 percent of the registered fishing boats and were issued with 6 percent of the 4 000 commercial fishing licences (Le May, 1995a). Similar, but more recent, evidence of the concentration of quotas in key South African commercial fisheries is provided in Table 2.3.

Table 2.3: Quota Distribution in Key South African Commercial Fisheries (1996)

	Total allowable catch (tons)	Number of quota holders	Catch held by largest three (%)	Catch held by largest ten (%)	Catch held by largest twenty (%)
Hake	148 300	49	72	82	87
West Coast Rock Lobster	1 500	104	23	51	73
South Coast Rock Lobster	427	6	82	100	-
Abalone	615	16	75	95	100
Pilchard	105 000	59	30	55	63
Anchovy	70 000	18	36	79	100
Sole	872	11	71	100	-

Source: Department of Environmental Affairs and Tourism (1997, 14).

It is worth noting, however, that, more recently, a number of the larger commercial operations – including Oceana Fishing Limited, Premier Group Limited, Sea Harvest Corporation Limited and Irvin and Johnson Limited – have made attempts to break with the historic domination of the fishing industry by white-owned and controlled companies. Oceana Fishing Limited has attracted the black-controlled Real Africa Investments Limited as a major shareholder, whilst Premier Group Limited has set up an employee share scheme through which employees have acquired a 20 percent stake in the company. Oceana Fishing Limited also plans to introduce an equity participation scheme for all employees and stakeholders (including fishers who work for the group for only a few months of each year) (Hirshon, 1995). Sea Harvest Corporation Limited has set up a R45 million share-purchase scheme that will result in 7.5-8.0 percent of the company being employee-owned (Sharpe, 1996);[44] and Irvin and Johnson Limited has implemented a share incentive scheme which is likely to result in about 9 percent of its share

[44] Similar moves have been undertaken by the major commercial companies operating in Namibian fisheries (Namibian Sea Products Limited, 1995).

capital, worth R95 million at current market value, being held by employees by 2006 (Zaina, 1996).[45]

Notwithstanding the efforts of some of the larger commercial operations to dilute ownership, control over resources effectively remains concentrated in the hands of large, white-owned companies. In this regard, Walker (1995b, 2) noted: '[the Quota Board] appears incapable of equitably dividing this country's fish resources'. Furthermore, the Quota Board's recent efforts to improve equity via the allocation of 'paper quotas' to, for example, fishing communities which have historically been denied access to fisheries, have been surrounded by allegations of political patronage and shrouded by controversy (*Business Day*, 1996; Cohen, 1996). As Hogg (1996) put it:

> The crux of the problem is the issuing of so-called 'paper quotas' to people whose only connection to the fishing industry is a knowledge of how it can make money for them. Holders of these pieces of paper (which are turned into money by selling them to the highest bidder) range from some fairly high profile businessmen to community clergymen, headmasters and even a well known professional soccer team. To put the matter more frankly, the Quota Board has placed political patronage above the desperate plight of fishing communities whose members are prepared to get salt water on their hands to earn an income.

In short, limited access has become an euphemism for privileged access, and this has given rise to high levels of concentration in the South African fishing industry. In turn, it is argued below that concentration of ownership and control in the industry has eroded the effectiveness of regulations on two fronts: (i) by providing fishers with the incentive to poach, and (ii) through over-capitalisation, encouraging further illegal activity, such as bribing of officials and intimidation, by fishers, of officials and other fishers.

With regard to the first of these, concentration of ownership, which has translated into lack of access, has encouraged poaching by coastal communities and small-scale commercial operations. The available evidence suggests that poaching is widespread, particularly in inshore fisheries (such as abalone and rock lobster) where barriers to entry are low (Garratt, 1993; Geldenhuys, 1994; Kyle, 1993; Le May, 1994a & 1994c; Yeld, 1994; Ensor,

[45] The figure of R95 million is based on a market capitalisation of R1 050 million (June 1997).

1996; Marais, 1996; Schronen, 1996; Shepherd-Smith, 1996). In turn, poaching has loosened the control of fisheries managers over resources, which has inevitably resulted in over-exploitation of those resources. For example, National Ocean Watch estimated that at least 1 000 tons of abalone was poached between Cape Hangklip and Coffee Bay – a 1 200-kilometre stretch of the South African coastline – in the 1994 season. This is almost double the total commercial quota of 605 tons (Le May, 1994a). Even more dramatic is a claim by the Hermanus Gemeenskaptrust – a group of small-scale fishers located on the south coast – that it had poached more rock lobster during the 1995 fishing season than the quota allowed for the entire industry (Walker, 1996, 2). Although the validity of the claim is questionable (the fishers' legal catch is very small, amounting to about 1 percent of the south coast rock lobster fishery), the nature of the claim is, none the less, indicative of substantial poaching, which is further evidenced by the almost complete collapse of the resource in some areas (Goosen, 1996).

The problems associated with poaching have been compounded by lack of policing. Specifically, severe financial constraints have precipitated a situation where there is insufficient policing and enforcement of regulations (Walker, 1995a; Marais, 1996; *Business Day*, 1997b). The *Report of the Committee of Inquiry into a National Maritime Policy* (Floor, 1993, 50-4) provided a very broad overview of the extremely limited capacity of the country's policing facilities. More specific detail is available elsewhere; for example, one source (*Econometrix*, 1997) noted:

> Whereas internationally the ratio of defence spending on capital equipment to personnel is around 55:45, in South Africa it is approaching 20:80. Thus, for example, the country's navy is operating with almost obsolete ships in a feeble attempt to protect the country's waters against foreign fishing trawlers which are depleting our sea's fishing stocks.

On the same subject, Coetzee (1993, 178) noted that there are nineteen officers responsible for monitoring the KwaZulu-Natal coastline and, on average (although not in practice), each officer is responsible for monitoring twenty kilometres of coastline, nine commercial ski-boats, 203 light-tackle boats, 2 339 bait licence holders, 7 782 rock and surf anglers and checking

141 tons of fish caught annually.[46] The net effect of such extreme limitations on policing capacity is that authorities have been unable to prevent the spread of illegal activity throughout the industry; and the problem is exacerbated by the fact that illegal activity has often taken place with the knowledge of the authorities (Walker, 1995b, 2-3). In addition to poaching by domestic fishers, there is evidence of other unlawful activities, including the use of illegal equipment, such as gillnets (Leisegang, 1995; Gowans, 1996); the existence of large-scale operations involved in the export of poached products (Streak, 1994); and poaching by foreign vessels in South African waters (Kantey, 1992, 35; Rice, 1997). All these activities have served to undermine regulations.

The second explanation for the erosion of the effectiveness of regulations lies in over-capitalisation. Over-capitalisation refers to the situation where the catching capacity of fishing fleets exceeds the available catch. The problem of over-capitalisation often arises where regulations aimed at reducing catches – but not fishing capacity – are introduced.[47] The effects of over-capitalisation are obvious: excessive pressure on resources results in over-harvesting which, in turn, leads to ever-diminishing returns to fishing effort and capital. Consequently, the system degenerates into a self-perpetuating downward spiral toward severe over-exploitation and, in the extreme, extinction of fish stocks. Although this caricature is perhaps overly dramatic, the conclusion holds: excess capacity encourages over-harvesting.

There is evidence of over-capitalisation in some South African fisheries. For example, in commenting on the pelagic fishery, Gertenbach (1973, 2077) noted: 'the results of the limitation of entry into the fishing phase and processing phases include several significant trends, among which are ... larger-size boats with more powerful engines and equipment'. More concrete

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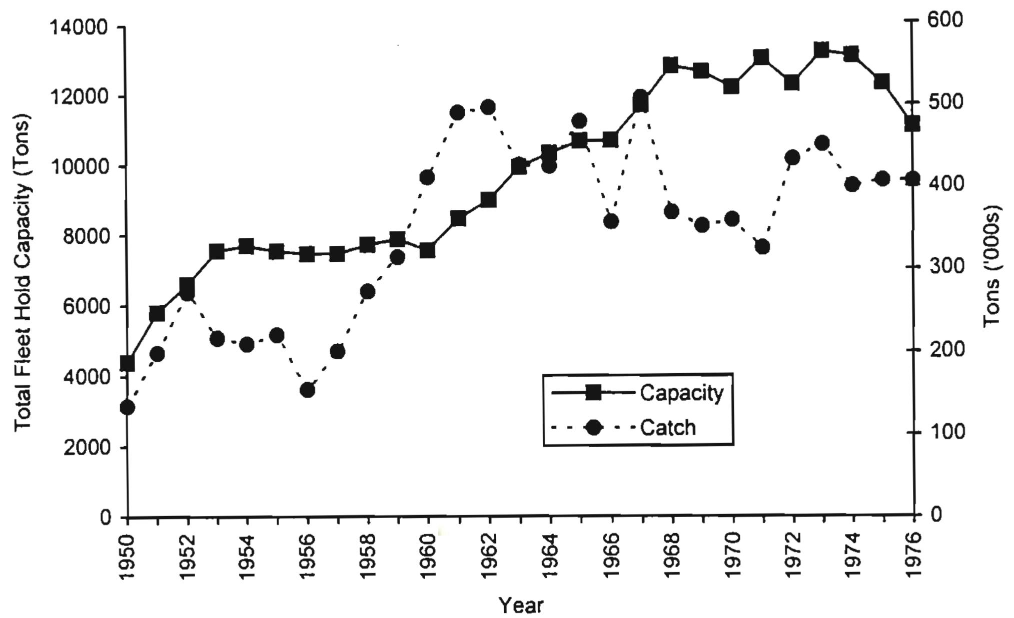
[46] See also Goosen (1996) and Rice (1997).

[47] This argument is returned to in Chapter 7 where it is explored in detail. An extreme example of over-capitalisation is the case of the herring-roe fishery in British Columbia, Canada, where the fishing period is limited to fifteen minutes, and is started by a radio signal (Kantey, 1992, 34). Fraser and Jones (1989, 274) also provide a review of the nature of the over-capitalisation problem in international fisheries.

evidence shows that capacity in the pelagic fishery increased steadily between 1950 and 1976, whilst total catch (which initially grew rapidly but then stagnated in the 1960s) fluctuated around 400 000 tons (see Figure 2.9). In addition, it has been argued by Kantey (1992, 35) that over-capitalisation in the lobster, anchovy and squid fisheries, has led to pressure being placed on the minister responsible to award larger quotas than are recommended by fisheries scientists.

Figure 2.9: Fleet Capacity and Total Catch in the South African Purse Seine Fishery (1964-76)



Source: Crawford (1981, 2-4).

The regulatory controls adopted by fisheries managers have been further weakened by intimidation and abuse by fishers of officials, as well as of newcomers who pose a threat to existing quota allocations (Geldenhuys, 1994; Le May, 1994c, 1995c & 1995d; Yeld, 1994; Pearce, 1995). It has also been suggested that efforts to regulate some fisheries have been undermined by collusion between officials and poachers (Le May, 1994b). Moreover, there is evidence of corruption (Streak, 1994), including the allocation of quotas to family and business associates of quota board members; and the sale, by officials, of forged fishing licences (Marais, 1996). Finally, the regulatory regime has been shown to have gaps in that some fisheries have remained partially and, in some instances completely, unprotected (Dudley and Cliff, 1993; Streak, 1994; Walsh, 1996).

2.5 Implications

The arguments outlined above lead to a number of important conclusions. First, although regulations have managed to reverse over-exploitation of some stocks (most notably in the pilchard and hake fisheries), generally, these efforts have been partial, weak and erratic. There are a number of explanations for this. In many instances regulations have failed as a result of the consequences of the regulations themselves. Problems that can be identified here include: concentration of ownership, which has promoted illegal activity, such as poaching; and over-capitalisation, which has provided an incentive for corruption (for example, the bribing of officials to secure higher quotas or to ignore illegal activity). Moreover, the situation has been compounded by lack of policing capacity which has often meant that regulations have gone unenforced; examples include poaching by domestic and foreign fishers and the use of illegal equipment. This amalgam of factors has undermined the potential and threatens the future of the fishing industry. In this regard, the World Resources Institute (1994, 184) noted:

> [U]nless nations take decisive, timely action to control fish exploitation, the population of many species will drop below levels necessary to assure maximum yields. This development could seriously disrupt ecosystems, put [hundreds of] thousands of fishers out of work, and prove disastrous to the estimated one billion people, most of whom live in developing countries, who depend on fish as their sole source of protein.

In short, it is vital that control mechanisms are introduced timeously, and that they are effective. Generally, this has not been the case in the management of South Africa's fisheries. Accordingly, given the socio-economic importance of the fishing industry, it is vital that a more appropriate set of controls is introduced as a matter of urgency.[48] Here, however, another major concern

[48] It should be noted that, at the time of writing, a new fisheries policy is being formulated. The Department of Environmental Affairs and Tourism's (1997) White Paper, *A Marine Fisheries*

regarding fisheries management emerges. The management of the country's fish stocks has been based primarily on biological principles which, it will be argued, are a necessary, but insufficient, basis for the management of fisheries (and, indeed, renewable resources in general). The argument is explored in detail in Chapter 3.

Policy for South Africa, and the subsequent *Marine Living Resources Bill* (Republic of South Africa, 1997) propose a number of significant changes to the structure, management and regulation of South Africa's marine fisheries. The Department proposes to have the new fisheries policy in place from January 1998. The proposed policy changes are given greater attention in Chapter 7.

CHAPTER 3: BIOECONOMIC MANAGEMENT OF FISHERIES[1]

3.1 INTRODUCTION

As argued in Chapter 2, the rapid growth in South Africa's commercial fishing industry resulted in a large portion of the country's most important commercial fish stocks being driven into a state of biological over-exploitation – evidenced by, *inter alia*, diminishing stock levels and falling catch per unit effort – between the late 1930s and the end of the 1970s. The regulatory controls that were introduced by fisheries managers in response to over-exploitation were often inadequate or inappropriate and, in many instances, exacerbated the problem of over-exploitation. This situation has been complicated by the fact that fish stocks have generally been managed and regulated on the basis of biological principles, despite the existence of a well-established body of theory, and substantial empirical evidence, which shows that biological principles are an insufficient basis for fisheries management.

This chapter explores this evidence in detail, and arrives at two main conclusions. First, unregulated (or, more correctly, open-access) fisheries are likely to be driven into a state of over-exploitation. This result provides the basis for managing fisheries by, for example, restricting access, limiting fishing effort or creating private property rights. Second, and more important, management based on biological principles is sub-optimal: it fails to maximise social welfare. Accordingly, biological management alone is rejected as a basis for fisheries management, and an appropriate alternative – bioeconomic (or biological and economic) management – is provided.

Section 3.2 provides a background to the argument for employing bioeconomic principles as the basis for fisheries management; and Section

[1] This chapter draws heavily on Clark (1976 & 1985), Munro (1982), Munro and Scott (1985) and Randall (1987). For a comprehensive review of the literature of fisheries economics, see Munro and Scott (1985).

3.3 offers concrete support for these arguments by developing a dynamic bioeconomic fisheries model. The implications for the management of fish stocks – regulation based on bioeconomic and not exclusively biological principles – are also spelt out. Section 3.4 goes on to suggest that the model presented here is easily adapted to take into account different biological and economic characteristics of fisheries. The implication is that the model is well-suited to represent a wide range of fisheries, including the most important components of the South African commercial fisheries, and this makes the model extremely attractive from a management point of view. However, the model does suffer from a number of limitations; these limitations are discussed in Section 3.4.

3.2 A Background to Bioeconomic Management

Concern amongst economists with fisheries dates back to Marshall (1890) who addressed the problems of diminishing returns to inputs in fishing;[2] and the work of some biologists over the early part of this century – for example, Baranov (1925) and Thompson and Bell (1934) – spilled over into economics. However, it was only in the early 1950s that the first article to direct its attention explicitly to the theory of fisheries policy was produced. The article proved seminal on two grounds. First, the writer – Gordon (1954) – showed that the reliance by fisheries managers on biological principles as the sole

[2] Curiously, Marshall misinterpreted the application of the law of diminishing returns to the fishing industry. In *Principles of Economics* (1890, book IV, chapter 3, 7) Marshall noted:

> In river-fisheries, the extra return to additional applications of capital and labour show a rapid diminution. As to the sea, opinions differ. Its volume is vast, and fish are very prolific, and some think that a practically unlimited supply can be drawn from the sea by man without appreciably affecting the numbers that remain there, or in other words, that the law of diminishing returns scarcely applies at all to the sea fisheries; while others think that experience shows a falling-off in the productivity of those fisheries that have been vigorously worked, especially by steam trawlers.

Marshall's reference to the law of diminishing returns is out of place in the above context: the law of diminishing returns refers to the decline in productivity of one variable factor of production, *ceteris paribus*. If the fish stock is regarded as an input, as it correctly should be, then Marshall's reference is to a scenario where more than one input is allowed to vary. This is not what is meant by the law of diminishing returns.

basis for policy formulation was an inappropriate management strategy. More specifically, Gordon strongly criticised biological management principles on the ground that such principles focus on the yields (or products) of resource exploitation, but fail to incorporate costs of harvesting as part of the management problem. Here it is worth quoting Gordon (1954, 128) at length:

> The term 'fisheries management' has been much in vogue in recent years, being taken to express a more subtle approach to the fisheries problem than the older term 'depletion' and 'conservation'. Briefly, it focuses attention on the quantity of fish caught, taking as the human objective of commercial fishing the derivation of the largest sustainable catch. This approach is often hailed in the biological literature as the 'new theory' or the 'modern formulation' of the fisheries problem. Its limitations, however, are very serious, and, indeed, the new approach comes very little closer to treating the fisheries problem as one of human utilization of natural resources than did the older, more primitive, theories. Focusing attention on the maximization of the catch neglects entirely the inputs of other factors of production which are used up in fishing and must be accounted for as costs.

To put matters slightly differently, biological management addresses the benefits of resource exploitation but completely ignores the cost side of cost-benefit considerations. This fundamental flaw means that biological principles, such as the maximum sustainable yield concept, are virtually useless for descriptive theories of renewable resource exploitation (Clark, 1975, 2).[3] Second, Gordon – and a number of other early writers[4] – went on to provide alternative resource management models, based on the merging of biological and economic – to form bioeconomic – principles.

Two broad approaches to bioeconomic modelling and resource management have developed within fisheries economics. These are commonly referred to as (i) the 'cohort approach' and (ii) the 'general production' or 'surplus production approach' (Munro and Scott, 1985; Getz and Haight, 1989). The cohort approach, which is primarily due to Beverton and Holt (1957), models on the basis of cohorts (or age- or year-classes). This approach views the population as a 'dynamic pool' with inflows due to recruitment of progeny to

[3] See Section 3.3.1 for a description of the concept of maximum sustainable yield.
[4] Until the late 1960s, the work in fisheries economics either refined or qualified the work of Gordon. See, for example, Scott (1955), Crutchfield (1956), Zellner (1962), Turvey (1964) and Christy and Scott (1965). Further, some writers undertook empirical investigations into particular commercial fisheries, developing the model from the basic theory as presented in Gordon's paper. See especially the work by Crutchfield and Zellner (1963) and Crutchfield and Pontecorvo (1969).

the population as well as growth of individuals within the resource, and decreases in the biomass caused by natural mortality (mainly due to predators) and fishing mortality. Thus, cohort models view the biomass as a dynamic series of cohorts or age-classes. Furthermore, cohort modelling suggests that, by manipulating the level of effort directed at the stock and the age-class which may be caught, it is possible to influence the stock biomass (and so the yield).

However, cohort models suffer from a number of drawbacks which render them practically useless for management purposes. Cohort modelling calls for fishing gear with what biologists refer to as 'knife-edge selectivity' (Wilen 1985, 93; Bjørndal, 1986, 14), that is, fishing gear that is able to precisely select (for example, through choice of appropriate mesh size) fish of a certain age (or size) and above. However, whereas in the case of trees, for instance. individuals in a forest stand can be seen and counted, and tree attributes such as stem diameter, height and volume can be measured, it is very difficult to measure accurately the size of individual cohorts in a fishery. Moreover, even with the aid of the most advanced fishing gear, it is virtually impossible to achieve knife-edge selectivity in harvesting (Getz and Haight, 1989, 143). Almost invariably fishing occurs on a multi-cohort basis; and the economics of multi-cohort fishing becomes very complicated indeed. Even with the aid of powerful mathematical tools, it is extremely difficult to produce satisfactory analytical results, and the success of economists in using Beverton-Holt type models as the basis for bioeconomic models has been extremely limited (Cushing, 1977; Munro and Scott, 1985, 625; Wilen, 1985, 93-112).

Furthermore, efforts made to extend the cohort approach into a dynamic setting have rendered the model virtually useless.[5] Indeed, Morey (1986, 33) argued: 'Dynamic analysis of the Beverton-Holt model is effectively intractable.' Before proceeding, however, two points should be noted. First, whilst the Beverton-Holt approach has been the most widely used to

[5] See Wilen (1985, 93-112) for a complete discussion of the difficulties involved in modelling on the basis of cohorts. The discussion also includes a useful review of the distinction between non-overlapping generation models and overlapping generation models.

represent the growth rate of fish stocks under cohort modelling, other recruitment models have been employed to model fish stocks, such as those studied by Ricker (1954) and Cushing (1981).[6] Thus, whilst the descriptive accuracy of these alternative models might vary through stocks, the essential problems of 'intractability' and 'knife-edge selectivity' remain. Second, while a few economists have had limited success with 'solving' models based on cohort population models (Hannesson, 1975; Getz, 1979; Bjørndal, 1986; Townsend, 1986; Deacon, 1988), this still does not remove the problem that management on the basis of cohorts demands 'knife-edge selectivity' which cannot be met, even by the most advanced harvesting technology. Thus, modelling and managing fisheries on the basis of cohorts must continue to be viewed as impractical.

In contrast, the surplus production or general production approach typically ignores cohorts by assuming that the most important determinant of stock size is the biomass itself. That is to say, rather than depending on many factors, such as the age structure of the population, food availability and natural mortality, the surplus production approach assumes that the rate of change of the biomass is determined by a single variable: the biomass itself (Butterworth and Newman, 1979, 307). Although there are drawbacks in not being able to model on the basis of cohorts – such as a loss of precision in describing population structures – these so-called single variable, or 'lumped variable' models, which served as the foundation for Gordon's work, have been employed in a majority of fisheries economics studies and continue to serve as the centrepiece in contemporary bioeconomic modelling.[7]

One of the main reasons for this success is that these models are easily adapted to represent a range of bioeconomic features of fisheries, and are considerably more amenable to bioeconomic analysis than cohort models

[6] A more novel approach to cohort modelling was employed by Lochner (1980), who attempted to describe the growth of a pelagic species in terms of electrical engineering principles. This effort has been severely criticised by scientists (Botha, 1980; Butterworth, 1980; Crawford, 1980; Field and Greene, 1980; Newman, 1980; Stewart, 1980).

[7] In general production modelling, the natural growth rate of fish stocks has usually been represented by the Schaefer model, although other growth functions, such as those studied by Pella and Tomlinson (1969), Fox (1970) and Shepherd (1982) have been used.

(Munro and Scott, 1985, 625-6; Bjørndal, 1986, 25; Morey, 1986, 31-2; Getz and Haight, 1989, 136). Moreover, cohort models require detailed biological information which is often not available, whereas general production models have low demands for data (Butterworth and Newman, 1979; Sobel, 1982). In addition, and as is argued later (see Chapter 4), general production models are widely applicable to South Africa's major commercial fish stocks.

In line with the above arguments, Section 3.3 develops a dynamic theory of renewable resource management, based on a general production model of biological growth that is readily applicable to the South African fishing industry, and which weights the benefits and costs of resource usage equally. The model is essentially a dynamic version of Gordon's static bioeconomic model, as discussed by Clark (1976 & 1985).

3.3 A Dynamic Bioeconomic Fisheries Model

As already noted, the modern economic theory of fisheries management has its origins in the open-access model developed by Gordon (1954).[8] Gordon's theory of 'common property' not only explained the low income of fishers, but also clarified in economic terms the so-called over-fishing problem. The model explained how economic over-fishing would be expected to occur in any unregulated fishery, while biological over-fishing would occur whenever cost:price ratios were sufficiently low. Gordon's model also suggested possible solutions to the over-fishing problem, and these have formed the basis for a number of 'limited entry' programs which have been introduced in various countries in an effort to overcome the 'tragedy of the commons' (Hardin, 1968).[9] Section 3.3.1 sets out the static Gordon model, and Section

[8] An open-access resource is one in which exploitation is completely uncontrolled: anyone can harvest the resource. Few present day resources satisfy this definition; most fall under some form of regulation.

[9] Hardin's (1968) seminal paper, *The Tragedy of the Commons*, describes the problems that arise when private property rights to a resource do not exist. In such a case, the 'commons' are treated as a free good. The inevitable result is argued to be the over-exploitation and possible destruction of the resource (Hardin, 1968, 1244-5). This is in direct contrast to the

3.3.2 explores the main policy implications of the model. However, experience and more detailed bioeconomic modelling now indicate that, in its original form, the Gordon model – and its proposed remedies – was far too simplistic (Clark, 1985). Accordingly, a more sophisticated model, which overcomes the major criticisms of Gordon's static model, is presented in Section 3.3.3.

3.3.1 The Static Gordon Model: A Review

The biological basis for Gordon's model is provided by the logistic model of population growth. The logistic growth equation, which was first proposed as a population model by Verhulst in 1838, (Clark, 1975), gives population growth as:

$$\frac{dX_t}{dt} = rX_t\left(1 - \frac{X_t}{K}\right). \qquad \text{(Equation 3.1)}$$

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notion, as argued by Adam Smith (1776, Book IV, chapter 2, 9), that the selfish pursuit of one's own interest brings the greatest gain to society:

> [E]very individual ... intends only his own gain, and he is in this, as in many other cases, led by an invisible hand to promote an end which was no part of his intention. Nor is it always worse for society that it was no part of it. By pursuing his own interest he frequently promotes that of society more effectually than when he really intends to promote it.

Two points should be made in this regard. First, *The Tragedy of the Commons* is a misnomer. Hardin's reference to the commons was as a 'free good', that is, property to which there is open (or unrestricted) access (such as *res nulliae*). However, under *res communes*, where there is communal or joint ownership, access may be open or restricted; furthermore, the right of joint owners to harvest the resource may be 'stinted' or limited (Pearse, 1980, 192-5). In other words, common property is not an euphemism for open-access. Indeed, open-access is only one of the many possible conditions that may apply to *res communes*. A more appropriate title for Hardin's paper would have been *The Tragedy of Open-Access*. Gordon (1954) and others (Crutchfield, 1956; Plourde, 1971; Bell, 1972; Brown, 1974; Anderson. 1976; Smith, 1976; Wilson, 1977; Visiglio, 1978; Pearse, 1980; Schworm, 1983; Koenig, 1984; Karpoff, 1987) were similarly at fault for attaching the label 'common property' to a theory of what are 'open-access' resources. None the less, the upshot of Hardin's (and Gordon's) argument holds: open-access results in over-exploitation. Second, and closely related to the first point, a substantial literature has been built up to show that common property management is not a *sine qua non* for over-exploitation (Runge, 1981; Acheson, 1989; Berkes, 1989b; Berkes and Farvar, 1989; Bromley and Cernea, 1989; Goodland *et al*, 1989; Jacobs, 1989; Hviding and Baines, 1994; Tisdell and Roy, 1997). This literature reveals the serious oversight of Hardin, Gordon and others: common ownership may be a viable management option, and has been shown, in places, to provide resource management outcomes that are often superior to private property outcomes (Tisdell, 1972; Mishra, 1982; Vink and Kassier, 1987; Acheson, 1989; Bromley and Cernea, 1989; Goodland *et al*, 1989; O'Boyle *et al*, 1991).

This model assumes population growth (dX_t / dt) is a function of the intrinsic growth rate of the fish stock (r), the existing level of the biomass (X_t) and the carrying capacity – or Malthusian limit – of the environment (K).

The model has a number of important features. First, the model possesses two equilibrium solutions, namely $X_t = 0$ and $X_t = K$. In other words, in an unexploited state, there are two levels of biomass at which the population does not grow: a zero population and the carrying capacity population. Moreover:

$$0 < X_t < K \text{ implies } \frac{dX_t}{dt} > 0,$$

whereas

$$X_t > K \text{ implies } \frac{dX_t}{dt} < 0.$$

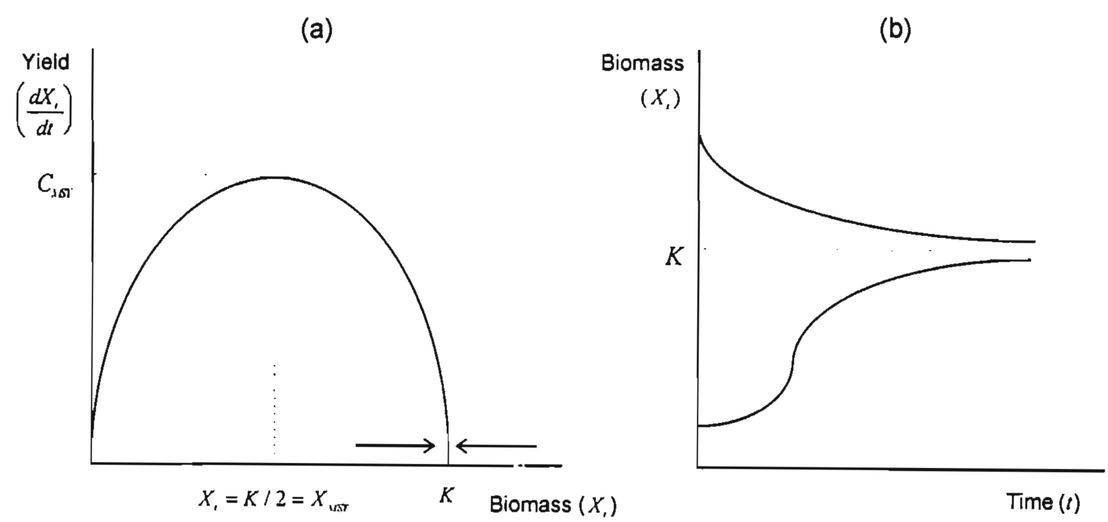
That is to say, the resource will exhibit positive growth as long as the existing biomass lies between zero and the carrying capacity population; and the biomass will exhibit negative growth if the existing biomass exceeds the carrying capacity of the environment. Thus it follows that K, the carrying capacity biomass, is a stable equilibrium. To be more precise, K is globally asymptotically stable for all positive X_t, in the sense that:

$$\lim_{t \to \infty} X_t = K, \text{ provided } X_t > 0.$$

In other words, in the absence of harvesting, and provided the biomass is non-extinct, the population will grow to the carrying capacity population; and the carrying capacity population constitutes a stable equilibrium.

A second important feature of the model is that the population level at which the productivity of the resource is maximised is not the natural equilibrium level K; indeed, there is no sustainable yield at K. Rather, the maximum sustainable yield or catch ($C_t = C_{MSY}$) occurs at the population level $K/2$, that is, at the population level equivalent to half the carrying capacity biomass. When a population has been reduced to a level below $K/2$ by harvesting, the resource is considered to be in a state of biological over-exploitation. These arguments are illustrated in Figure 3.1 below. Figure 3.1(a) shows the growth function (dX_t / dt) and the maximum sustainable yield population $(X_t = K/2 = X_{MSY})$; the arrows indicate the direction of change of X_t with increasing time (t). Figure 3.1(b) illustrates the solution curves to X_t approaching K from above and below. The lower curve with its characteristic ogive shape is usually referred to as the logistic growth curve.[10]

Figure 3.1: The Logistic Model of Population Growth and Typical Solution Curves.



Source: Wilen (1985, 71).

To model the effect of fishing on population dynamics, Gordon drew from Schaefer (1954 & 1957).[11] Schaefer altered the logistic equation to reflect harvesting such that:

[10] A more detailed discussion of this model and the ecological significance of the parameters is given in May (1981) and Wilen (1985).

[11] For this reason, the model is often referred to as the Gordon-Schaefer model. This terminology is not employed here because Gordon's model is easily altered to incorporate other specifications of the population growth function, such as those discussed by Pella and Tomlinson (1969), Fox (1970) and Shepherd (1982). Nevertheless, and importantly, for the purposes of this study, all references to the Gordon model assume a Schaefer growth function – as per Gordon (1954) and Clark (1976 & 1985) – unless otherwise stated.

$$\frac{dX_t}{dt} = rX_t(1 - \frac{X_t}{K}) - C_t \qquad \text{(Equation 3.2)}$$

where C_t denotes the catch rate. That is, under conditions of harvesting, population growth (dX_t / dt) becomes a function of the intrinsic growth rate of the fish stock (r), the existing level of the biomass (X_t), the carrying capacity of the environment (K) as well as the catch rate (C_t).

Schaefer expressed catch in terms of effort by the relation:

$$C_t = qE_tX_t \qquad \text{(Equation 3.3)}$$

where E_t denotes fishing effort and q is the constant called the catchability coefficient.

Related to this, fishing mortality (F_t) is defined as the relative mortality due to fishing:

$$F_t = \frac{C_t}{X_t}. \qquad \text{(Equation 3.4)}$$

Thus:

$$C_t = F_tX_t \qquad \text{(Equation 3.5)}$$

where, by Equation 3.3:

$$F_t = qE_t. \qquad \text{(Equation 3.6)}$$

Equation 3.5 is merely a definition (of fishing mortality), whereas Equation 3.6 constitutes an important hypothesis of the Schaefer model. Specifically, this hypothesis states that fishing mortality is directly proportional to fishing effort.[12] Then, by rewriting Equations 3.5 and 3.6 in the form:

[12] This hypothesis is thought to be seriously misleading for pelagic species (Punt, 1994, 944), because pelagic fish usually school, and a single 'lucky' catch can substantially alter the ratio of catch to effort. However, the hypothesis is widely accepted as valid in the instance of demersal species, because demersal fish stocks are generally distributed relatively evenly across fishing grounds, implying that increases in fishing effort bring proportionate increases

$$\frac{C_t}{E_t} = qX_t \qquad \text{(Equation 3.7)}$$

the Schaefer hypothesis asserts that catch per unit effort is a direct index of stock abundance (X_t).

From this simple model, it is possible to establish the equilibrium solution to the Gordon model. By setting $dX_t / dt = 0$ in Equation 3.2 and copying Equation 3.3:

$$C_t = rX_t(1 - \frac{X_t}{K}) \text{ and } C_t = qE_tX_t.$$

These equations imply that:

$$X_t = K\left(1 - \frac{qE_t}{r}\right) \qquad \text{(Equation 3.8)}$$

and, therefore:

$$C_t = qKE_t\left(1 - \frac{qE_t}{r}\right). \qquad \text{(Equation 3.9)}$$

Equation 3.9 thus represents the relationship between catch and effort at equilibrium. The graph of Equation 3.9 is the well-known yield-effort curve, as per Schaefer (1957), and is set out in Figure 3.2. This curve depicts sustainable yield or catch as a function of fishing effort E_t. The yield-effort curve reaches a maximum, referred to as the maximum sustainable yield, and then declines as effort is further increased. At sufficiently high sustained effort levels, yield falls to zero. Biological over-fishing is said to occur whenever the level of effort is in excess of the level required to generate the

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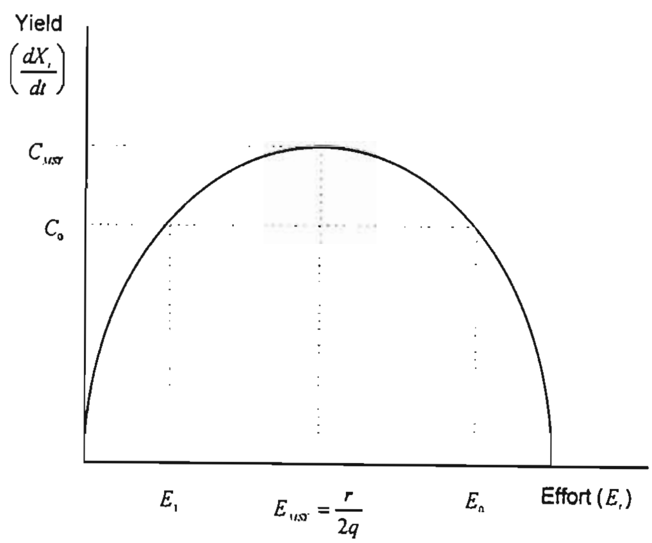
in catch. Indeed, Gordon (1954, 129) explicitly acknowledged that his theory was developed for a 'typical demersal fish'.

maximum sustainable yield. Point E_0 in Figure 3.2 represents a position of biological over-fishing where, as a result of excessive effort, the catch (C_0) is less than the maximum sustainable yield (C_{MSY}). Importantly, at points such as E_1 in Figure 3.2, although the catch (C_0) is less than the maximum sustainable yield, increases in effort will yield higher catches, and so this situation is not regarded as biological over-fishing. Indeed, biologists would generally regard a level of effort such as E_1 as a situation of 'under-fishing'. Maximum sustainable yield is given by:

$C_{MSY} = \frac{rK}{4}$, and is associated with the effort level $E_{MSY} = \frac{r}{2q}$.

Figure 3.2: The Sustainable Yield-Effort Curve



Source: Randall (1997, 321).

The Schaefer yield-effort curve also provides a crude method for estimating maximum sustainable yield from catch-effort data. Specifically, Equation 3.9 implies a linear relation:

$$\frac{C_t}{E_t} = a - bE_t$$

whose coefficients can be estimated by simple linear regression. Maximum sustainable yield is then given by:

$$C_{MSY} = \frac{a^2}{4b}.$$

The bioeconomic model is completed by adding Gordon's economic component to the Schaefer model. Given that each point on the Schaefer curve (Figure 3.2) corresponds to the sustainable catch (C_t), if a constant price for fish (p) is received by the fishers, the total sustainable revenue function can be written as:

$$Revenue = pC_t.$$

Furthermore, the costs of applying one unit of fishing effort (c) are assumed to be constant, such that the total cost of fishing is given by:

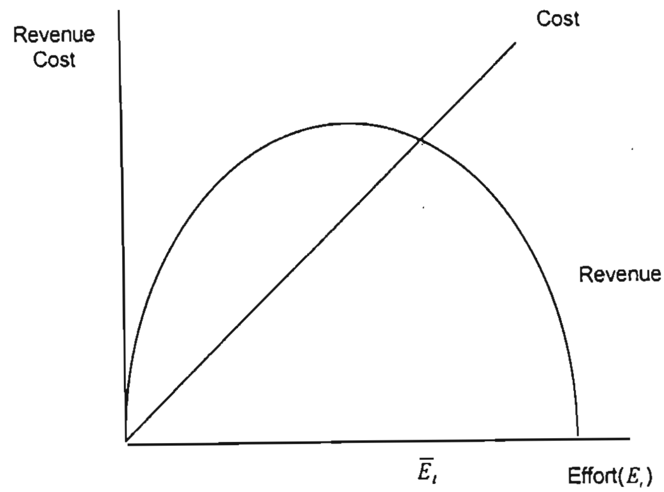
$$Cost = cE_t.$$

These cost and revenue curves are shown in Figure 3.3.

A central assumption of the above analysis is that the demand for fish and the supply of fish are both perfectly elastic. Moreover, it is assumed implicitly that the price of landed fish accurately represents the marginal social benefit of harvested fish, and that the unit cost of fishing effort is a true measure of the marginal social cost of such effort (Munro and Scott, 1985, 628). Accordingly, the difference between total sustainable revenue and total cost, at each given level of effort (E_t), is the sustainable economic rent (Π_t), and is given by:

$$\Pi_t = pC_t - cE_t = (pqX_t - c)E_t. \qquad \text{(Equation 3.10)}$$

Figure 3.3: The Static Gordon Model: Effort-Revenue (Cost) Relationship



Source: Clark (1985, 3).

Gordon's principal result for the open-access fishery can now be given. In the open-access fishery, effort tends to reach an equilibrium, commonly referred to as bionomic equilibrium, at the effort level $E_t = \overline{E}_t$ at which total revenue equals total cost. In other words, the economic rent is completely dissipated. The main justification for rent dissipation is that the fishery would earn economic profits for any level of effort for which $E_t < \overline{E}_t$; and, under open-access conditions, this would attract additional fishers, effort would increase and, ultimately, all rents would be dissipated.[13]

As noted by Gordon (1953, 131): 'The fisherman has no legal title to a section of ocean bottom. Each fisherman is more or less free to fish wherever he pleases. The result is a pattern of competition among fishermen which culminates in the dissipation of ... rent.' In similar vein, Turvey (1964, 71) argued: 'In an unregulated fishery ... resource allocation is non-optimal, [and] free entry means that ... no rent of the fish stock is achieved'; and Gould (1972a, 383) noted: 'It is widely-accepted doctrine that free-access resources

[13] By the same token, no level of effort $E_t > \overline{E}_t$ can be maintained indefinitely, for this would produce a situation where the total cost of fishing would exceed total revenue. At least some of the fishers would suffer economic losses and would withdraw from the fishery, thereby reducing the level of effort back to $\overline{E}_t$.

are over-exploited – that is, that they attract more [harvesting] resources than is required for allocative efficiency.' However, the point has, perhaps, been most forcibly made by Hardin (1968, 1244): 'Freedom in a commons brings ruin to all.' This is the so-called 'Class I common property problem' of too many resources chasing too few fish (Munro and Scott, 1985, 631; Johnston, 1992, 6).[14]

It should be noted that Gordon's result has been challenged by some writers. For example, Smith (1968 & 1969) argued that competitive pressures influence the cost of fishing via stock externalities (reduced fish stocks increase harvest costs), crowding externalities (over-crowding of fisheries by fishers and vessels increases operating costs) and, possibly, mesh externalities (use of certain gear, such as small mesh size, alters the growth rate of fish stocks); and the higher costs of fishing reduce the level of effort (and catch) directed at the fishery.[15] Thus, in contrast to the received theory of, *inter alia*, Gordon (1954), Scott (1955), Crutchfield (1956), Turvey (1957 & 1964) and Crutchfield and Zellner (1963), Smith (1969, 191) concluded:

> ... it is clear that the competitive [open-access] equilibrium may require a larger or smaller amount of capital than sole ownership [maximum economic yield]; also, the harvest, which is the same as the population yield ... may be larger or smaller [at the competitive outcome than at the sole ownership outcome].

However, while the validity of Smith's result has been questioned (Fullenbaum *et al*, 1972; Hartwick, 1982) and (partly) defended (Smith, 1972) in the literature,[16] it is the lack of empirical evidence to show that competitive-access does not result in over-exploitation of fish stocks that provides the greatest defence of the result of the 'traditional' fisheries literature, that is competition results in rent depletion in open-access fisheries.

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[14] In line with earlier comments, the correct label for the over-fishing problem is 'Class I open-access problem', which has little or nothing to do with common-property conditions. This more correct label is used elsewhere in this study.

[15] See also Degnbol (1992).

[16] In similar vein, Rosenman (1986) employed the theory of the firm to reveal a set of conditions under which the unregulated equilibrium of a competitive industry will be the dynamic maximum economic yield outcome, but suggested that it is unreasonable to expect the necessary set of conditions to exist, thereby rendering regulation of the competitive industry necessary (Rosenman, 1986, 357).

The upshot of this result is that applying a level of effort $E_t = \overline{E}_t$ (Figure 3.3) is economically inefficient: the fishery resource, which is capable of producing positive economic rent, is producing zero economic rent because an excessive level of effort is being utilised. Neither the fishers nor society at large are enjoying the benefits that would accrue if the fishery were under bioeconomic management. This situation may be defined as economic over-fishing. In addition, the fishery may also suffer from biological over-fishing, a situation in which the biomass yield is less than the maximum sustainable yield (as in Figure 3.2, where $C_0 < C_{MSY}$).

Under the Gordon yield-effort model, the zero-rent or bionomic equilibrium is given by:

$$\frac{dX_t}{dt} = rX_t(1-\frac{X_t}{K}) - C_t = 0, \text{ and}$$

$$\Pi_t = Revenue - Cost = pC_t - cE_t = (pqX_t - c)E_t = 0.$$

These equations can readily be solved for the bionomic equilibrium level of effort ($E_t = \overline{E}_t$):

$$\overline{E}_t = \frac{r}{q}\left(1 - \frac{c}{pqK}\right). \qquad \text{(Equation 3.11)}$$

The bionomic equilibrium biomass level ($X_t = \overline{X}_t$), is also determined from the condition $\Pi_t = 0$, and is given by:

$$\overline{X}_t = \frac{c}{pq}.^{17} \qquad \text{(Equation 3.12)}$$

This result provides for the important conclusion that, if the biological parameters r and K are assumed given, then the bionomic equilibrium effort level becomes a function only of the cost:price ratio (c/p). From this, it can be shown that if costs decline, say over time as a result of technological

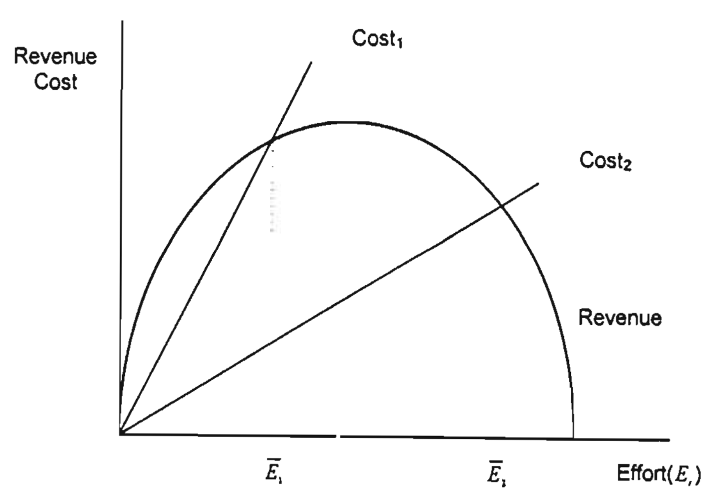
[17] The bionomic equilibrium catch is given by $\overline{C}_t = q\overline{E}_t\overline{X}_t$.

advances (as has been the case in many fisheries), then the fishery will be driven into a state of biological over-exploitation. To show this, if costs are sufficiently high relative to the price of fish, namely:

$$\frac{c}{K} > p,$$

the fishery will not be exploited at all. However, if technology generates a sufficient fall in costs, then harvesting becomes profitable with bionomic equilibrium established at a level such as $\overline{E}_1$ ($Cost_1$) in Figure 3.4. At this stage, the level of effort is less than that required to generate the maximum sustainable yield (E_{MSY}), and biological over-fishing does not occur. But if the cost:price ratio becomes sufficiently low ($Cost_2$), bionomic equilibrium would be established at a level such as $\overline{E}_2$, where biological over-fishing occurs. Furthermore, it should be noted that $\overline{X}_t > 0$ for every possible parameter combination (except $c = 0$). Thus the Gordon model predicts that extinction of the fish population will not occur, the reason being that at sufficiently low stock levels, the sustainable harvest (given by qE_tX_t) is too small to render fishing profitable; harvesting therefore ceases.[18]

Figure 3.4: Bionomic Equilibrium



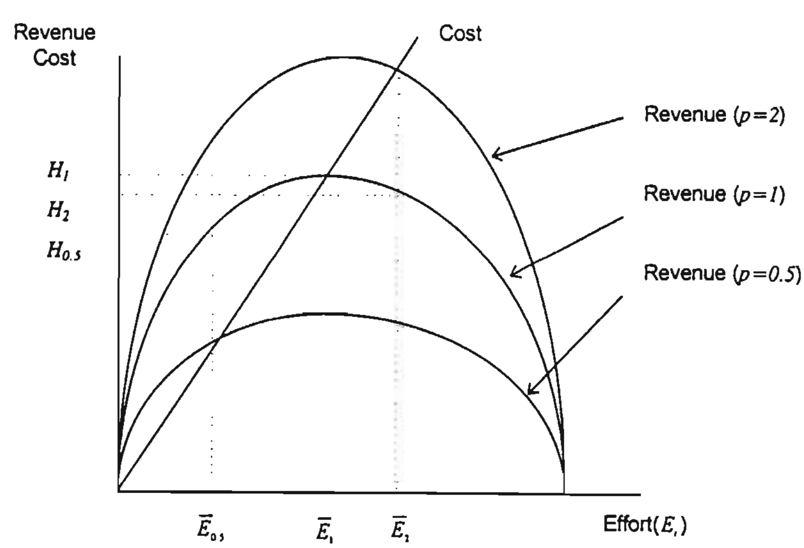
Source: Clark (1985, 3).

[18] Actual extinction of formerly productive fish stocks seems to be a rarity, but this 'prediction' is of course a feature of the model, and not a basic bioeconomic principle, as has sometimes been asserted.

It can also be shown that increases in price, brought about, for example, by a growth in the demand for fish (as has been the case in many markets), result in a rise in sustainable revenue, although not necessarily an increase in harvest. Figure 3.5 provides a summary of the implications for bionomic equilibrium given a rise in prices from $p=0.5$ to $p=1$ and $p=2$. The outcome is clear: as prices increase, sustainable revenue tends to rise, but the total catch first increases (from $H_{0.5}$ to H_1) and then falls (from H_1 to H_2).[19]

In summary, the static Gordon model yields two important results. First, the model predicts that economic over-fishing will occur in any unregulated fishery. That is, under open-access conditions, the level of fishing effort will exceed the level of effort required to generate maximum economic rents, and the fishery will yield zero economic rents. Second, under open-access conditions, biological over-fishing will occur whenever cost:price ratios are sufficiently low. That is, the level of effort will expand beyond that required to catch the maximum sustainable yield, such that the biomass is reduced to below the maximum sustainable yield population ($K/2$ in Figure 3.1).

Figure 3.5: Bionomic Equilibrium under Changing Prices



Source: Adapted from Clark (1976, 25).

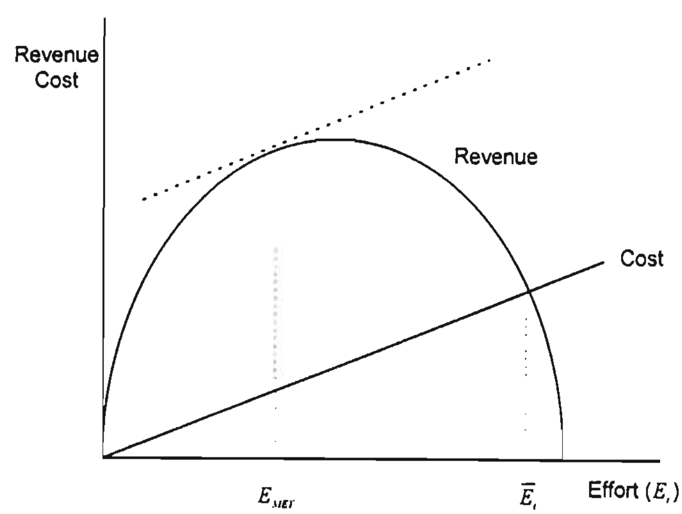
[19] This is commensurate with the widely recognised backward-bending supply curve for fisheries (Copes, 1970 & 1972; Clark, 1976). Chapter 6 is devoted to a fuller investigation of variable prices and the derivation of the market supply curve.

The history of many fisheries provides strong support for this outcome (Clark, 1975, 29-33; World Bank, 1992, 13-5; Arnason, 1994b; Hannesson, 1994a). Specifically, the rapid growth in world catch that occurred over the period 1950-70 (World Bank, 1992, 12) was primarily facilitated by the development and introduction of new technologies that could be applied to finding and catching fish: radar and loran for navigation; sonar for underwater location and net minding; synthetic fibres for lines and nets; intercooled turbo-supercharging for engines; and hydraulics for winches and lifts (Neher, 1994). These developments – which ultimately served to lower the costs of fishing through improving efficiencies in the finding and catching of fish – were spurred by the increase in the demand for fish which resulted from growing populations and increasing *per capita* incomes. These effects, in turn, helped to put fish prices on an upward trend. In short, developments that took place, particularly between 1950-70, helped to drive cost:price ratios down, which provided the necessary incentives to fishers to increase the rate of exploitation of fish stocks. The end result was that by the middle of the 1970s many fisheries had been driven into a state of over-exploitation, evidenced by a stagnation and, in instances, decline, in the global catch (World Bank, 1992, 12). Knowing that open-access results in a bionomic equilibrium level of effort that is economically (and possibly biologically) inefficient, it becomes imperative to establish the optimum, that is, rent maximising, level of effort. Section 3.3.2 explores this issue.

3.3.2 The Static Gordon Model: Maximum Economic Yield

The next step in the Gordon model is to suggest that restricting effort to some level below $\overline{E}_t$ will produce positive economic rents. The effect is particularly strong when $\overline{E}_t > E_{MSY}$, because a reduction in effort has the double effect of increasing revenues and decreasing fishing costs. Continuing along these lines, the optimum level of fishing effort occurs where total sustainable revenue less total costs, that is, sustainable economic rent (Π_t), is maximised. In Figure 3.6, maximum sustainable economic rent occurs at the level of effort E_{MEY}. As Gordon (1953, 130) noted: 'Thus, the optimum economic fishing intensity is less than that which would produce the maximum sustained physical yield.' In the literature, E_{MEY} is typically referred to as the position of maximum economic yield; and Gordon (1954) argued that both the fishers' and the social welfare optimum occur at this point E_{MEY}, where economic rent is maximised. (An important point to note here is that the rent is a social surplus yielded by the resource, not in any part due to artificial scarcity, as in monopoly profit or rent.)

Figure 3.6: Maximum Economic Yield in the Static Gordon Model



Source: Clark (1985, 9).

Because of its particular functional form, the Gordon model predicts that:

$$E_{MEY} = \frac{\overline{E}_t}{2}.$$

That is to say, the bionomic equilibrium level of effort is exactly twice the rent maximising level of effort.[20] This reinforces the result that economic over-fishing is expected to occur in the open-access fishery; and points to the main result of the Gordon model: in order to generate maximum economic rents, it

[20] In practice, however, the effort capacity of fishing fleets is often much larger than twice the optimum level (Clark, 1985, 7 & 28-32).

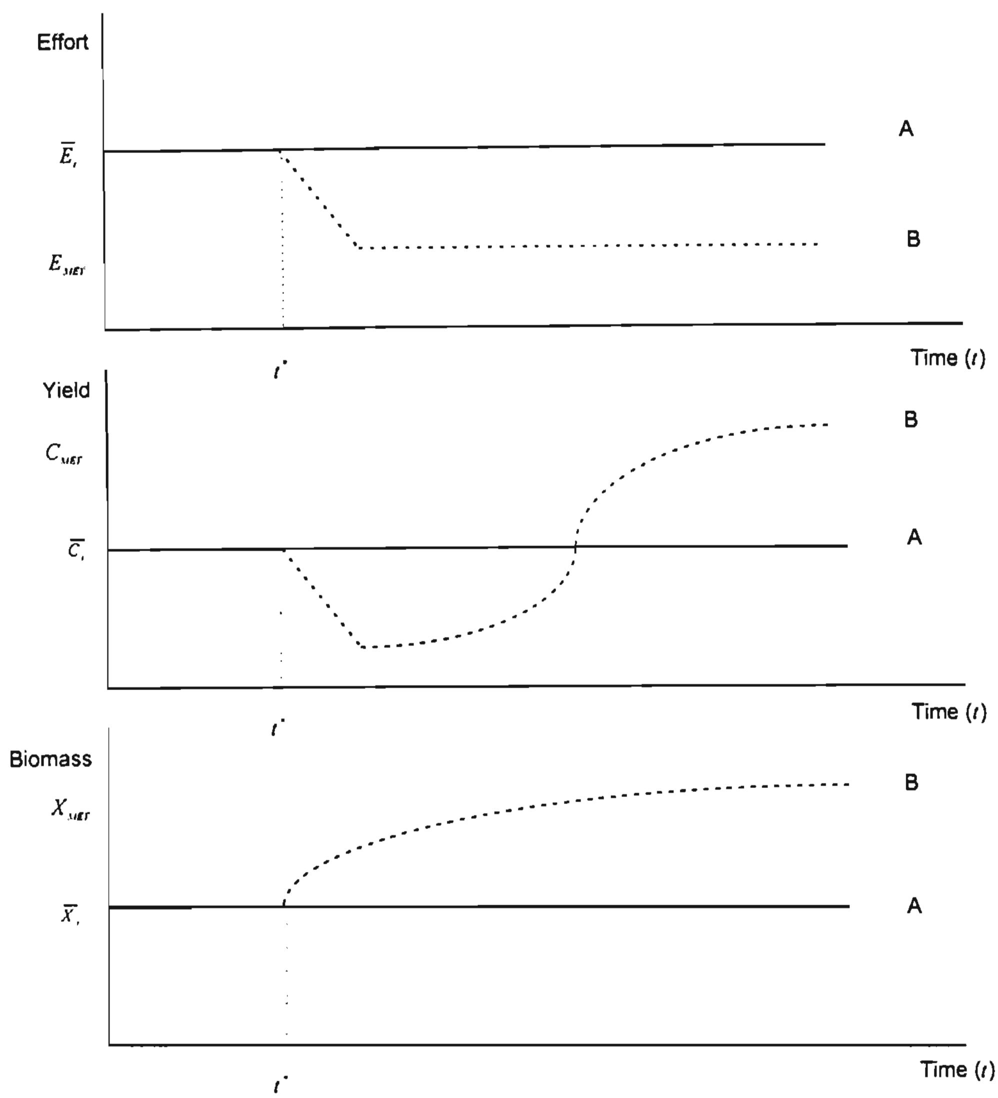
is necessary to introduce regulations, such as limited access or the creation of private property rights, which result in a (substantial) reduction in effort in the fishery.

For completeness, it should be noted that the Gordon model assumes that the optimum outcome (E_{MEY}) is achieved by placing the fishery under a rent maximising sole owner. However, two points must be made in this regard. First, the sole owner should be thought of as either a private firm or government agency that owns complete rights to the exploitation of fish (such that the fishery effectively becomes a limited access fishery). The term 'monopolist' is deliberately avoided here, because this term refers to a firm that possesses market control – the monopolist can fix prices. The present Gordon model assumes no such market power; indeed, the model assumes that firms are 'price takers', facing an autonomous (that is, fixed and constant) price level (p).

Second, but more important, the assumption of sole ownership is not required to show that rent maximisation occurs at a level of effort that is different to the open-access outcome. Indeed, by adopting the assumption, the model presupposes that sole ownership is a 'desirable' (in the same sense as 'optimal') regulatory framework. This may not necessarily be the case. Preferably, all that need be noted is that some arrangement (other than open-access) which facilitates the necessary reduction in effort is required to produce an optimal outcome. The issue of regulation is dealt with in detail in Chapter 7.

It must be acknowledged, however, that the results presented above are partial, because Gordon (and many subsequent writers) neglected the dynamics of the biological and economic processes (Butlin, 1975, 89). Reconsidering the static Gordon model presented in Figure 3.6, if the fishery were at $\overline{E}_t$, it would be desirable to reduce effort such that the fishery moved to E_{MEY}. However, although a reduction of effort would ultimately lead to an increase in yield, its immediate effect would be to decrease yield. The resulting dynamics involving effort, biomass and yield are shown in Figure 3.7, where the curves labelled 'B' show the result of a reduction in fishing effort, and the curves labelled 'A' correspond to no reduction in fishing effort.

Figure 3.7: Effects of Reducing Fishing Effort Across Time



Source: Clark (1976, 31).

Seen in this fashion, it seems obvious that the appropriate analysis is not static, but rather capital-theoretic or dynamic. To be more specific, the stock does not adjust from the original biomass to the rent maximising biomass instantly but, rather, over time. Furthermore, and as outlined in Section 3.5.3 below, revenues from harvesting the stock are earned over time, and so it becomes necessary to view revenues as a flow – rather than a stock – of earnings. Accordingly, if the Gordon model is to provide an accurate description of the fishery, and the associated optimum effort and harvest rates, it is necessary to extend the analysis into a dynamic setting. Section 3.3.3 is devoted to this consideration.

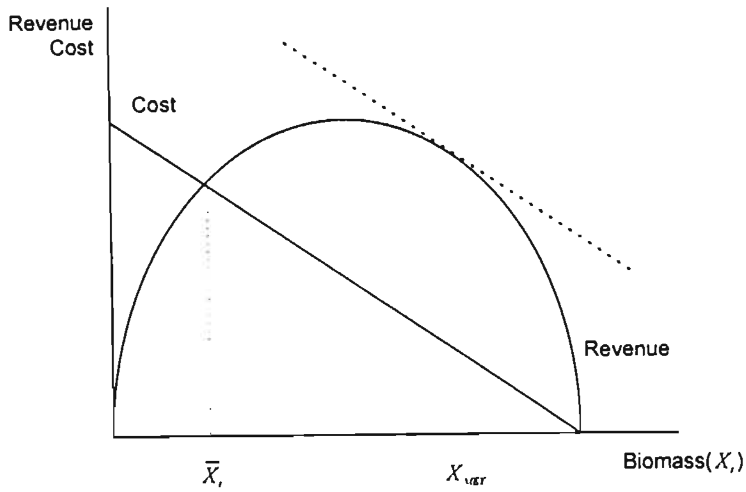
Before proceeding, it should be noted that in order to extend the Gordon model into a dynamic setting, it is necessary to restate the cost and revenue functions – and so the optimal solution to the model – in terms of biomass, rather than effort (as above). The reason for this is that the solution to the dynamic model, as developed below, requires the use of optimal control theory, where biomass constitutes the state (or control) variable (Munro, 1982). Accordingly, the revenue and cost functions in the model are rewritten, such that:

$$Revenue = pC_t = prX_t\left[1-\frac{X_t}{K}\right], \text{ and} \qquad \text{(Equation 3.13)}$$

$$Cost = cE_t = \frac{cr}{q}\left[1-\frac{X_t}{K}\right]. \qquad \text{(Equation 3.14)}$$

These curves, as well as the (static) rent maximising biomass (X_{MEY}) and zero-rent biomass ($\overline{X}_t$), are shown in Figure 3.8 below. Note that the outcomes X_{MEY} and $\overline{X}_t$ correspond with the effort levels E_{MEY} and $\overline{E}_t$ respectively (see Figure 3.6).

Figure 3.8: The Static Gordon Model: Biomass-Revenue (Cost) Relationship



Source: Clark (1985, 15).

3.3.3 A Dynamic Bioeconomic Fisheries Model[21]

The standard device used to handle questions of intertemporal economic costs and benefits is time discounting; and there are strong economic arguments to support the hypothesis that the decisions of firms are based on the objective of maximising the discounted present value of future net income flows (Hicks, 1946).[22] Accordingly, it is argued that the optimal harvest rate in a dynamic setting, is established by maximising the present value of all net future economic rents. Following these arguments, the present value of net revenue flows (Π_t), over a time interval of length dt, can be written as $\Pi_t e^{-\delta t} dt$. The total net present value of Π_t over the time horizon from $t = 0$ to $t = T = +\infty$ (the end of the discounting period) is therefore given by:

[21] Scott (1955) published an article one year after Gordon's (1954) article that represents a pioneering attempt to re-cast the Gordon model in a dynamic framework. This – and other attempts (for example, Crutchfield and Zellner, 1962) – produced models that were complex and difficult to apply. However, with the development of optimal control theory, it became a substantially easier task to cast economic models of fisheries in a capital-theoretic (or dynamic) framework which yielded results that are both general and readily comprehensible (Clark and Munro, 1975; Munro and Scott, 1985, 637). The model presented in this section is based on these developments.

[22] This is not to say that there are no objections to the use of discounting as a measure of intertemporal costs and benefits. Indeed, there are strong arguments against the discounting technique. These arguments are explored in Chapter 6, where the discounting technique is dealt with in greater detail, and where a defence of the discounting technique is provided.

$$\int_0^\infty \Pi_t e^{-\delta t} dt. \qquad \text{(Equation 3.15)}$$

It is both convenient and natural to assume that $T = +\infty$ (otherwise undesirable 'horizon effects' occur, that is, costs or benefits occurring beyond a certain time are not considered, and this is contrary to the ethos of discounting which gives weights to all future events). Furthermore, given the argument that the sole owner is a rent maximiser, and substituting from Equation 3.10, the objective function can be formulated as:

$$Maximise \int_0^\infty (pqX_t - c)E_t e^{-\delta t} dt \qquad \text{(Equation 3.16)}$$

subject to the equation:

$$\frac{dX_t}{dt} = G(X_t) - qE_t X_t, \quad (X_0 \text{ given}) \qquad \text{(Equation 3.17)}$$

where E_t satisfies $E_t \geq 0$.[23]

The differential equation (Equation 3.17) is often referred to as the general production model; and the solution to the foregoing mathematical optimisation problem is referred to as the optimal solution.[24]

The maximisation problem can then be solved by integrating by parts (as shown in Appendix 1). The optimal size of the stock (X^*_{MEY}) is determined by the equation:

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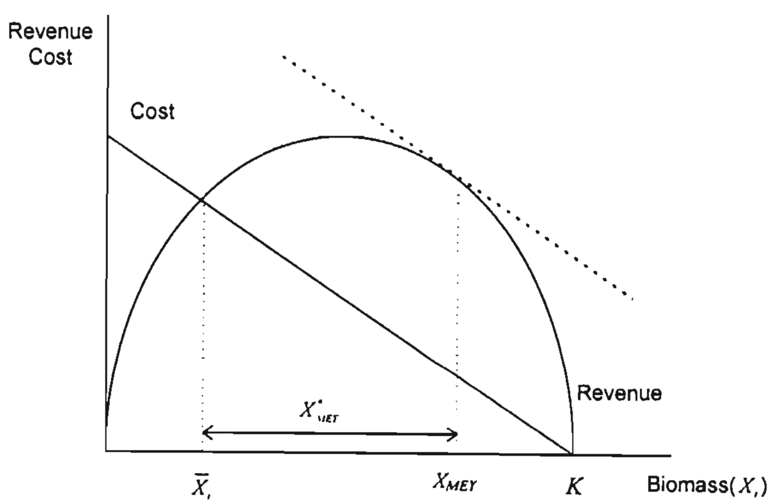
[23] Note that although it is assumed above that $G(X_t)$, the natural growth rate of fish, is based on the Schaefer model, the growth function can be easily altered to represent different growth functions, such as those discussed by Pella and Tomlinson, Fox and Shepherd (Clark, 1985, 19).

[24] No value judgement is attached to this purely technical use of the word optimal; however, the social implications of the profit-maximising optimal harvest policy are discussed in greater detail in Chapter 7.

$$G'(X^*_{MEY}) - \frac{c'(X^*_{MEY})G(X^*_{MEY})}{p - c(X^*_{MEY})} = \delta . \qquad \text{(Equation 3.18)}^{25}$$

It is shown in Appendix 1 that the optimal dynamic biomass, X^*_{MEY}, lies between the bionomic equilibrium $\overline{X}$ and the Gordon static optimum X_{MEY}. Furthermore, X^*_{MEY} equals X_{MEY} for zero discounting ($\delta = 0$), and $\overline{X}_t$ is the limiting position of X^*_{MEY} as $\delta \to +\infty$. In short, X^*_{MEY} diminishes as the discount rate increases. To put matters more simply, the optimal solution for a finite, positive discount rate will always lie between the two extremes of $\overline{X}_t$ and X_{MEY}. Furthermore, as costs fall, X^*_{MEY} will approach $\overline{X}_t$. That is, lower costs will reduce the equilibrium biomass, as will higher discount rates.[26] These outcomes are summarised in Figure 3.9.[27] Section 3.3.4 is devoted to exploring the policy implications of the bioeconomic model.

Figure 3.9: The Optimal Biomass in the Dynamic Bioeconomic Model



Source: Clark (1985, 23).

[25] This equation is the so-called 'Golden Rule' equation as it provides a rule for determining the extent to which society should invest (or disinvest) in a resource (Munro and Scott, 1985, 640-1).

[26] Flaaten (1991) shows that under the special conditions of zero costs and a zero discount rate, the bioeconomic optimum will coincide with the maximum sustainable yield biomass.

[27] Chapter 6 is devoted to a detailed examination of the impact of variable parameter values on the optimal dynamic outcome.

3.3.4 Policy Implications

Biological principles are an insufficient basis for managing fisheries because biological management incorporates the benefits of harvesting fish stocks, but completely ignores the costs of harvesting. In support of this, it has been argued that:

> ... maximum average production [maximum sustainable yield] is not an adequate development objective. It fails to take into account the cost of fishing, and leads to excessive use of capital and labour. In the long run, net economic benefits are achieved at a level of fishing effort lower than maximum average yields (World Bank, 1992, 24).

In spite of this obvious – and fundamental – flaw, some writers have continued to defend biological principles as an optimal management strategy. For example, with respect to maximum sustainable yield, Barber (1988, 153) argued: 'Maximum sustainable yield ... is a simple concept that is readily understood by the fishing industry, administrators, and managers'; and went on to suggest that: 'There seems to be no operational rival to MSY [maximum sustainable yield].' This view is rejected here because 'simplicity' and 'comprehensibility' are hardly grounds for selecting maximum sustainable yield – or any other strategy for that matter – as an appropriate basis for fisheries management. Given the insights provided by Gordon and others, it is clear that there is an obvious and appropriate alternative to biological principles such as maximum sustainable yield: maximum economic yield. In this regard, Arnason (1994a, 5) has been particularly vociferous:

> The apparent conclusion is thus that biological fishery management measures, although well suited for preserving fish stocks, are useless from an economic point of view. In fact, the outcome is typically worse. Setting and enforcing biological fishery restrictions is invariably costly ... And because biological fishery management generates no economic benefits to speak of, these costs represent a net loss. Thus, we must conclude that fishery management based on biological conservation measures will generally generate a negative economic return.

The second major policy implication of the model is that, left to their own devices, fishers will compete in an open-access fishery until all rents are dissipated. This is the so-called Class I open-access problem. To achieve maximum economic yield, it is necessary that managers restrict the access and/or level of effort of fishers.

These two points have been well summarised by Cunningham (1983, 69-70):

> Until very recently, the regulation of marine fisheries was dominated largely by biological considerations, and this was [and still often is] reflected in both the goals which management sought to achieve and the methods used to achieve them. [B]iological techniques are increasingly falling into disfavour. This is mainly because it has become clear that economic control methods can result in the achievement of most biological goals as a welcome side effect of their principle economic aim of preventing rent dissipation.

In other words, if the objective of fisheries managers is to maximise economic welfare, then two steps are required. First, it is necessary to establish the rent maximising level of the biomass and, from that, the optimum level of effort and catch. Second, in the instance of open-access fisheries, it is necessary to regulate the fishery by, for example, restricting access, limiting effort or creating private property rights.

In line with the arguments presented in this section, a principle aim of this study is to develop a bioeconomic model for South Africa's commercial fisheries; and the model presented above is argued to be well-suited to this task for three main reasons. First, the model is easily adapted to take into account a range of different biological and economic characteristics. Second, the model is easily extended, which facilitates the exploration of a number of important applications, such as the impact of changing economic or biological conditions on the optimal outcome. Third, although the model does suffer from a number of serious limitations, these are limitations that apply to bioeconomic modelling generally, and not to this model specifically. These points are reviewed briefly in Section 3.4.

3.4 The Dynamic General Production Model: Advantages and Limitations

The bioeconomic model developed above is particularly useful from both a modelling and management point of view in that the model is easily adapted to reflect a range of biological and economic characteristics of fisheries. In effect, the model is not restricted to the Schaefer growth function, and is easily adapted to other growth functions such as those studied by Pella and Tomlinson (1969), Fox (1970) and Shepherd (1982).[28] In addition, the model is easily extended to allow for the impact of environmental effects, such as changes in sea temperature (Bell, 1972) or the incidence of pollution (Saville and Lumby, 1997). Although the model only allows for extinction under the extreme condition of zero-cost harvesting, the model is easily extended to allow for alternate formulations of the growth function – for example, critical depensation, whereby the stock exhibits negative growth below some critical or threshold biomass and so declines to zero (extinction) – in terms of which extinction is feasible where costs are positive ($c > 0$) (Clark, 1976, 17-21).[29]

Furthermore, the model assumes a constant linear cost function; however, this assumption is easily relaxed to allow for non-linear and/or non-autonomous costs (Clark and Munro, 1975 & 1982; Munro and Scott, 1985, 642-4). The model also rests on the restrictive assumption of autonomy in prices, although it can be extended to allow for non-autonomy in prices (Clark and Munro, 1982, 42-6). These issues are explored in greater detail in Chapter 6 where the assumptions of autonomy in prices and autonomy and linearity in costs are examined. Finally, the model can be extended to allow for a 'heterogeneous' fishery, such as South Africa's hake or rock lobster fisheries which consist of distinct stocks.[30]

In short, the model presented above is argued to be widely applicable, as well as easily extended to allow for variations in biological features such as different specifications of the fish population growth function, as well as variations in economic features such as variable prices or changing cost

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[28] For applications of these alternative growth functions, see, for example, Andrew and Butterworth (1987) and Geromont and Butterworth (1995).

[29] For a fuller discussion of the economic viability of extinction, see Smith (1968 & 1976), Plourde (1970 & 1971), Gould (1972), Peterson and Fisher (1977), Clark and Munro (1978), Hoel (1978a), Berck (1979), Cropper *et al* (1979) and Chang and Wang (1981). For practical investigations into 'economically exhaustible' but biologically renewable resources, see Clark (1973a & 1973b) and Bjørndal *et al* (1993).

[30] See Chapter 4.

conditions. These features render the model extremely attractive in terms of the task of modelling South African commercial fisheries.

The model does, however, suffer from a number of limitations, the most serious of which is the failure to incorporate the problems of uncertainty and multi-species interaction into the model (Munro, 1982). With regard to the former, it should be noted that the types of uncertainty relevant to renewable resources have typically been grouped into three broad categories (Spulber, 1982a): (i) uncertainty resulting from imperfect knowledge concerning the state of the system (involving environmental 'surprises' such as a collapse in stocks); (ii) random environmental disturbances; and (iii) lack of information concerning the population growth function. However, the issue of uncertainty is extremely complex and there are large gaps in the literature. Only the second of these three forms of uncertainty has received attention on any significant scale (Spulber, 1982a; Munro and Scott, 1985).[31] Furthermore, efforts to explore the issue of uncertainty have generally not translated intc real changes in modelling. Indeed, in many instances, efforts to move from deterministic modelling to stochastic modelling have simply involved cosmetic changes, such as the inclusion of a random error term in model specification.[32] Nevertheless, it is important to note that this criticism is not model specific, but rather pertains to bioeconomic modelling generally. In short, the inadequate treatment of uncertainty remains a problem for modelling generally and, likewise, for the model presented above.

A second criticism of the general production model is that it ignores multi-species interaction. Several authors have extended single-species analysis to include multi-species interaction, including competing species and predator-prey systems (Clark, 1976; May *et al*, 1979; Sobel, 1982; Hannesson, 1983; Flaaten, 1991; Harwood, 1992; Wickens *et al*, 1992).

[31] The bulk of the literature on uncertainty is devoted to exploring biological and environmental uncertainty. See, for example, Jacquette (1972 & 1974), Reed (1979), Lewis (1981), Charles (1983), Andersen and Sutinen (1984), Getz and Haight (1989) and Hannesson and Steinshamn (1990).

[32] Clark and Kirkwood (1985) offered an interesting exception in this regard by employing a Bayesian approach to model optimal quota decisions in the instance of uncertainty with respect to stock abundance.

Some writers have also investigated harvesting interactions in multi-species systems (Quirk and Smith, 1970; Anderson, 1975 & 1982; Clark, 1976; Silver and Smith, 1977; Huppert, 1979; Sobel, 1982; Clark, 1985; Lipton and Strand, 1989; Smale, 1993; Fischer and Mirman, 1996). But, multi-species modelling is an extremely complex task. In support of this point, Munro and Scott (1985, 646) argued: 'Modelling [multi-species] interactions in the real world proves to be a daunting undertaking. Indeed, more often than not, the undertaking proves to be impossible.'[33] For this reason, the biological model employed in this study does not attempt to incorporate species-interaction. Again, however, it should be noted that this short-coming is not model specific, but rather pertains to bioeconomic modelling generally.

Criticisms aside, it is argued here that the general production model provides an appropriate basis for fisheries modelling for at least two important reasons. First, the model is easily adapted to take the different biological and economic characteristics of fisheries into account. Second, the model provides an excellent framework for exploring a number of important extensions of the autonomous model. For these reasons, it is employed in this study as the foundation for constructing a bioeconomic model of South Africa's commercial fisheries which, in turn, serves as the basis for establishing optimal use rates, that is, extraction rates that maximise the rents generated by the resource across time.

[33] An interesting example of multi-species interaction is provided in Muraoka (1990, 141) where it is noted that: 'Ironically, today's sea urchin fishery owes its existence to excessive harvests of two other species, the sea otter and abalone.'

CHAPTER 4: AN OVERVIEW OF THE BIOLOGY AND ECONOMICS OF THE SOUTH AFRICAN COMMERCIAL HAKE FISHERY

4.1 INTRODUCTION

As noted in Chapter 2, South Africa's fishing industry is extremely diverse. For this reason, it is necessary to construct a number of models to represent the various sectors of the industry. This is a considerable task, which is well beyond the scope of the current study. Rather, this study focuses on the country's most important commercial fishery, the hake fishery, which represents approximately one-third of the commercial catch by value and one-quarter by mass. In order to apply the dynamic bioeconomic model to the commercial hake fishery, it is first necessary to provide some background on the major biological and economic features of the hake fishery. This chapter is devoted to this task. Section 4.2 analyses the main biological characteristics and life history of hakes; while Section 4.3 provides a review of the economic history of the hake fishery and summarises the main economic features of the modern day fishery. Section 4.4 is devoted to a brief review of regulations adopted in the hake fishery.

4.2 BIOLOGICAL FEATURES OF HAKES AND THE HAKE FISHERY[1]

South Africa's coastal waters are extremely productive, particularly along the west coast where major upwelling occurs, and although the waters of the south and east coasts are less productive, upwelling does take place (particularly around capes) (Schumann *et al*, 1982). These conditions provide for substantial and relatively diverse stocks of commercial (as well as

[1] See Payne and Punt (1995) for a comprehensive review of the biology and life history of Cape hakes.

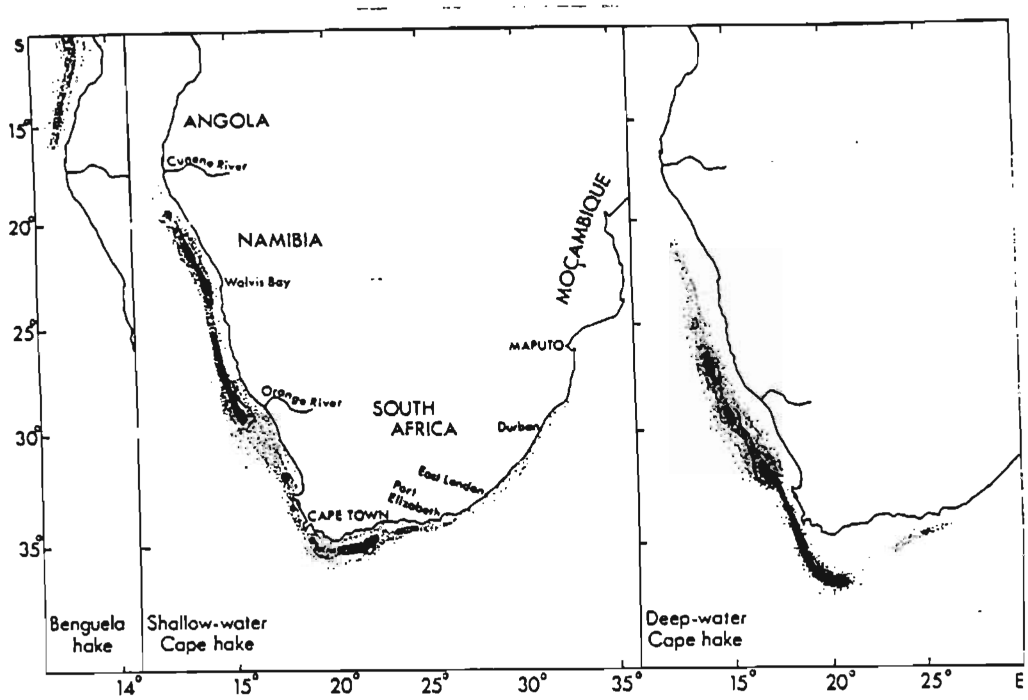
recreational) fish species. In turn, these commercial species serve as the basis for a relatively large and diversified fishing industry located on the west and south coasts. Nevertheless, the demersal sector constitutes the backbone of the fishing industry, where hakes (*Merlucciidae*) typically account for about one-quarter of the total commercial catch by mass and over one-third by value.

There are two species of hake in South African waters, namely *Merluccius capensis* and *M. paradoxus* (Cohen, 1986, 325).[2] These two species are commonly referred to as shallow-water hake and deep-water hake respectively. Shallow-water hake, as the name suggests, are generally found close inshore up to a depth of about 400 meters. Deep-water hake are typically not found shallower than 150 meters, but their depth range extends to almost 900 meters (Botha, 1985). The distribution of both species is virtually continuous around the west and south coasts of South Africa, with limited horizontal migration taking place. Figure 4.1 provides a summary of the distribution of hake species in southern African waters.

Cape hakes spawn in a double wave: the first wave in November/December, sustained by both species; and the second wave in February/March, sustained mainly by *M. paradoxus* (Botha, 1986). Hakes are fast growing fish. At age one year, hakes attain approximately 160 millimetres in length; over the second and third year, annual growth is approximately 130 millimetres. Hakes mature at between three and four years old, at which point they are 400-450 millimetres in length (Bergh and Barkai, 1993, 93) Thereafter, growth slows to around 80-90 millimetres per year at the age of five and to 20-30 millimetres per year at the age of ten. By that stage, the fish are almost 1 000 millimetres long (Botha, 1971; Payne, 1986; Payne and Punt, 1995, 27).

[2] A third species of hake, *M. polli*, occurs in south-east Atlantic waters, but is only caught off northern Namibia and Angola (Punt, 1988, 15).

Figure 4.1: Distribution of Hake Stocks by Species



Source: Payne (1989, 137).

Cape hakes are opportunistic feeders (Payne and Punt, 1995, 29-31). Juveniles feed mainly on crustaceans and the diet becomes increasingly piscivorous with age. For both species, the general movement of fish is offshore as the fish grow, so adults are always found in deeper water than juveniles. This results in mixing between medium to large *M. capensis* and small *M. paradoxus* in shallower water which, in turn, leads to predation on *M. paradoxus* by *M. capensis* (Botha, 1982; Cohen, 1986, 325; Payne *et al*, 1987; Lipinski *et al*, 1992). In addition, both species are cannibalistic (Punt *et al*, 1992, 622; Payne and Punt, 1995, 31). Although the two species mix, integrity between the species is maintained, because the adults of the two species do not mix and so spawn at different depths (Botha, 1973).

For the purposes of modelling, two further points should be noted. First, although differences between rates of growth in *M. paradoxus* and *M. capensis* have been noted (Payne and Punt, 1995, 28), these differences are not usually taken into account explicitly in modelling. Further, whilst females grow slightly faster than males (Botha, 1986; Punt and Leslie, 1991) the difference in growth rates between sexes is also ignored in modelling (Punt, 1992; Leslie, 1994; Punt, 1994). The primary reason for ignoring differences in growth rates between species and sexes lies in the fact that, because of 'mixing', it is impossible to target fish on the basis of species or sex, and so there is little justification for modelling or, for that matter, attempting to manage the fishery along these lines.

Second, because *M. paradoxus* normally occur in deeper water than *M. capensis*, it is reasonable to expect that the ratio of *M. paradoxus* to *M. capensis* will increase as the depth of fishing increases. Similarly, it is reasonable to expect that the proportion of females in the catch increases with depth because the general movement of fish is offshore as fish grow, and, as noted, females are faster growing than males (Japp, 1995, 36). However, whilst the breakdown of the catch by species and sex is depth dependent, it is impossible to target fish by species or sex. That is to say, it is impossible to be guaranteed of a certain ratio of, for example, *M. capensis* or females. In addition, the differences in growth rates between sexes has been found to be considerably less significant than was originally thought to be the case (Punt and Leslie, 1991). Furthermore, although areas of capture of the species of Cape hakes are known, it is impossible to distinguish between the two species at the landing point (Cohen, 1986, 326; Payne and Punt, 1995, 23). Based on these arguments, the view is held that it is unreasonable to model the fishery along the lines of sex or species composition.[3]

While no distinction is made between species or sex for modelling purposes, the fishery is distinguished along geographic lines. Specifically, for modelling and management purposes, the hake fishery is divided into two geographic

[3] This point should be qualified here, as later (see Chapter 5) it is argued that an uneven distribution of species across the west and south coast fisheries is partly responsible for the west coast biomass growing at a slower rate than the south coast biomass. Given that the hake fishery is divided into west and south coast fisheries, this has implications for modelling. In other words, differences in growth rates between species, whilst not considered explicitly, are considered implicitly in the modelling process.

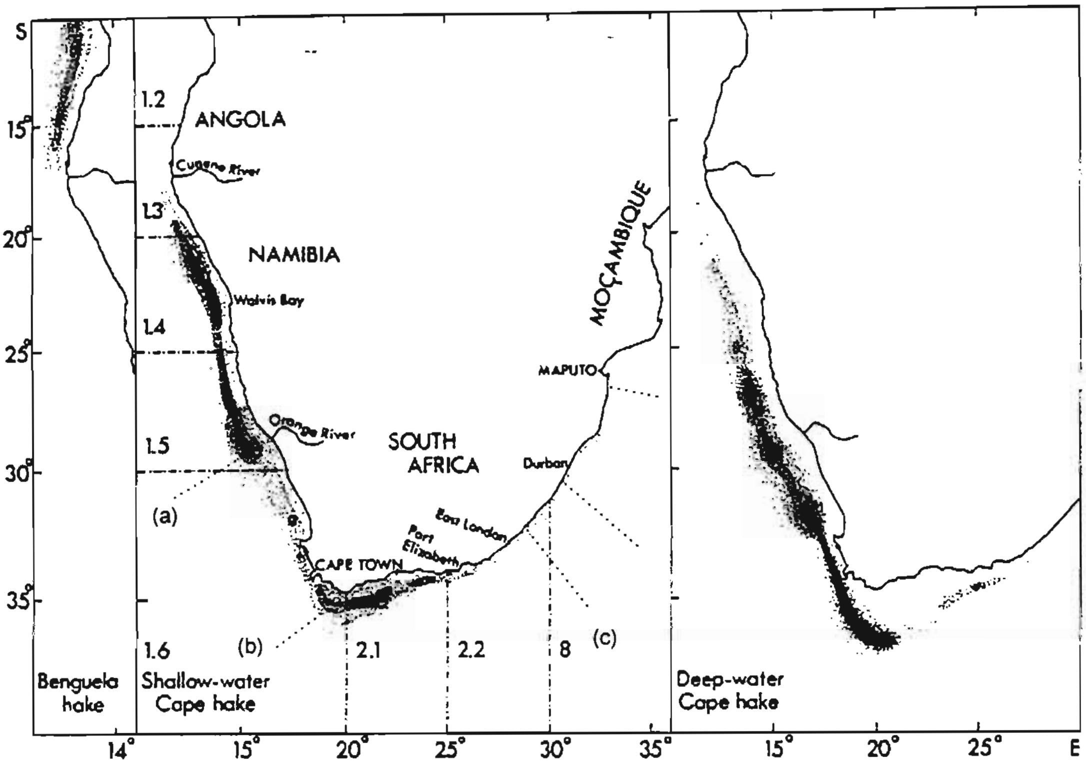
areas: the west coast and south coast fisheries. The division of the hake fishery into west and south coast fisheries dates back to the divisions that were set by the International Commission for the Southeast Atlantic Fisheries (ICSEAF) in the 1970s. ICSEAF divided the offshore region, somewhat arbitrarily, into two areas:

(i) Division 1.6, the so-called west coast fishery, was defined as the region between the line of latitude touching the mouth of the Orange River and the line of longitude touching Cape Point; and

(ii) Divisions 2.1 and 2.2, the so-called south coast fishery, was defined as the region that lay between the boundary with Division 1.6 to the line drawn south from a point east of the Kei River mouth.

A slight modification of the ICSEAF demarcation between Division 1.6 and Divisions 2.1 and 2.2 has been made by the South African authorities. Instead of a line of latitude touching Cape Point, the boundary between the west and south coast fisheries is now taken as a line stretching south-west from Cape Point, crossing a very productive trawling ground known as Browns Bank (Andrew and Butterworth, 1987, 926; Bergh and Barkai, 1993, 94). A slight modification has also been made to the boundary between Division 2.2, on the south coast, and Division 8.2, on the east coast (see Figure 4.2). In addition to the above divisions, the south coast fishery is further divided between an inshore and a deep-sea fishery; the division is given by the 110 meter contour (Van D. Boonstra, 1993, 35).

Figure 4.2: Boundaries of the Major Commercial Hake Fisheries: The West Coast Fishery Lies between Boundaries (a) and (b); and the South Coast Fishery Lies between Boundaries (b) and (c).



Source: Adapted from Payne (1989, 137).

For management purposes, the west and south coast resources, as depicted in Figure 4.2, are regarded as biologically distinct and separate (or 'closed'). As Punt (1992b, 946) noted:

> The Cape hake resource is divided into [two] 'stocks' for management purposes, and it is tacitly assumed that the rate at which fish migrate from one of these stocks to another is negligible compared to the rates of reproduction and natural and fishing mortality.[4]

The observed differences that exist between the two stocks have important implications for modelling purposes. The most important differences are twofold: the instantaneous growth rate (r) of the two stocks is not the same,

[4] Hutton (1993, 4) noted that some writers have suggested that there may be modest interaction between the *M. paradoxus* stocks on the south and west coasts. It should also be noted that vertical migration takes place: hakes leave the bottom at night. For this reason, trawling at night off South Africa is generally not considered to be a viable economic proposition (Payne, 1989).

and the carrying capacity (K) of the fisheries is not the same. Taking this into account, estimates of the instantaneous growth rate of hakes in the south coast fishery are typically modelled as being twice as great as the growth rate in the west coast fishery (Punt, 1992a, 1992b & 1994; Leslie, 1994 & 1996; Payne and Punt, 1995). The reason for this is that fish occurring in the warmer south coast waters grow faster than those occurring in the cooler west coast waters.[5] Also, the south coast biomass is dominated by the faster growing *M. capensis*. Thus, whilst differences in growth rates between species are not considered explicitly in the modelling process, they do become important once the geographical distinction between fisheries is made, because the composition of the biomass in either fishery is different. The carrying capacity of the west coast fishery (approximately 1 665 000 tons) is generally considered to be about five to six times as great as that of the south coast fishery (approximately 275 000 tons) (Punt, 1992a, 1992b & 1994; Bergh and Barkai, 1993; Leslie, 1994; Payne and Punt, 1995).

In line with the differences in the growth rate of stocks and the carrying capacity of the habitats, a considerable difference also exists between the size of the annual catch in the two fisheries (this issue is discussed in greater detail in Section 4.3). Furthermore, the composition of the catch varies substantially between the two fisheries. Although shallow-water hake, *M. capensis*, dominates off Namibia, it is the deep-water species, *M. paradoxus*, which is dominant off the west coast of South Africa, making up about 85 percent of the catch by mass in Division 1.6. On the south coast, in Divisions 2.1 and 2.2, *M. paradoxus* only makes up about 30 percent of the hake catch by mass (Botha, 1985; Bergh and Barkai, 1993, 93). However, annual and seasonal variations cause these figures to fluctuate widely (Payne and Punt, 1995, 23).

Having noted some of the more significant biological features of the hake resource and its life history, Section 4.3 provides a brief review of the

[5] Pers. comm. (Japp, Sea Fisheries Research Institute).

economic history of the hake fishery, and examines the major economic features of the present-day fishery.

4.3 Economic Features of the Hake Fishery

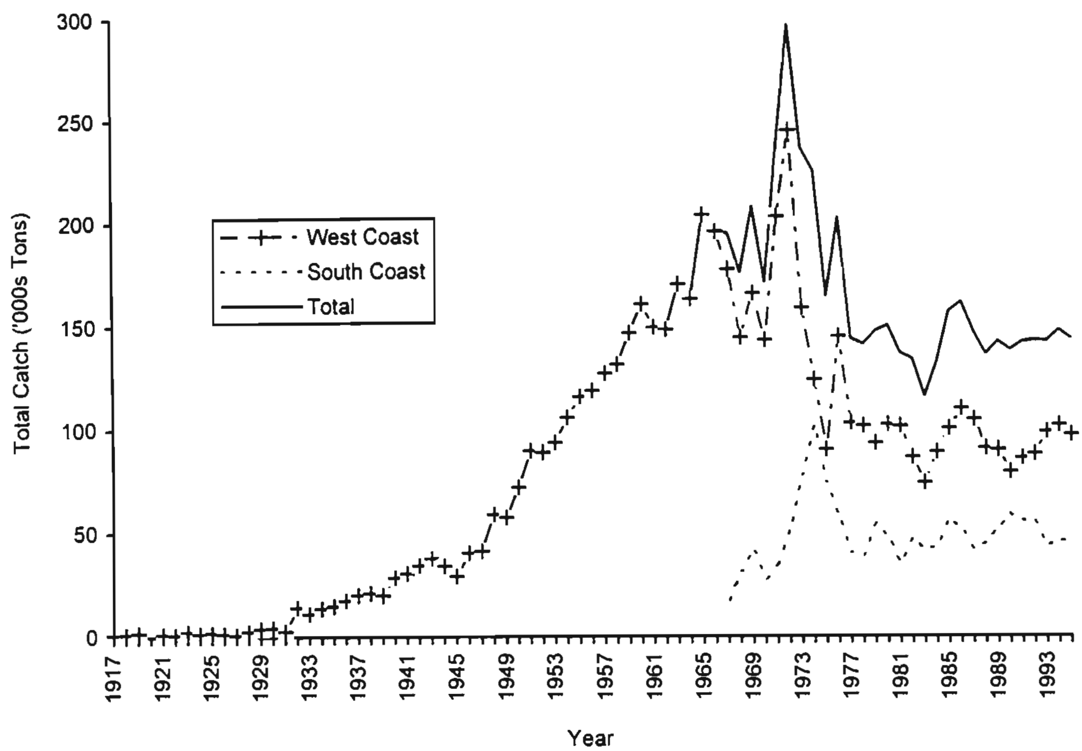
4.3.1 A Brief Economic History of the Hake Fishery

The South African demersal, or trawl, fishery commenced around the turn of the twentieth century.[6] However, up to the time of the First World War, there was little demand for hake in the South African fish-market and hakes were not systematically exploited (Gilchrist, 1914).[7] It was the First World War and the consequent food shortages which initiated an interest in fish, particularly in the hake and sole resources which had been discovered just off the main fishing metropolis of Cape Town. By the end of the First World War, the total hake catch stood at around 1 000 tons (Payne and Punt, 1995, 17). However, the hake catch remained relatively low over the next decade, and it was only in 1932 – when, for commercial reasons, trawlers switched their effort from 'sole-directed' to 'hake-directed' – that an annual haul in excess of 10 000 tons was recorded for the first time (Japp, 1995). Thereafter, the hake catch grew rapidly to 15 330 tons by 1938 and to around 50 000 tons by the late 1940s (Japp, 1995). Over the next two decades, the rate of growth in the total hake catch remained high, reaching 115 000 tons in 1955 and 203 000 tons in 1965. The mid-1960s also saw the inception of the south coast fishery, which served to bolster the growth in the hake catch generated by the extant west coast fishery. To this end, the

[6] Two principal methods are employed to catch hake: trawling and longlining. Trawling involves dragging a weighted net on the sea bottom; whereas longlining involves fishing with a line of up to 30 kilometres in length, baited with as many as 15 000 hooks placed at intervals on the line. The line is deployed with a buoy at one end, and the other end is attached to the fishing vessel. Apart from a limited amount of longlining which occurred during the 1980s, and the more recent longlining experiment (Japp, 1993), trawling has been the only recognised method of taking hake in South Africa (Bergh and Barkai, 1993, 94). For this reason, this study focuses exclusively on the trawl sector of the hake fishery, although the issue of longlining is given some consideration in Chapter 7.
[7] See Gilchrist (1914) for a review of the early history of the South African fishing industry.

south coast fishery immediately began catching in the order of 20 000-30 000 tons of hake per annum (Punt, 1994, 161; Payne and Punt, 1995, 19) Figure 4.3 shows the growth in catch for the west coast fishery from 1917 onwards, and from 1969 onwards for the south coast fishery.[8]

Figure 4.3: Total Hake Catch in South African Waters (1955-95)

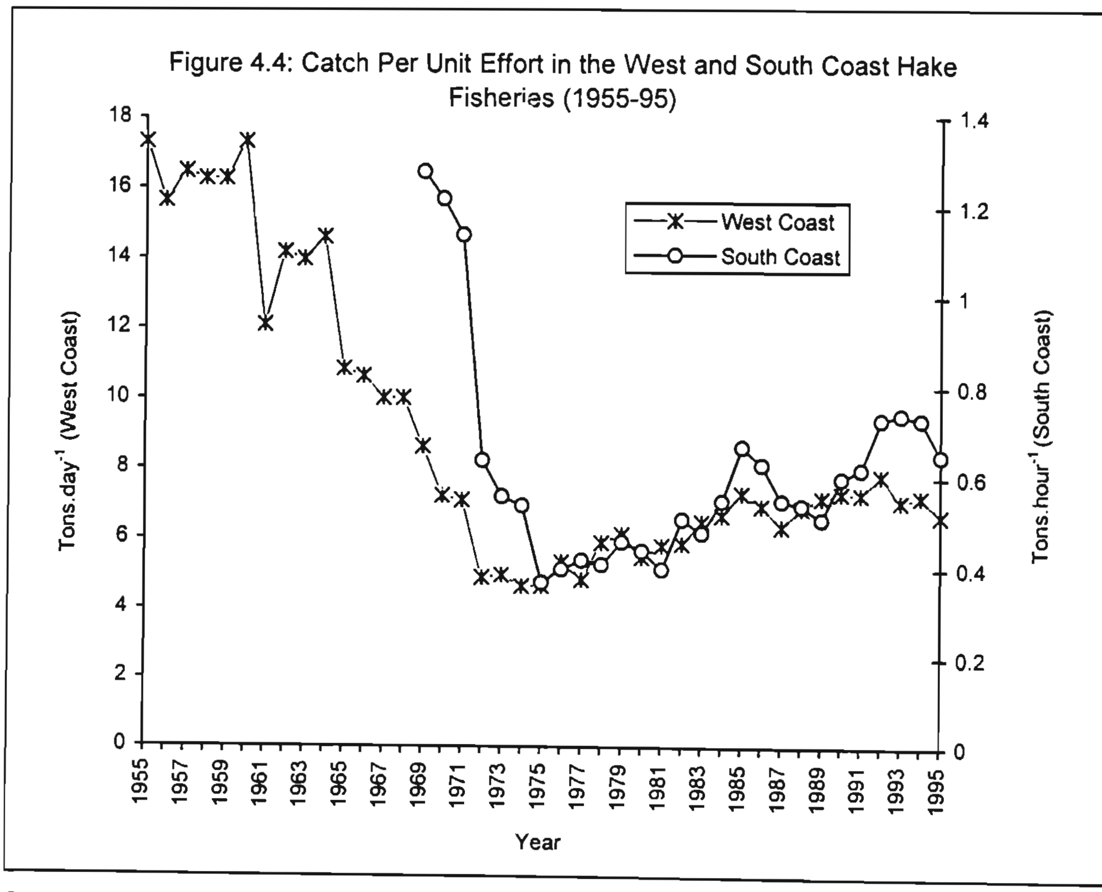


Source: Adapted from Punt (1994, 161); Leslie (1994); Japp (1995); Payne and Punt (1995, 19).

The rapid growth in catch in the west coast fishery, and even faster growth in the south coast fishery, soon attracted distant-water fleets. Initially, these fleets were drawn primarily from Japan and Spain, although later trawlers from the Soviet Union and other former Eastern bloc countries joined the fishery. The increased harvesting capacity brought by these foreign trawlers ensured that the total hake catch continued to grow, to the extent that by 1972 the total catch had reached almost 300 000 tons (243 900 tons in the west coast fishery and 51 400 tons in the south coast fishery) (see Figure

[8] The south coast fishery (other than a few small-scale operators) only commenced in the mid-1960s, and 1969 is the first year for which reasonably accurate data exist (Payne and Punt, 1995, 19).

4.3). However, hakes are prone to large fluctuations in recruitment and availability, and in poor years, many of the foreign fleets were content with catching large quantities of small fish to maintain volumes. The result was inevitable: the resource was unable to withstand the onslaught, and was quickly driven into a state of severe over-exploitation. As is evident in Figure 4.4, catch rates (or catch per unit effort) fell dramatically. In the west coast fishery, the catch rate more than halved between 1965 and 1975, from 10.84 tons per day to 4.66 tons per day. The catch rate had averaged 16.54 tons per day over the period 1955-60. The collapse in the south coast fishery was equally dramatic, with the catch rate falling from 1.28 tons per hour in 1969 to a low of 0.37 tons per hour in 1975, a fall of over 70 percent in just six years.



Source: Adapted from Japp (1995).

Over-exploitation of the hake resource had two main implications. First, hake catches declined dramatically through the 1970s. For instance, in the west coast fishery, the catch fell from its peak of 243 900 tons in 1972 to

89 600 tons three years later; and on the south coast, the catch fell from a peak of just over 100 000 tons in 1974 to 40 500 tons three years later (see Figure 4.3). Second, although South African trawlers (and the trawlers from western nations) preferred to catch larger sized hake, as noted, a considerable portion of the distant-water fleet (especially those from the east) were concerned with volume, not size. Two effects flowed from this. Initially, in an effort to land larger sized hake, wide-scale discarding of smaller fish took place by South African and other western nation trawlers; adjustments made to catch per unit effort data by Sea Fisheries Research Institute (SFRI) (in conjunction with ICSEAF) suggest that the discard rate prior to 1971 was as high as 39 percent, although Bergh and Barkai (1996. 4) commented that the accuracy of the figure is highly debatable, and have suggested that the true discard rate was probably lower (between 11.2 and 25.1 percent). Irrespective of the precise value of the discard rate, it is widely accepted that a high degree of discarding took place, resulting in severe depletion of the hake resource. In turn, depletion of the hake stock led to the South African fishing fleet becoming marginalised as the bulk of the catch came to be made up of smaller fish, which were targeted by distant-water fleets. Indeed, between 1970 and 1975, foreign fleets accounted for over 50 percent of the nominal hake catch in South African waters (Stuttaford, 1995, 11).

The dramatic collapse that took place in the hake fishery during the 1970s, coupled with the marginalisation of the South African fishing fleet, precipitated a number of regulatory measures which were aimed at reversing the over-exploitation of the hake resource and, at the same time, ensuring that the bulk of the catch reverted to South African fishers. To start with, ICSEAF introduced a system of international inspection and allocations of quotas to each member country participating in the hake fishery; and efforts were made to bolster these controls by declaring other regulations. For example, in 1975, the minimum mesh size was increased from 102 millimetres to 110 millimetres in an effort to reduce catchability by allowing for additional escapement of smaller fish (Punt, 1988, 12; Bergh and Barkai, 1996, 15). Efforts were also made to encourage the withdrawal of foreign fleets. The success of these measures was, however, mixed. For example, the catch taken by foreign fleets fell steadily from the mid-1970s, as did tota' fishing effort (see Figures 4.6 and 4.7). However, the introduction of larger minimum mesh sizes, paradoxically, led to higher levels of resource depletion. The reason for this has been well explained by Bergh and Barkai (1996, 15):

> The reason behind this phenomenon [of higher exploitation in the face of stricter regulations] is well known tc scientists and enforcement and industry personnel who were involved with the fishery at the time. The economic impact of the legislated increase in the codend mesh size [which reduced catchability] could not be absorbed by the industry without seriously threatening its economic viability. It was not possible to continue to use the 102 mm [millimetre] codends since there was probably effective enforcement of the minimum codend mesh size of 110 mm. The only solution for the trawlermen and their skippers, whose earnings were directly linked to their fishing performance, was to make use of so-called liners and panty-hoses. These are small mesh inserts placed inside codends which reduce the intended escapement properties of the legal 110 mm mesh codends.

In short, although it is very difficult to replicate the performance of 102 millimetre codends using 110 millimetre mesh plus liners, the liners served to effectively reduce the intended escapement properties of the legal 110 millimetre mesh. Indeed, Bergh and Barkai (1996, 15) commented: 'It is well known that the only company that attempted to comply with the new regulations went out of business.' Furthermore, Punt (1988, 12) pointed to evidence which shows that the new mesh size restriction had little or no effect in terms of beneficially altering the age composition of the hake catch. More recently, Bergh and Barkai (1996, 16) have suggested that the new restrictions actually reduced the mean age of hake landed by 25-30 percent. Finally, the situation was exacerbated by the fact that whilst legal powers of enforcement existed for South African-registered vessels, this was not the case for foreign vessels; and it is probable that foreign vessels continued to use mesh sizes ranging between 70 millimetres and 110 millimetres (Bergh and Barkai, 1996, 15). In summation, the more stringent restrictions introduced through the early and mid-1970s were of little or no benefit to the hake resource.

However, the late 1970s marked a reversal of fortunes for the hake resource. In November 1977, the South African authorities declared a 200-nautical-mile EEZ which facilitated the virtual removal of foreign fishing effort (Stuttaford, 1995, 11). At the same time, enforcement against the use of illegal equipment, such as the smaller codend mesh size used by foreign fishers and the liners used by domestic fishers, became more effective (Bergh and Barkai, 1996, 16). Furthermore, in the same year as the South African authorities declared the EEZ, ICSEAF agreed to an initial total allowable catch of 165 000 tons, with 110 000 tons reserved for South African fishers (SADSTIA, 1994, 19). However, as noted by SADSTIA (1994, 19), 1977 'proved to be a bad year with no indication that foreign effort was actually being reduced. The domestic trawling industry was plainly on its knees.' Thus, 1978 witnessed the setting of a pilot domestic total allowable catch (of 125 000 tons) for the hake fishery, shared by two major and four smaller companies (Access Rights and Resource Implications Task Group, 1995, 58).[9]

It was soon realised, however, that a 'global' quota was ineffective, and that a shift to 'specific' quotas would be necessary to prevent over-fishing. One result of this was that investment in the fishery crept up 'as participants competed to improve their positions in anticipation of this final [sic] step in the evolution of rights' (SADSTIA, 1994, 19). In confirmation of these suspicions, company quotas were introduced in the deep-sea trawl fishery in 1979; and the process of allocating individual rights in the hake fisheries was completed in 1982 with the allocation of individual quotas to the inshore trawlers located along the south coast.

These regulations facilitated a recovery in the hake resource, evidenced by a recovery in catch rates from the lows recorded around the mid-1970s (Department of Environmental Affairs and Tourism, 1997, 4). In the west

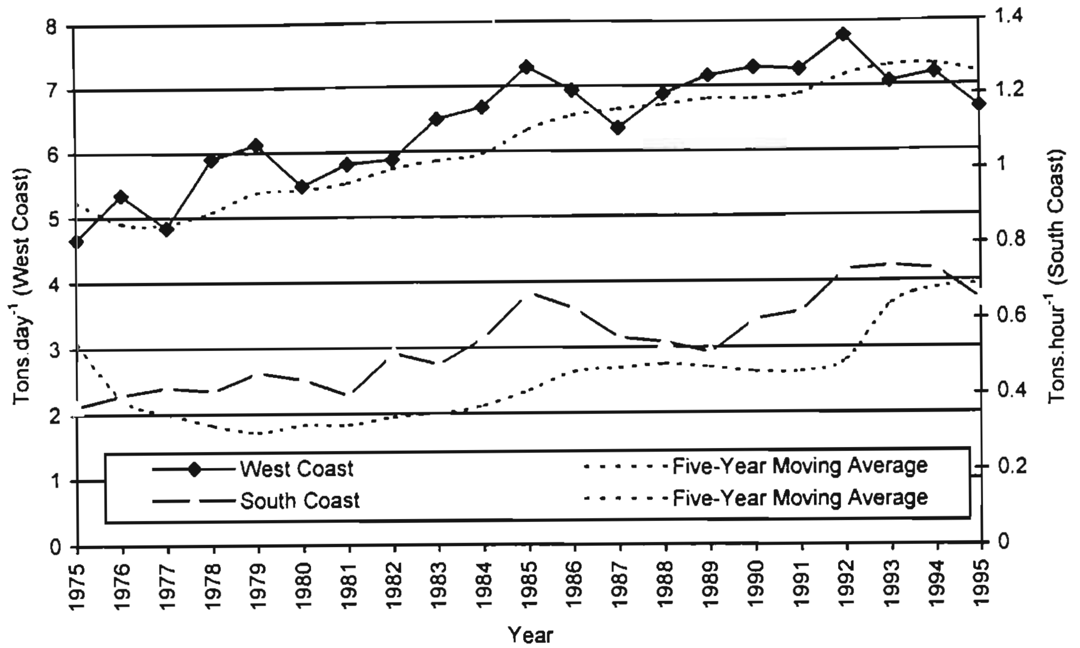
[9] An allocation of 5 000 tons was awarded to South East Coast Fishing Industry Association (SECIFA) members; and, in terms of bilateral agreements, Japan and Israel were granted hake quotas of 5 000 tons and 1 500 tons respectively from the tonnage available (Access Rights and Resource Implications Task Group, 1995, 58).

coast fishery, the average catch rate increased by just over 20 percent between the introduction of regulations in the late 1970s and the end of the 1980s. This trend continued into the 1990s, with catch per unit effort growing by almost 10 percent between 1989 and 1992 (Japp, 1996). Since then, catch rates have fallen back to levels observed in the mid-1980s (Japp, 1996). However, Japp considers this most recent fall in the catch rate to be within the expected range of 'natural fluctuations' in catchability, and argues that the fall in the catch rate is not necessarily indicative of 'resource decline'.[10] Similar evidence exists for the south coast fishery. Between the end of the 1970s and the end of the 1980s, catch rates climbed by nearly 25 percent, and increased by a further 43 percent over the first half of the 1990s. However, much like the west coast fishery, the period 1994-95 saw the catch rate fall back to levels observed in the mid-1980s. Again, Japp considers this to be within the parameters of natural fluctuations and not necessarily due to over-fishing.[11]

Figure 4.5 provides further evidence of the recovery in the stock. Specifically, the figure shows that catch per unit effort in the west and south coast fisheries has stayed above the five-year moving-average virtually continuously since the introduction of regulations in the late 1970s. However, the fall in catch per unit effort below the five-year moving-average in both fisheries in the mid-1990s, and the resultant reversal of the upward trend of the moving average in the west coast fishery, are indicative of a break in the recovery of stocks. As already noted, this reversal is thought to be temporary and within the boundaries of natural fluctuations.

[10] Pers. comm. (Japp, Sea Fisheries Research Institute).
[11] Pers. comm. (Japp, Sea Fisheries Research Institute).

Figure 4.5: Catch per Unit Effort in the West and South Coast Hake Fisheries (1975-95)

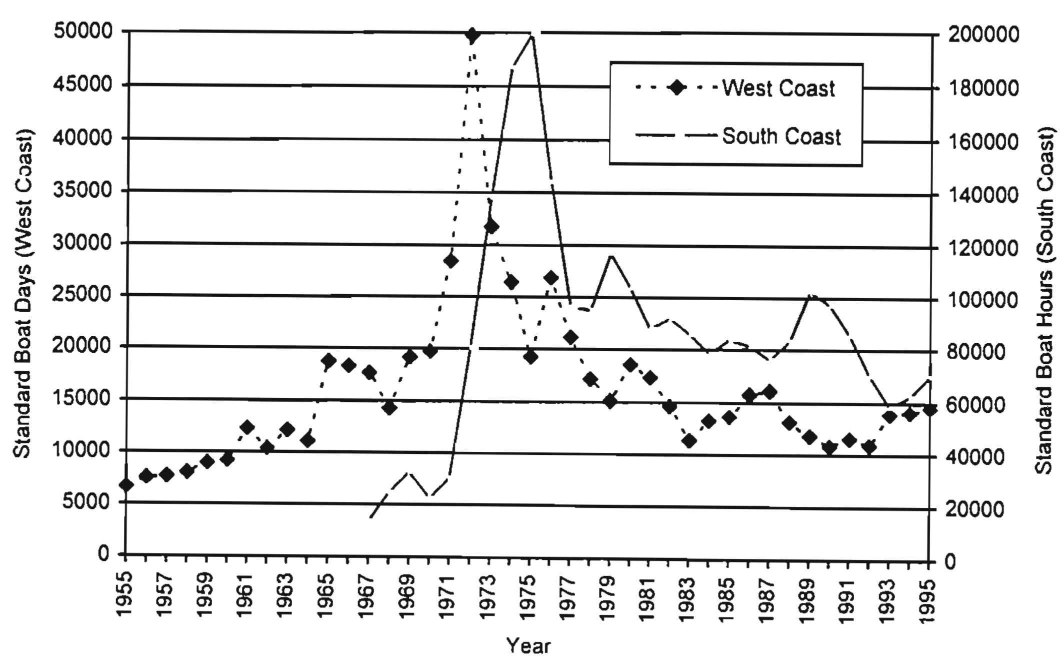


Source: Adapted from Payne and Punt (1995); Japp (1996).

However, while catch rates have been in an upward trend since the introduction of regulations in the late 1970s and early 1980s, indicative of a recovery in hake stocks, three caveats should be noted here. First, catch per unit effort is the ratio of catch to unit effort. Thus, rising catch per unit effort figures may be as much a result of higher catches as they are of falling effort. In other words, higher catch per unit effort figures are not necessarily indicative of recovery in fish stocks. Rather, it is necessary to establish the extent to which lower effort is responsible for a higher ratio of catch to effort. In this regard, whilst catches in the South African hake fisheries have been in a broadly upward trend, increasing by about 25 percent from the lows recorded in the early 1980s (see Figure 4.5), effort has been in a downtrend, falling by 15 percent in the west coast fishery and 25 percent in the south coast fisheries, since the introduction of regulations in the late 1970s (see Figure 4.6). Thus, it is declining effort coupled with increasing catch – rather than just an increasing catch resulting from a recovery in the biomass – that has translated into rising catch per unit effort figures. From this, it follows that the recovery in the hake biomass has been somewhat weaker than that suggested above on the basis of rising catch per unit effort figures.

Second, although effort levels have fallen, and catches have risen, catch rates still stand well below the levels recorded in the 1950s and 1960s (see Figure 4.4). Third, more recently, concern has been expressed about the accuracy of catch per unit effort figures in the hake fishery (Geromont and Butterworth, 1995; Bergh and Barkai, 1996; Japp, 1996). This point is returned to in Chapter 5 where it is discussed in detail. Notwithstanding these caveats, it can be concluded that whilst the recovery in the hake fishery has not been spectacular, there has been a modest recovery; and this, coupled with the ejection of foreign fleets from the hake fishery, has served to bolster the importance of the hake fishery to the South African economy.

Figure 4.6: Effort Levels in the West and South Coast Hake Fisheries (1955-95)



Source: Adapted from Japp (1996, 14).

While regulations have facilitated a recovery in the hake stocks, and have reduced the level of effort, they have also served to mould certain features in the hake fishery. The most important of these features is the concentration of the catch in the hands of a few participants. Specifically, when company quotas were introduced in the deep-sea fishery in 1979, 95 percent of the total allowable catch for hake was allocated to three large companies: the so-called 'foundation' companies, which include Irvin and Johnson Limited (which has participated in the trawl fishery since the turn of the century), Amalgamated Fisheries Limited (now Atlantic Trawling Limited) and Sea Harvest Corporation Limited (the latter two companies entered the trawl fishery through the course of the 1960s). Atlantic Trawling Limited falls under the ownership of Sea Harvest Limited, and therefore, 95 percent of the catch fell into the hands of two companies. The balance of the initial quota was allocated to three newly established smaller enterprises (SADSTIA, 1994, 17-19). Since then, a number of new (small-scale) fishers have entered the hake fishery. Nevertheless, the structure of the industry has essentially remained unaltered, with the bulk of the total allowable catch concentrated in the hands of a few large companies (see Section 4.3.2 below).

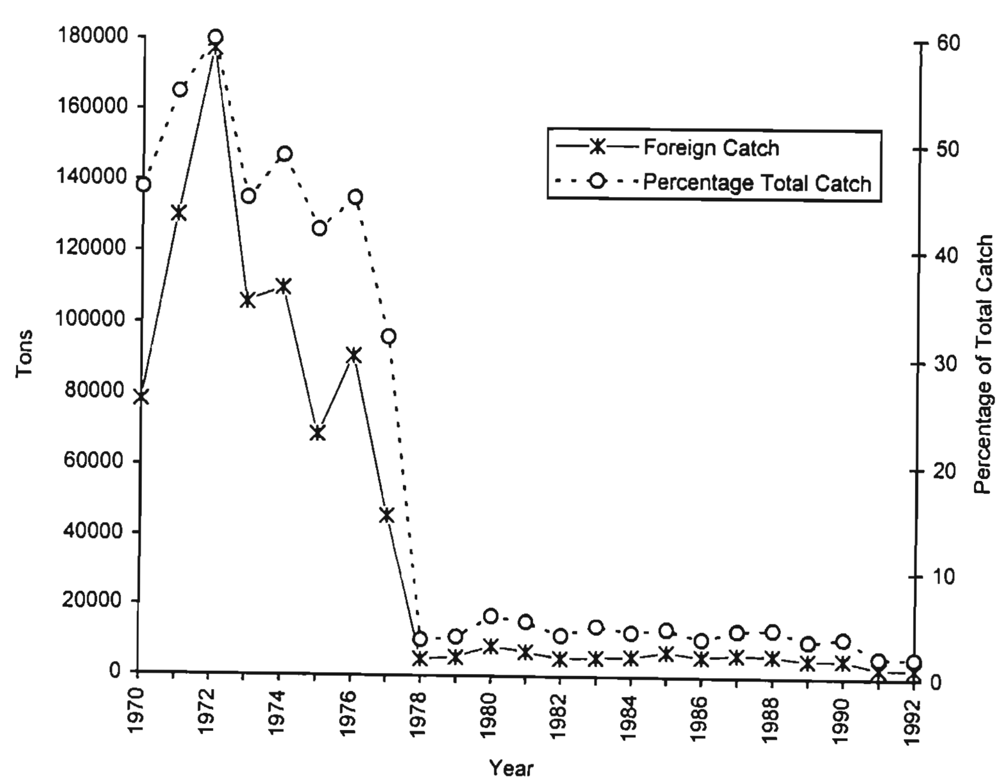
In summary, regulations introduced in the hake fishery in the late 1970s and early 1980s facilitated a modest recovery in the hake stock. The recovery has manifested itself in the form of higher catch rates as well as small increases in total catch. In addition, regulations also served to eject foreign fleets from the fishery. These two effects, in turn, have ensured that the hake fishery continues to constitute the single most substantial component of South Africa's fishing industry. However, the regulations that were put in place to reverse over-exploitation and to eject foreign fleets from the fishery have also had adverse effects on the industry, the most obvious of which is an excessive concentration of ownership and control in the fishery. Furthermore, in spite of efforts on the part of fisheries authorities to introduce new participants into the hake fishery, the high degree of concentration of power remains largely undiluted. This has had a major impact on the structure and characteristics of the present-day industry. Section 4.3.2 explores the major economic features of the contemporary hake fishery.

4.3.2 THE PRESENT-DAY HAKE FISHERY

Between 1985 and 1993, the total catch of the hake fishery averaged approximately 140 000 tons per annum, accounting for about 25 percent of South Africa's total commercial catch by mass (Stuttaford, 1995, 20-21). In terms of commercial value, the hake fishery is by far the most important fishery. In 1993, the estimated value of total landings and products of South African commercial fisheries was of the order of R1.2 billion, of which the wholesale (processed) value for the hake fisheries was in the region of R400 million (Stuttaford, 1995, 11; Van D. Boonstra, 1995, 5). In line with these figures, hakes typically account for approximately one-third of the total commercial catch by value. In contrast, the pelagic species collectively account for around one-quarter of the commercial catch by value (Van D. Boonstra, 1995, 5). In addition, since the declaration of the EEZ in 1977, the domestic fleet has accounted for virtually the entire catch. According to Figure 4.7, foreign fleets accounted for as much as 60 percent of the total hake catch in South African waters in the early 1970s, but the figure fell dramatically from the mid-1970s onwards. Foreign catches – taken primarily by Israeli, Japanese, Portuguese, Spanish and Taiwanese fishers (Stuttaford, 1994, 27) – averaged less than 4 percent of the total hake catch between 1978 and 1992. From 1993 onwards, the only foreign quota in the demersal fishery has been an annual quota of 1 000 tons of hake – equivalent to less than 1 percent of the total annual quota – awarded to a joint venture between South Africa and Mozambique in terms of a bilateral fishing agreement (Department of Environmental Affairs and Tourism, 1997, 4).

As noted earlier, the hake fishery is divided geographically between the west coast and south coast, with the annual catch typically broken down in the ratio 2:1 in favour of the west coast fishery (Japp, 1994,1). However, there are a number of further important features of the breakdown of the catch which deserve particular mention. First, as noted in Section 4.2, the catch is broken down between the deep-sea and inshore fisheries. The deep-sea catch is typically in the region of ten times greater than the inshore catch, making the deep-sea fishery by far the more important component. It should be noted, however, that hake is also taken in the midwater trawl fishery; although the size of the hake catch in the midwater trawl fishery is small (amounting to less than 0.2 percent of the total hake catch) as the bulk of effort in the midwater trawl fishery is directed at Cape horse mackerel (Japp, 1996). There is also a small (but growing) number of fishers involved in the hake-directed longline and handline fisheries. Figure 4.10 shows the breakdown of the hake catch between the various fisheries. From this it is apparent that trawling in the deep-sea and inshore fisheries accounts for virtually the entire hake catch. For this reason, this study focuses on the deep-sea and inshore trawl fisheries for hake.[12]

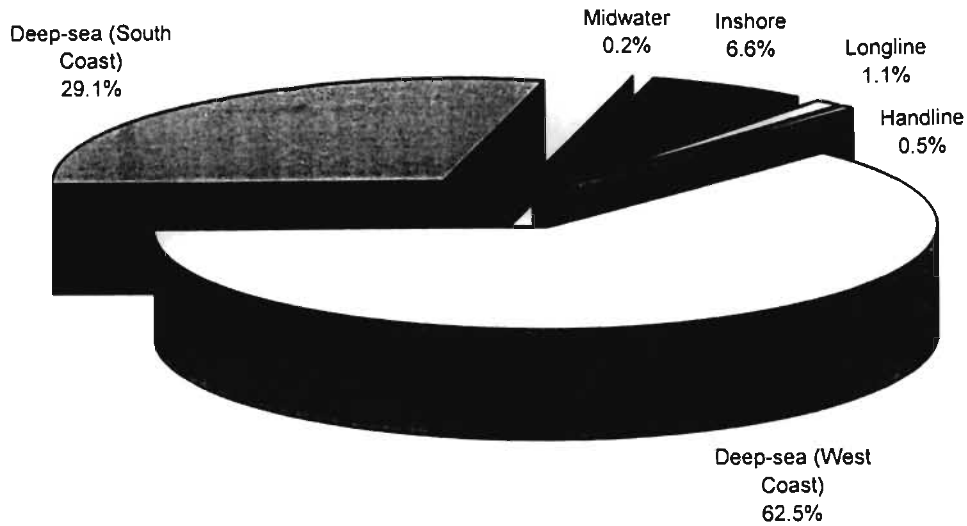
Figure 4.7: Foreign Catches of Hake in South African Waters



Source: Adapted from Punt (1994, 162); Stuttaford (1994, 27).

[12] This point is returned to in Chapters 5 and 7.

Figure 4.8: Breakdown of Hake Catch by Fishery (1995)

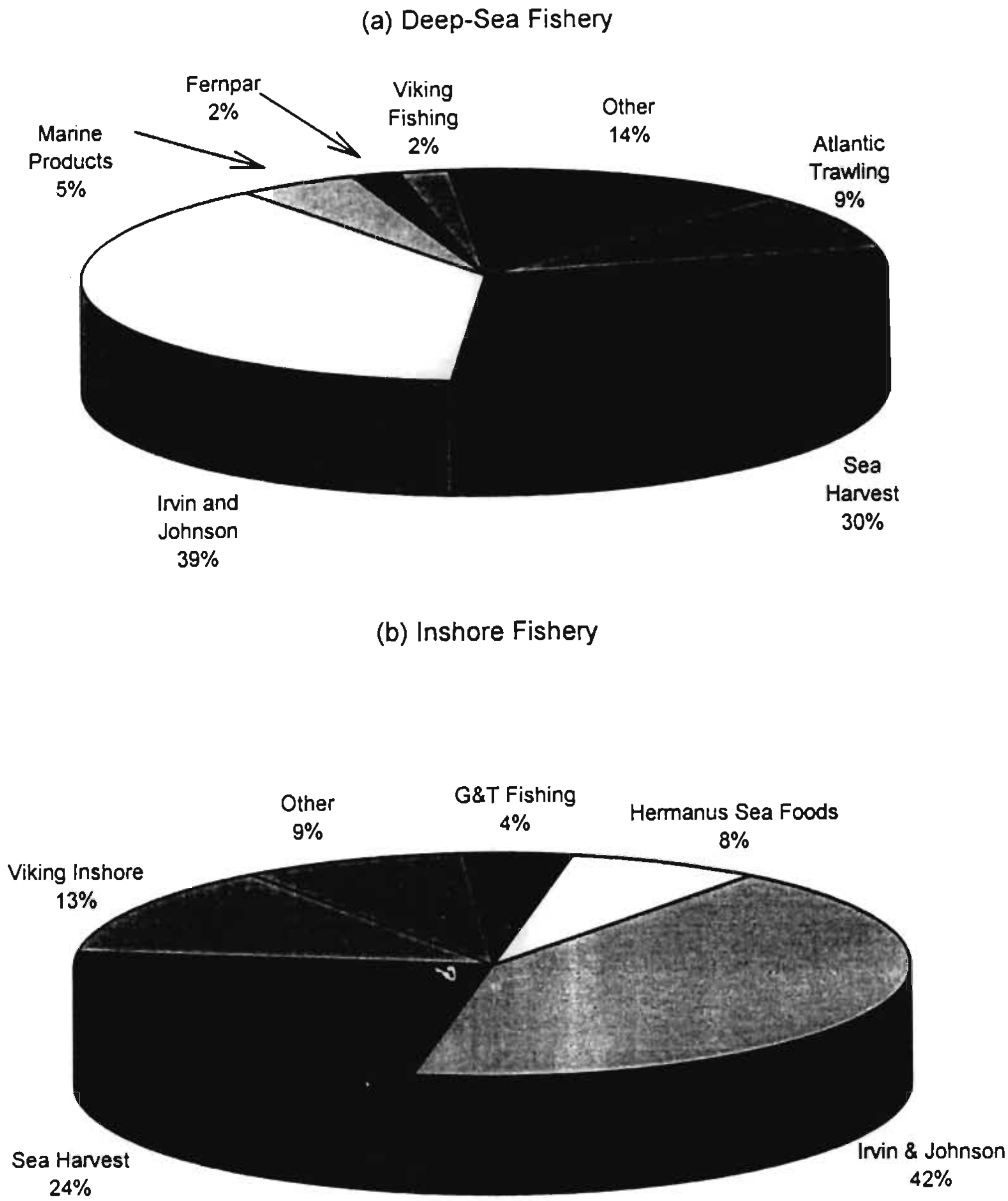


Source: Adapted from Japp (1996).

A second important feature of the present-day hake fishery is that two large companies continue to control the bulk of the total allowable hake catch (Japp, 1994, 1). The two companies, Irvin and Johnson Limited and Sea Harvest Limited (and its subsidiary Atlantic Trawl Limited), typically account for approximately 80 percent of the total allowable catch in the deep-sea hake fishery and about two-thirds of the catch in the inshore hake fishery. Figure 4.9 provides a breakdown of quotas between major participants in the deep-sea and inshore fisheries respectively.[13] Based on this data, it is apparent that control over resources in both the deep-sea and inshore fisheries remains highly concentrated in the hands of two players.

[13] The category 'Other' incorporates a wide array of fishers, including the Independent Fish Processors Association, longliners, community trusts and a number of small participants (Stuttaford, 1995, 12).

Figure 4.9: South African Hake Quotas (1995)



Source: Adapted from Stuttaford (1995, 12-13).

Accordingly, the Herfindahl-Hirschman Index (HHI), a measure of industry concentration, is 3041.6 for the deep-sea fishery and 2661.3 for the inshore fishery.[14] However, numerous transfers of quotas take place within

[14] The HHI is calculated by summing the squares of the individual market shares of all the firms in a given market. By incorporating all the firms in the market, the HHI accounts for both the distribution of the market shares of the top firms and the composition of the market outside of these firms; and so the HHI is arguably a far better measure of industry concentration than, say, the *n*-firm concentration ratio which only factors the largest *n*-firms

companies (across fisheries) and between companies. For example, the 'paper' quotas allocated to coastal communities (the so-called Community Trusts) are generally fished by larger companies. In 1995, for example, the entire issued Community Trusts quota (just over 1 600 tons) was fished by six 'large' companies. This served to increase the market share of larger players and dilute the share of smaller players. Thus, it is likely that concentration ratios are even higher than those quoted above. Inter-company and intra-company transfers also take place: for example, between inshore and deep-sea operators where, again, larger companies tend to gain market share (Japp, 1996, 6-10), further increasing concentration ratios. In short, there can be little doubt that the structure of both fisheries could be described as oligopolistic (or even duopolistic in the case of the deep-sea fishery).

With regard to the structure of the contemporary fleet, the deep-sea fishery consists of about 34 wetfish vessels (ice-carriers) and 50 factory-freezers, all of which are stern trawlers (Japp, 1995, 9).[15] Another 35 small trawlers (with an average length of 23 meters) operate inshore on the south coast (Payne and Punt, 1995, 20). These vessels are estimated to have a combined tonnage in excess of 50 000 registered tons valued at over R400 million in 1993 (SADSTIA, 1994, 6). It should also be noted that the industry is a significant employer of labour. The deep-sea sector alone employs 8 600 people, of which about 2 850 people are sea-faring (SADSTIA, 1994, 6). Employment is permanent and non-seasonal. Total labour costs (including remuneration, fishing commissions, bonuses and company contributions to employee benefit schemes) amount to about R250 million per annum (SADSTIA, 1994, 6). Total investment in productive assets, including ships, plant and machinery, buildings and vehicles, adds up to nearly R700 million (SADSTIA, 1994, 6). There are 57 land-based factories involved with the

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(where *n* is usually three, four or ten) into calculations of industry concentration. The highest possible HHI is 10 000 when one firm has 100 percent of the market; the lowest HHI is near zero in the case of a perfectly competitive market. An HHI of greater than 1800 is typically treated as indicative of a high degree of concentration (Shepherd, 1997, 74).

[15] Substantially less than the more than 300 trawlers that were in operation at one stage during the 1970s.

trawl fishery (Bergh and Barkai, 1993, 96). The hake fishery also has substantial spill-over effects. For example, it is estimated that the purchases of goods and services by the deep-sea sector from domestic suppliers amounts to well over R300 million per annum (1993 values) (SADSTIA, 1994, 6).

About two-thirds of the demersal catch is landed fresh and processed in extensive shore-based facilities. The balance of the catch is processed at sea into marketable products aboard factory ships. The trawling industry supplies the majority of the fresh and frozen seafood consumed by South Africans, and is a supplier of raw material to an extensive range of fish processors. The bulk of the demersal catch is meant for direct human consumption, although a small portion – around 1 percent of the total catch by mass – is processed as fish meal at sea, and a similar amount is made into fish meal from offal and damaged fish at land-based facilities (Van D. Boonstra, 1995, 43). The industry has also developed an extensive international market, with an accent on high value-added products. About one-third of the fresh or frozen demersal catch is sold on the export market; and hakes typically account for between one-quarter and one-third of the fishing industry's annual export earnings (which were in excess of R800 million in 1994) (Monthly Abstract of Trade Statistics, 1994).

It is evident, therefore, that the present-day hake fishery is of considerable socio-economic importance. The fishery is the backbone of South Africa's large fishing industry, and is a substantial investor and creator of jobs, as well as an important source of income and foreign exchange.[16] For these reasons, it is vital that the resource be managed on an appropriate basis. At present, the fishery is governed by a number of regulations, and these are outlined in Section 4.4 below.

[16] Apart from the socio-economic importance of the resource, hake have a significant function in the south-east Atlantic ecosystem (Payne, 1989; Payne and Punt, 1995; Pitcher and Alheit, 1995).

4.4 Regulation of the Present-Day Hake Fishery

A number of regulations are in place in the present-day hake fishery which are intended to protect the stock from over-exploitation as well as reverse the effects of past over-exploitation. Deep-sea vessels are restricted to nets of 110 millimetres stretched mesh on the west coast, but nets of 75 millimetres mesh are permitted east of 20^0E (these are areas located on the south and east coast, but are off the main fishing banks). Recently (1991), however, the offshore fleet has been permitted to use 75 millimetre mesh (bottom trawls) in the south coast fishery (Japp *et al*, 1994, 125).[17] Vessels in the hake fishery have no limit to operating depth on the west coast, but bottom trawling is not allowed within five nautical miles of the coast from Cape Point northwards (Field and Glazewski, 1992, 320). On the south coast, from 20^0E to 27^0E, deep-sea trawlers are not permitted to operate shallower than the 110 meter contour, although inshore trawlers which operate on the Agulhas Bank may fish outside of the 110 meter contour (Japp, 1995, 9). There is no closed season for the deep-sea trawl fishery, but certain inshore areas are closed during the squid spawning season to protect spawning grounds on the south coast (Field and Glazewski, 1992, 320). Certain bays, including False Bay, Plettenberg Bay, St Sebastian Bay and St Francis Bay, are permanently closed to trawlers.

Whilst operators in the hake fishery require licences, in the first instance, the level of catches is controlled by the allocation of annual quotas to companies and individuals; a complete discussion of the current process by which total allowable catches are set for South Africa's fisheries is undertaken in Butterworth *et al* (1992). Currently, hake quotas are allocated on the basis of an $f_{0.2}$ strategy, as per Andrew and Butterworth (1987). The implication of this strategy is that the total allowable catch is set equal to approximately 18 percent of the biomass at the beginning of the year (Getz and Haight, 1989,

[17] Recently, it has been claimed that some fishers may still use small mesh 'liners', thereby enabling (illegal) larger catches of small fish (Field and Glazewski, 1992, 320).

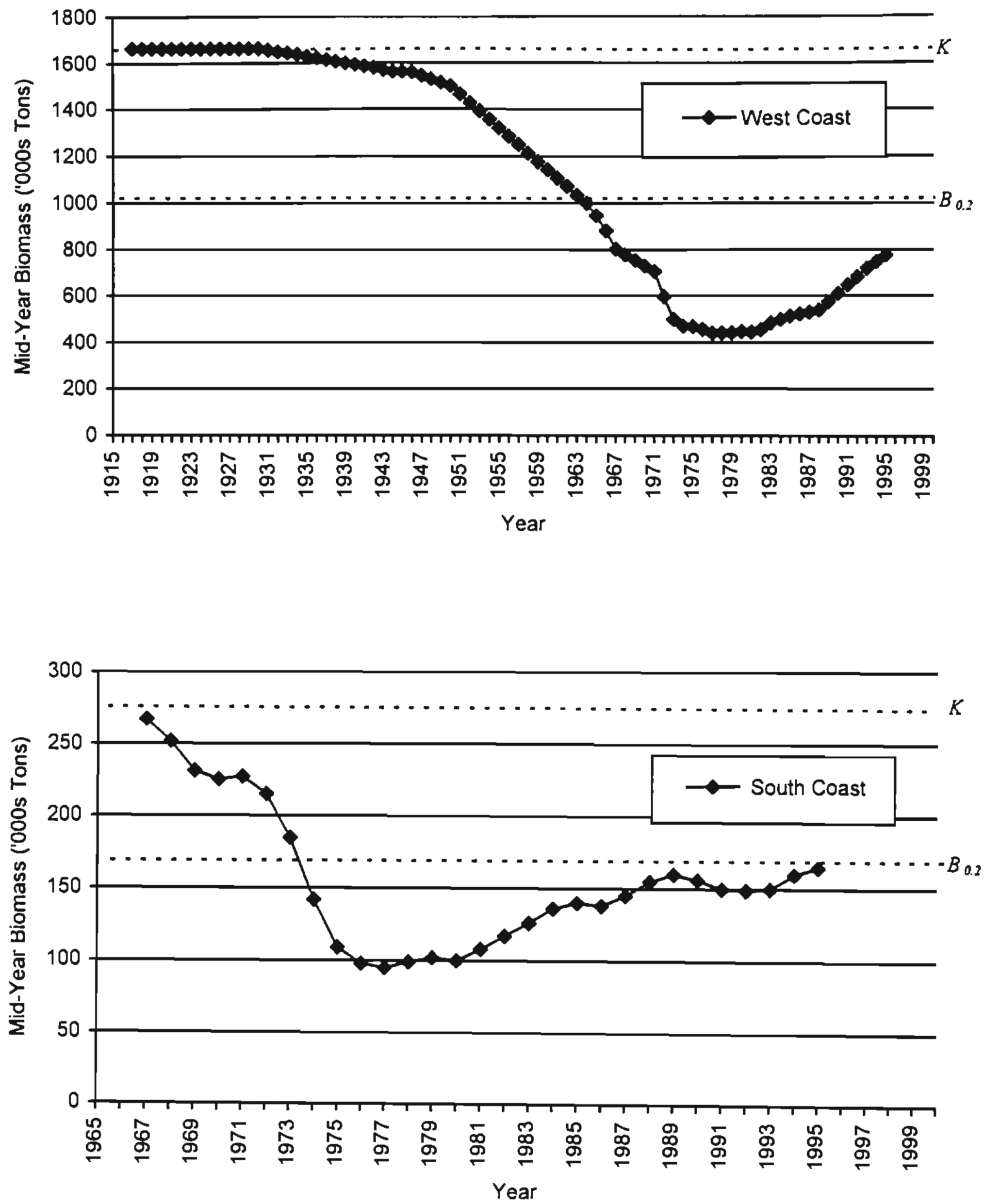
150). Accordingly, the $f_{0.2}$ strategy aims to exploit the hake resource at a rate which would allow the biomass to recover to 60 percent of its pristine level: equal to approximately 1 000 000 tons on the west coast and 165 000 tons on the south coast (Punt, 1994, 161). The total allowable catch associated with this strategy is presently in the region of 145 000 tons per annum (1993), which includes an annual allocation of 7 000 tons to new entrants (Bergh and Barkai, 1993, 95). However, it has been estimated elsewhere (Bergh and Barkai, 1993, 95) that the resource could withstand harvesting of up to 185 000 tons per annum, which implies that the industry is currently sacrificing an additional catch of 40 000 tons per annum. It is argued that this potential sustainable yield is not being harvested in an effort to increase the hake biomass as part of a long-term strategy which will ultimately generate higher sustainable yields; and compliance by the industry was achieved by coupling the $f_{0.2}$ strategy with an undertaking that the original quota holders would receive 80 percent of any increases in the total allowable catch. However, whilst the '80 percent arrangement' has since been abandoned as politically unacceptable, in that it perpetuates the high levels of concentration in the fishery (Bergh and Barkai, 1996, 7), the $f_{0.2}$ strategy has been relatively successful in rebuilding the hake resource, having produced an increase in the biomass of between 0.1 and 1.5 percent per annum since 1978 (Bergh and Barkai, 1996, 7).

It should be noted that, because of the use of inaccurate data in the modelling procedure, the growth in the biomass was initially thought to be of the order of 2.5 percent, which translated into similar increases in the size of the annual total allowable catch. According to Bergh and Barkai (1996, 7), this has meant that the total allowable catch has probably been in the region of 20 000 tons greater than it should have been. Some writers have gone on to suggest that this 'over-harvesting' may be the reason for the less than robust recovery (particularly on the west coast) in hake stocks (Geromont and Butterworth, 1995); although other writers have suggested that reasons for the poor recovery in biomass may also include a declining trend in the catch coefficient; transient age-structure effects (such as 'bad' year classes passing through the fishery); model misspecification; poor data (for instance, the high catch per unit effort data for the 1960s may be unrealistic because of, for example, incorrect allowances for discards); and the recent move to allow fishers to 'carry-over' unused quotas (Punt, 1992; Japp, 1994; Geromont and Butterworth, 1995).

Figure 4.10 shows the carrying capacity of the respective fisheries (that is, the pristine or unfished biomass) and the estimated biomass up to 1995. The rapid decline in biomass after 1950 in the west coast fishery, and after 1970 in the south coast fishery, also provide further evidence of the over-exploited states of the stocks under consideration. Similarly, the recovery in the biomass in both fisheries after the introduction of regulations in the late 1970s, provides support for the argument that regulations have served to at least partially reverse the level of exploitation of the hake stocks.

Nevertheless, the current basis for fisheries management is inappropriate. More to the point, as argued in Chapter 3, regardless of the degree of success enjoyed by fisheries managers in reversing the over-exploited state of the hake resource – which, at best, can only be described as modest – the strategy has relied almost entirely on biological principles, and largely ignores economic principles. In this regard, Payne and Punt (1995, 32-3) comment that: 'modelling [of the hake resource] has been kept very simple – to date ... economic effects have not been taken into account directly when providing management advice for species'. In short, South Africa's fisheries managers have yet to incorporate economic principles into the management of fisheries. This is the task of Chapters 5, 6 and 7.

Figure 4.10: Estimates of the Mid-Year Biomass Series for Cape Hakes off the West and South Coasts (1915-95)



Source: Leslie (1996).

CHAPTER 5: DYNAMIC BIOECONOMIC MODELLING OF SOUTH AFRICA'S HAKE RESOURCE

5.1 Introduction

The aim of this chapter is to apply the dynamic bioeconomic model developed in Chapter 3 to South Africa's hake fishery as described in Chapter 4. In order to do so, it is necessary to reduce the dynamic bioeconomic model to a form in which it is easily estimated, and Section 5.2 is concerned with this task. Furthermore, it is necessary to provide estimates of the various biological and economic parameters of the bioeconomic model, and this task is undertaken in Section 5.3. Section 5.4 is devoted to applying the dynamic bioeconomic model to the hake fishery as well as providing a detailed examination of the modelled results, while Section 5.5 is concerned with establishing the implications for economic rent levels of adopting bioeconomic principles, as opposed to biological principles, as the basis for regulating the fishery. Section 5.6 provides a summary of the implications of shifting the west and south coast hake fisheries from dynamic maximum sustainable yield positions to dynamic rent maximising positions and suggests a number of extensions to the modelling process.

5.2 Estimatable Form of the Dynamic Bioeconomic Model

In order to apply the dynamic bioeconomic model, as developed in Chapter 3, to South Africa's hake fisheries, it is first necessary to reduce the model to a form in which it is easily estimated. To begin this process, it was concluded in Chapter 3 that the optimal (that is, dynamic rent maximising) size of a fish stock (X^*_{MEY}) is determined by the fundamental equation:

$$G'(X^*_{MEY}) - \frac{c'(X^*_{MEY})G(X^*_{MEY})}{p - c(X^*_{MEY})} = \delta . \qquad \text{(Equation 3.18)}$$

In the event that the surplus production function $[G(X_t)]$ is the Schaefer logistic function, Equation 3.18 reduces to a quadratic equation, from which the following closed formula for X^*_{MEY} is readily obtained:

$$X^*_{MEY} = \frac{1}{4}\left\{ \bar{X}_t + K(1-\frac{\delta}{r}) + \sqrt{\left[\left(\bar{X}_t + K\left(1-\frac{\delta}{r}\right)\right)^2 + \frac{8K\bar{X}_t\delta}{r} \right]} \right\} \qquad \text{(Equation 5.1)}$$

where, as before, $\bar{X}_t = \frac{c}{pq}$, where c is the cost of harvesting (per unit of catcher effort), p is the price per unit of fish and q is the catchability coefficient.

It should be emphasised that while the model developed here is based on the Schaefer population growth function – as per Gordon (1954) and others (Clark, 1976 & 1985) – the model is easily adapted to different specifications of the growth function, such as those discussed by Pella and Tomlinson (1969), Fox (1970) and Shepherd (1987).[1] Nevertheless, in line with arguments presented elsewhere (Punt, 1988a, 1988b, 1989a, 1989b & 1991; Punt *et al*, 1992 & 1994), it is assumed that the Schaefer function constitutes the 'best' choice of function in the case of the South African hake resource. This issue is elaborated upon in Section 5.3.1 below.

The operation of the reduced form of the model is usefully illustrated by way of two special cases: zero cost and zero discount rate scenarios. First, if the discount rate is set equal to zero ($\delta = 0$), it can be shown that:

$$X^*_{MEY} = \frac{1}{2}\left(\bar{X}_t + K \right) = X_{MEY} . \qquad \text{(Equation 5.2)}$$

[1] See Andrew and May (1986) and Geromont and Butterworth (1995) for examples of applications of these alternative specifications of the biological growth function.

That is to say, if the discount rate is assumed to be zero (meaning that an equal weighting is attached to all future periods), then the dynamic rent maximising biomass (X^*_{MEY}) will be the same as the static rent maximising biomass (X_{MEY}). This outcome is shown in Figure 4.13 (Chapter 4) where, under the assumption of a zero discount rate, the dynamic rent maximising outcome is given by $X^*_{MEY} = X_{MEY}$ and, as the discount rate rises, so the dynamic rent maximising biomass falls such that $X^*_{MEY} < X_{MEY}$. In the extreme, $X^*_{MEY} \to \overline{X}_t$ as $\delta \to +\infty$, that is, the dynamic rent maximising biomass approaches the static zero-rent biomass as the discount rate approaches infinity.

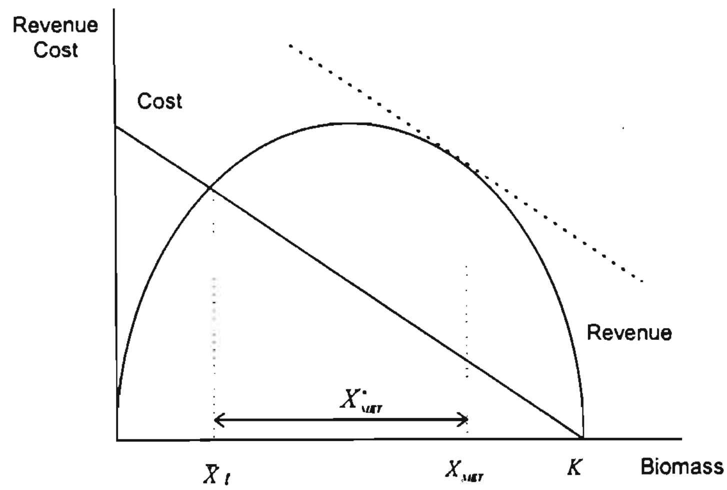
Second, if costs are set equal to zero, that is, fishing is assumed to be a costless activity, then it can be shown that:

$$X^*_{MEY} = \frac{K}{2} = \left(1 - \frac{\delta}{r}\right) = X_{MSY}. \qquad \text{(Equation 5.3)}$$

In other words, if it is assumed that the costs of fishing are zero, then irrespective of the discount rate, the rent maximising biomass will be the static maximum sustainable yield biomass ($X^*_{MEY} = X_{MSY}$).[2] In short, assuming a constant discount rate, say $\delta = 0$, the dynamic rent maximising biomass (X^*_{MEY}) falls with costs. Similarly, assuming constant costs, the dynamic rent maximising biomass (X^*_{MEY}) falls as the discount rate increases. Figure 5.1 provides a summary of the above results, where the dynamic rent maximising biomass (X^*_{MEY}) is shown to lie between the static zero-rent biomass ($\overline{X}_t$) and the static maximum economic yield biomass (X_{MEY}).

[2] In the instance of positive fixed costs, the result remains intact as long as the marginal cost of harvesting is zero and total fixed costs are less than the maximum sustainable revenue.

Figure 5.1: The Optimal Biomass in the Dynamic Bioeconomic Model (Schaefer Growth Function)



Source: Clark (1985, 23).

Having reduced the model to a form in which it is easily estimated, and having noted the central features of the model's mechanics, the discussion turns to establishing values for the various biological and economic parameters (r, K, q, δ, c and p), as set out in Equation 5.1, required in estimation of the bioeconomic model.

5.3 Parameter Estimation

The concern of this section is with establishing accurate and reliable estimates of the values to be taken by the biological and economic parameters (r, K, q, δ, c and p) in the case of South Africa's hake fisheries. For convenience, the parameters are divided into two subsets, namely (i) biological parameters, which include r, K and q; and (ii) economic parameters, which include δ, c and p.

5.3.1 BIOLOGICAL PARAMETERS: r, K AND q

Estimates of biological parameters in demersal fisheries – which, in the present context, include the intrinsic growth rates, environmental carrying capacities and the catchability coefficients of the west and south coast hake fisheries – are typically generated by applying surplus production modelling to data on commercial catch rates (catch per unit effort) and gross landings (Bergh and Barkai, 1996, 5). These estimates are then generally refined (or 'fine tuned') with the aid of supplementary analyses based on the age structure of the catch and independent scientific surveys using, for example, a swept (sampling) method to obtain an index of resource characteristics. Modelling of the South African hake resource involves similar procedures. Specifically, scientists, located mainly at the Sea Fisheries Research Institute (SFRI) in Cape Town, employ a surplus production model, based on the Schaefer growth function, tuned with age-structure analyses and sample surveys, to estimate the intrinsic growth rate, the current biomass and the catchability coefficient of the hake stocks (Punt 1988b, 1992a & 1994; Andrew and Butterworth, 1994).

Table 5.1 summarises the estimates provided by the Sea Fisheries Research Institute, for the period 1984-95, of the three biological parameters r, K and q for the west coast (r_w, K_w and q_w) and south coast (r_s, K_s and q_s) hake stocks. While a full interpretation of the parameter estimates is provided below, it is first necessary to note two qualifications pertaining to the data set.

Table 5.1: Estimates Generated by the Schaefer Form of the Dynamic Surplus Production Model for the West and South Coast Hake Stocks for the Parameters r, K and q

	West Coast				South Coast		
	r_w	K_w	$q_w \times 10^{-3}$		r_s	K_s	$q_s \times 10^{-3}$
1984	0.4643	1 226 000	0.0166	1984	0.7275	288 700	0.0039
1985	0.4480	1 262 000	0.0161	1985	0.7834	273 800	0.0042
1986	0.4806	1 192 000	0.0171	1986	0.8003	269 600	0.0043
1987	0.4085	1 357 000	0.0150	1987	0.7617	279 600	0.0041
1988	0.3871	1 415 000	0.0144	1988	0.7248	290 100	0.0039
1989	0.4383	1 282 000	0.0160	1989	0.6850	302 500	0.0037
1990	0.4061	1 362 000	0.0150	1990	0.6852	302 400	0.0037
1991	0.3557	1 510 000	0.0134	1991	0.6955	299 100	0.0037
1992	0.3417	1 557 000	0.0129	1992	0.7289	288 700	0.0039
1993	0.3269	1 610 000	0.0125	1993	0.7578	280 100	0.0041
1994	0.3191	1 639 000	0.0122	1994	0.7724	276 000	0.0042
1995	0.3092	1 678 000	0.0119	1995	0.7630	278 700	0.0041
Mean	0.3905	1 424 167	0.0144		0.7405	285 775	0.0040
Minimum	0.3092	1 192 000	0.0119		0.6850	269 600	0.0037
Maximum	0.4806	1 678 000	0.0171		0.8003	302 500	0.0043
Standard Deviation	0.0596	170 099	0.0018		0.0386	11 243	0.0002
Coefficient of Variation (%)	15.26	11.94	12.50		5.21	3.93	5.34

Source: Adapted from pers. comm. (Leslie, Sea Fisheries Research Institute).

First, only the estimates for the west coast stock have been tuned (using survey data); the south coast estimates are fitted for catch per unit effort data only. The reason for this is that the survey series for the south coast is too short to allow for tuning; it will only be possible to tune the south coast series using sample survey data from 1997 onwards.[3] Nevertheless, the tuning of parameter estimates using survey data has little real impact on parameter values: estimates of r_w, K_w and q_w, based only on catch per unit effort data differ by less than 10 percent from the tuned estimates over the period 1984-95; and the rK values on the tuned and raw parameter estimates differ, on average, by a modest 1.5 percent on the west coast over the period under consideration.[4]

A second point to note is that parameter estimates are only given from 1984 onwards. There are two reasons for this. Prior to the declaration of the EEZ in November 1977, the hake fishery was dominated by foreign fishers. Thereafter, however, harvesting hake became an almost exclusively South African exercise. Recalling the earlier argument (Chapter 4) concerning the use of illegal equipment by foreign vessels in South African waters, it follows that the late 1970s saw a dramatic 'switch' in harvesting equipment. However, this change in equipment – and so effort – has not been factored into catch per unit effort figures. For this reason, it is inappropriate to view the historic catch per unit effort data as an homogenous series. Instead, the data should be viewed as two distinct series, namely, pre-1977 and post-1977 (Geromont and Butterworth, 1995). The concern of this study is with the more recent series. However, a further issue arises here: a large portion of the catch per unit effort data has been corrupted by inaccurate catch figures and misspecification of effort levels.[5] Indeed, the degree of corruption is so extreme that Bergh and Barkai (1996, 17) have suggested that the catch per

[3] Pers. comm. (Leslie, Sea Fisheries Research Institute).
[4] The rK value is the product of the intrinsic growth rate of the resource and the carrying capacity biomass and, as such, measures the surplus production characteristic of a resource. In the context of surplus production modelling, the rK value should be regarded as more significant than r or K values independently.
[5] The matter of data corruption is discussed more fully below.

unit effort data for the period 1976-83 are unusable.[6] This argument is accepted here and, accordingly, parameter estimates are only given for the period 1984 onwards.

Turning to discuss the values of parameter estimates, it is useful to point out a number of features of the figures. First, the dynamic surplus production model assumes that the values of the parameters r, K and q are constant. However, the estimates of these parameters, as presented in Table 5.1, are not constant. Rather, the estimates vary across time. Estimates of the intrinsic growth rate range from $r_w = 0.3092$ to $r_w = 0.4806$ on the west coast, and from $r_s = 0.6850$ to $r_s = 0.8003$ on the south coast. Similarly, estimates of the environmental carrying capacity range between $K_w = 1192000$ tons and $K_w = 1678000$ tons on the west coast, and $K_s = 269600$ tons and $K_s = 302500$ tons on the south coast. Estimates of the value of the catchability coefficient range between $q_w = 0.0119 \times 10^{-3}$ and $q_w = 0.0171 \times 10^{-3}$ per standard-boat-day on the west coast, and $q_s = 0.0037 \times 10^{-3}$ and $q_s = 0.0043 \times 10^{-3}$ per standard-boat-hour on the south coast.

There are two possible explanations as to why parameter estimates are not constant. First, recent evidence suggests that the Schaefer form of the dynamic surplus production function no longer provides the best fit for the stocks under consideration. In other words, there may be model misspecification. Evidence of model misspecification is provided by deviations in predicted catch per unit effort values generated by the Schaefer form of the dynamic surplus production model from observed catch per unit effort figures. Figure 5.2 gives a comparison of the actual and predicted catch per unit effort figures for the west and south coast hake fisheries based on the Schaefer form of the dynamic surplus production model. From this, it is apparent that, historically speaking, the predicted catch per unit effort

[6] The arguments presented by Bergh and Barkai (1996) pertain specifically to the west coast fishery. However, there is no reason to expect that the data problem does not apply equally to the south coast fishery.

figures correspond closely with actual data. Although, in more recent years, (1993 in the west coast fishery and the late 1980s in the south coast fishery), the two series have become increasingly less correlated.

Figure 5.2 (a): Predicted and Actual Catch Per Unit Effort in the West Coast Hake Fishery (1955-95)

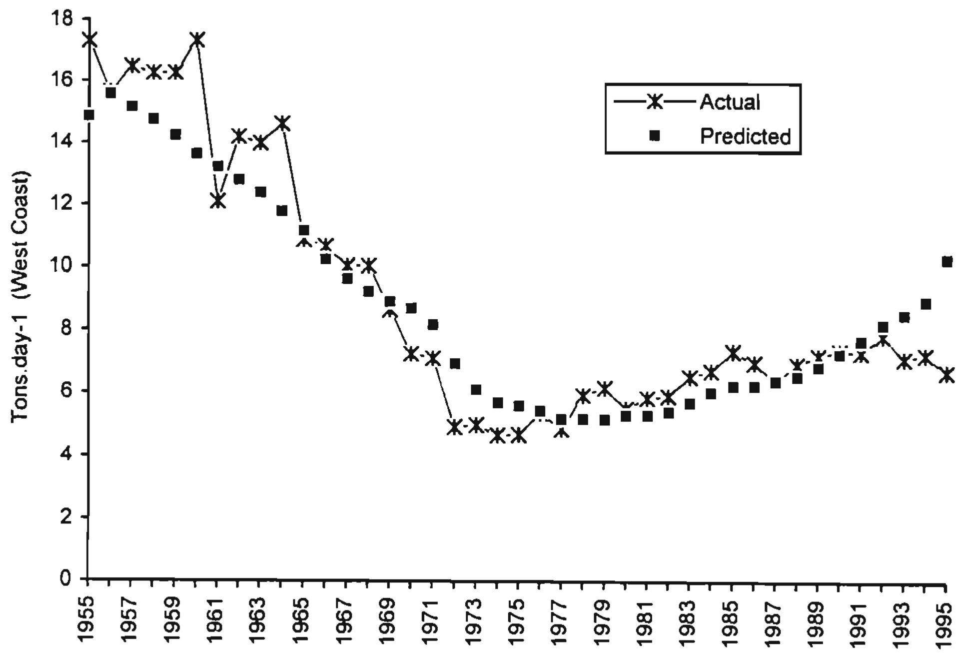
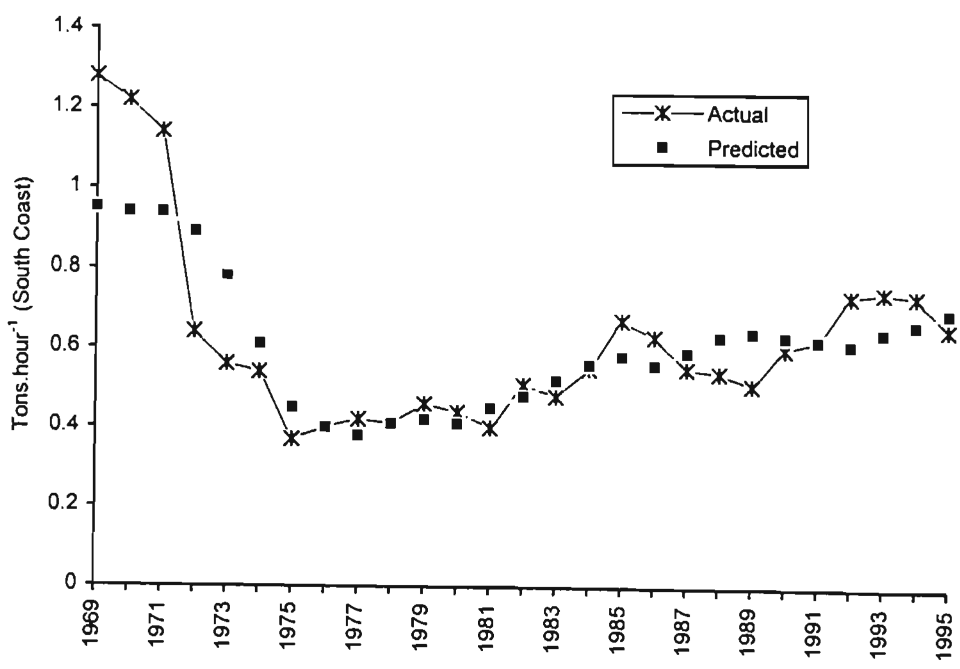


Figure 5.2 (b): Predicted and Actual Catch Per Unit Effort in the South Coast Hake Fishery (1969-95)



Source: Adapted from pers. comm. (Japp, Sea Fisheries Research Institute).

On the basis of the above, Geromont and Butterworth (1995) suggested that the Schaefer form of the surplus production function may no longer constitute the most appropriate model specification *vis-à-vis* South Africa's hake stocks. However, this view is not subscribed to in this study for a number of reasons. From Figure 5.2(a), it is apparent that the actual and predicted values of catch per unit effort are closely correlated for the period 1955-95 for the west coast fishery. Indeed, the correlation coefficient for that period is 0.9495, and is highly significant at the 1 percent level. Based on this result, the Schaefer form of the model can hardly be considered inappropriate. It would seem that since 1992 the predictive capabilities of the Schaefer form of the dynamic surplus production model have diminished as the predicted and actual catch per unit effort figures on the west coast diverge after 1992; however, this series is not sufficiently long to override the conclusions drawn from the fuller 1955-95 data set. This is particularly true in light of deviations between predicted and actual catch per unit effort figures at earlier stages in the data series, for example, between 1956-60 and 1969-75.

On the south coast, the correlation coefficient between predicted and actual catch per unit effort figures for the period 1969-95 is 0.8566, and the correlation coefficient is highly significant at the 1 percent level. Again, this would suggest that the Schaefer form of the dynamic surplus production model provides an appropriate basis for modelling the stock under consideration. The fact that the actual and predicted values begin to move in opposite directions after the late 1980s must be considered evidence of the reduced appropriateness of the Schaefer form of the model (the correlation coefficient for the period 1987-95 is only 0.2656 and is not significant at the 25 percent level). Nevertheless, it would seem inappropriate to allow the observations based on a considerably shorter (albeit more recent) data set to override the results generated by a longer data set.

Accordingly, based on the available time series data, it is accepted here that the Schaefer form of the dynamic surplus production model provides, at least for the period under consideration, a reasonably accurate specification of the production behaviour of the hake stocks located along the South African coastline. This conclusion is given further support by the results produced by Punt (1988a & 1988b), which showed that, of 106 model estimation procedures, the Schaefer form of the surplus production model provided the best estimates for the South African hake resource.[7] Accordingly, the case against the use of Schaefer form must be recognised, but can not be considered to be well-established.[8] Thus, model misspecification might be considered as part of the cause of fluctuations in estimates of parameter values, but further evidence is need to support this view.

Upon closer inspection of catch per unit effort data, a second possible explanation for variation of parameter estimates emerges. Recently, concern has been expressed about the accuracy of catch per unit effort figures in the hake fishery. The primary reason for this is that changes have taken place in the industry structure as well as in harvesting techniques that have not been factored into the catch and effort figures (Geromont and Butterworth, 1995; Bergh and Barkai, 1996). As evidence of this, Bergh and Barkai (1996, 4) argued that: 'There has been no correction [in catch per unit effort data] for the fact that prior to 1976 the effective codend mesh size in the fishery was 70-110 mm [millimetres], and after 1976 the legal minimum mesh size was 110 mm.' Furthermore, although the legal mesh size in the deep-sea fishery was changed in the mid-1970s, the change may not have been universally adopted in practice due to the presence of a substantial and largely unregulated foreign fleet (Bergh and Barkai, 1996, 15). A further distortion to catch per unit effort figures is due to the fact that most of the foreign component of the fishery was excluded after the late 1970s. In line with

[7] This does not imply that the Geromont and Butterworth (1995) argument is thought to be incorrect, but rather that, to date, there is insufficient evidence to suggest that the Schaefer form of the surplus production function is inappropriate.

[8] In this regard, it should be noted, for example, that Geromont and Butterworth (1995) offer no view on the south coast fishery. Furthermore, although the writers found that the Pella-Tomlinson function provides a more satisfactory fit to the data, this model was rejected in favour of the Fox form. The writers suggested that the reason for rejecting the better fitting model is that, in order to generate the maximum sustainable yield [sic] under the Pella-Tomlinson model, it is necessary to reduce the biomass to very low levels ($0.2K$); and reducing the biomass to such low levels is considered to involve an unjustifiable level of risk (Geromont and Butterworth, 1995, 3).

earlier arguments, this would suggest a change in the size of fish targeted which, in turn, would have implications for the size of the catch relative to effort.

Furthermore, ongoing re-evaluations of power factors for vessels in the hake fishery suggest that the increasing trend in catch per unit effort over the past fifteen years may, in part, also reflect increasing vessel and fleet efficiency rather than increased stock abundance alone (Geromont and Butterworth, 1995; Bergh and Barkai, 1996, 5).[9] In addition, whilst the reporting procedures with regard to hake catches are generally well established, and the bulk of the catch is effectively reported, some operators do not submit returns (Japp, 1996). This requires estimates to be made as to the size of some operators' catches which could lead to further (albeit small) discrepancies in catch per unit effort statistics. Finally, Bergh and Barkai (1996, 5) noted that shifts in the preferred depth and/or geographic location of fishing activity, as well as changes in discarding patterns of the commercial fleet, may have caused further distortions to catch per unit effort data.

With the above in mind, Bergh and Barkai (1996, 7) argued: 'We are now potentially faced with the reality that the CPUE [catch per unit effort] data are biased by a number of factors.' More importantly, the writers went on to suggest that removing the bias from the data might require considerable adjustments in catch per unit effort figures. This allows for two conclusions. First, given the difficulties involved in establishing a consistent data set, it is not surprising that the extant catch per unit effort data do not generate constant parameter estimates across time. Second, by employing a general linear model, which takes into account data revisions for bias, Bergh and Barkai (1996, 29) showed that the rK estimate may increase by a factor of approximately 1.13 under the revised data. However, the writers also suggested that additional problems regarding data inaccuracy may exist, and

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[9] See also Huang and Lee (1976), Squires (1987b) and Bjørndal (1989 & 1993) for reviews of the problems involved with measuring catch per unit effort over time.

argued that further analysis is required in order to generate a set of estimates in which managers can have confidence.

Accordingly, Bergh and Barkai concluded that, until such time as these further problems are resolved, managers should ignore the output of the general linear modelling process and continue to employ extant estimates. This study partially subscribes to this view: given that the estimates of the parameters r, K and q do not behave as expected, but rather vary across time, that substantial data problems exist and that the modelling process is not likely to be able to model effects such as random environmental shocks, it is argued here that range values, which allow for a considerable margin of error (rather than point values), based on extant parameter estimates provide for a more sensible modelling procedure.

The range values of the parameters r, K and q are generated by employing probability theory to construct a confidence interval which has a reasonably high probability (in this case, 95 percent) of containing the 'true' values for each of the parameter estimates. Thus, assuming that the error term of each of the parameter estimates is randomly distributed, it is possible to construct a confidence interval, such that the interval has a certain probability of containing the 'true' (but unknown) value of the mean (Gujarati, 1988, 98-102). More specifically, assuming that the 'random' variable estimates of the parameter values for the period 1984-95 are normally distributed about the mean, then the formula for the (100-α) confidence interval is given by:

$$\bar{x} \pm t_{\alpha/2} sd(\bar{x}), \qquad \text{(Equation 5.4)}$$

where $\bar{x}$ is the mean (or arithmetic average) for the given sample, $sd(\bar{x})$ is the estimated standard deviation for the sample, and $t_{\alpha/2}$ is the value of the t variable obtained from the t distribution (or small sample distribution) for the

$\alpha/2$ level of significance and $N-2$ degrees of freedom, where N is the number of observations.[10]

Returning to the sample at hand, the mean for each of the samples is given in Table 5.1, along with the standard deviation for each of the sets of observations. Assuming that $\alpha = 5$ percent, which generates a 95 percent confidence interval, from t-tables it is given that $t_{\alpha/2} = t_{0.025} = 2.228$ for ten degrees of freedom (Gujarati, 1988, 677). Based on this information, Table 5.2 sets out the upper and lower limits of the 95 percent confidence intervals of the various parameters. For completeness, the table also gives the minimum, maximum, mean and standard deviation of observed values for each of the parameters estimates, as per Table 5.1.

Table 5.2: Range Estimates for the Parameters r, K and q for the West and South Coast Hake Stocks Based on 1984-95 Values

West Coast			Parameter Scenario	South Coast		
r_w	K_w	$q_w \times 10^{-3}$		r_s	K_s	$q_s \times 10^{-3}$
0.5233	1 045 186	0.0184	Lower confidence Limit	0.8265	260 726	0.0044
0.4806	1 192 000	0.0171	Minimum	0.8003	269 600	0.0043
0.3905	1 424 167	0.0144	Mean	0.7405	285 775	0.0040
0.3092	1 678 000	0.0119	Maximum	0.6850	302 500	0.0037
0.2577	1 803 148	0.0104	Upper confidence Limit	0.6545	310 824	0.0036
0.3923	1 172 100	0.0144	Mean	0.7414	305 885	0.0040
0.0596	170 099	0.0018	Standard Deviation	0.0386	11 243	0.0002
15.19	14.51	12.50	Coefficient of Variation (%)	5.21	3.68	5.00

At this point, it is useful to clarify the interpretation of the confidence interval. Given the 95 percent confidence interval of, for example $0.2577 \le r_w \le 0.5233$, it can be concluded that, in the long run, such intervals will contain the true estimate of r_w in 95 out of 100 cases. In other words, given the sample at hand, there is a high degree of confidence (95 percent)

[10] For a fuller discussion, see Gujarati (1988, 98-104 & 642-3) or Maddala (1989, 19-20).

that the interval $0.2577 \leq r_w \leq 0.5233$ contains the true estimate of r_w. Another point to note is that, without exception, the minimum and maximum estimates for each of the parameters lie within 1.5 standard deviations of the mean (see Table 5.2). Given that the critical t value for a two-tailed 95 confidence interval with 10 degrees of freedom is $t_{\alpha/2} = t_{0.025} = 2.228$, it holds (from Equation 5.4) that the confidence interval constructed for each of the parameters has a range of 2.228 standard deviations either side of the mean. In other words, the 95 percent confidence interval has a total range of 4.456 standard deviations around the observed mean. In turn, this implies that the lower and upper critical limits for each of the confidence intervals allows, approximately, for a further 0.7 standard deviations either side of the minimum and maximum estimates generated by the Schaefer form of the dynamic surplus production model for the period 1984-95.

In short, the 95 percent confidence interval allows for lower and upper critical limits that incorporate at least a 50 percent margin of error (0.7228 as a percentage of 1.5000) in the estimates of the lower and upper limits of the confidence interval. Although the range may seem exaggerated, it is by no means out of step with the arguments forwarded by Bergh and Barkai (1996), where it is suggested that removing bias from catch per unit effort data may alter extant parameter estimates by as much as 50 percent. Furthermore, range estimates provide a guide to the sensitivity of the model to parameter values.

Before proceeding, a few notes should be made on the relative size of the confidence intervals for each of the biological parameters. The intrinsic growth rate of the biomass, which represents the maximum relative growth rate of the population (and is the approximate rate of exponential growth when the biomass is less than the carrying capacity of the environment) (Clark, 1985, 12), is given by the intervals $0.2577 \leq r_w \leq 0.5233$ and $0.6545 \leq r_s \leq 0.8265$ for the west coast and south coast stocks, with means of $r_w = 0.3905$ and $r_s = 0.7405$ respectively. From this information, it can be concluded that the south coast hake stock is modelled as having a growth rate of the order of 1.5-2.5 times greater than that of the west coast stock. There are two main reasons for this. First, in line with the arguments presented in Chapter 4, the South African hake stock consists of two species, *M. capensis* (shallow-water hake) and *M. paradoxus* (deep-water hake). *M. capensis* exhibits a faster growth rate than *M. paradoxus*. Although this difference is not recognised explicitly in modelling, it is recognised implicitly, because the species occur in different proportions in the west and south coast fisheries. On the west coast, it is the slower growing *M. paradoxus* which dominates the fishery (making up about 85 percent of the catch by mass). In contrast, the south coast fishery is dominated by the faster growing shallow-water hake, *M. capensis* (making up about 70 percent of the catch by mass). Second, the difference in growth rates between the west and south coast stocks is accentuated by the fact that, *ceteris paribus*, the warmer south coast waters promote higher growth rates than the cooler west coast waters.

It was noted in Chapter 4 that the sizes of the pristine or unexploited biomass of the west and south coast fisheries are significantly different. Typically, the carrying capacity of the west coast fishery is modelled as being as much as six times greater than the south coast fishery (Punt, 1992a, 1992b & 1994; Bergh and Barkai, 1993; Leslie, 1994 & 1996; Payne and Punt, 1995). The range estimates produced by the 95 percent confidence interval reflect this feature. Specifically, the confidence interval for the west coast resource is $045186 \leq K_w \leq 1803148$, whereas that for the south coast is $260726 \leq K_s \leq 310824$, with the means of the estimates of carrying capacity for the west and south coast equal to $K_w = 1424167$ and $K_s = 285775$ respectively. From this it can be concluded that, for the purposes of this study, the carrying capacity of the west coast is assumed to be 4.0-5.8 times greater than the south coast fishery. This substantial difference between the carrying capacities of the two fisheries can be ascribed to a single factor: upwelling. The west coast stock is located in an area of major upwelling which produces waters that are rich in biological resources.[11] The impact of

[11] There are only five areas of major upwelling in the world: the west North American and South American coasts; the north-west and south-west African coasts; and the east African/south Arabian coast.

upwelling on resource richness is underscored by the fact that the sizes of the west and south coast fisheries are comparable: the west coast fishery covers approximately 34 000 square nautical miles, versus the 29 000 square nautical miles taken by the south coast (Japp *et al*, 1994, 123).

The 95 percent confidence interval for the catchability coefficient (q) – where q measures the rate at which fish are harvested per unit of standard effort and level of biomass (Equation 3.7) – is calculated as $0.0104 \leq q_w \leq 0.0184$ and $0.0036 \leq q_s \leq 0.0044$ in the west and south coast fisheries respectively. The means of the estimates of the catchability coefficient for the west and south coast fisheries are $q_w = 0.0144$ and $q_s = 0.0040$.

Two points should be noted with respect to the catchability coefficient. First, although the parameter estimates of r and K are assumed to be fixed and constant, this is not necessarily the case with the catchability coefficient, because improvements in fishing technology or the skill of fishers will lead to a higher catch for given biomass and effort levels. As noted in Chapter 4, technological improvements have had an effect on catch rates in most fisheries. However, the effect has been significantly less pronounced in the hake fishery. As Payne and Punt (1995, 21) noted: 'apart from electronics, the gear in use today has not changed very much [since the inception of the trawl fishery]'. Given the problems of modelling and data measurement noted above, it is reasonable to expect that the estimated value of the catchability coefficient varies over the period under consideration. None the less, it is surprising that the coefficient falls across time in the west coast fishery. Again, one must point to biased (or inaccurate) data and possible model misspecification, as well as the failure or inability of scientists to factor all influences into the modelling process (such as changes – albeit very modest – in technology levels or environmental influences). Furthermore, the fact that the catchability coefficient in the west coast fishery not only varies across time but, counter-intuitively, falls across time, reinforces the need to employ range values for parameters as opposed to point values.

Second, the range estimates of the catchability coefficient are 3-4 times greater for the west coast than for the south coast. There are several factors that should be brought into consideration in relation to this differential. The west coast biomass is substantially larger than the south coast biomass. Accordingly, it is expected that, for a given level of effort and technology, the west coast catch would be higher. However, effort in the south coast fishery is measured in standard-boat-hours, whereas the unit of measurement in the west coast fishery is standard-boat-days. This complicates comparisons between the catchability coefficient in the two fisheries, because a standard-boat-day in the west coast fishery is not measured as a twenty-four hour period but rather consists of as many as three to four trawls a day depending on the time of year.[12]

The reason for measuring effort in the west coast fishery in days is that it is assumed that no fishing time is lost getting to the west coast fishing grounds due to their proximity to the major fishing ports of Cape Town and Saldanha. To get to the south coast grounds, however, normally requires a day's steaming. For this reason, effort on the west coast is measured in days and on the south coast in hours. More recently, however, it has become practice for skippers to fish 'on the way' to the south coast ground, effectively maximising their fishing time. Finally, as noted in Chapter 3, there are differences between gear employed in the west and south coast fisheries. All of the above factors effectively render comparisons of the catchability coefficients in the west and south coast fisheries largely meaningless.

5.3.2 ECONOMIC PARAMETERS (δ, c AND p)

The second set of parameter values required to estimate the bioeconomic model are the various economic parameters, which include the cost of harvesting (c), measured as cost per unit of catcher effort; the price received

[12] Trawling takes place during daylight hours because hake migrate vertically upwards at night, so trawling is not considered to be economically viable after dark. Accordingly, trawling generally commences at first light and ends at sunset. As a result, fishing hours are longer in summer, which allows for a greater number of trawls. Between 1989 and 1995, the average number of hours trawled has fluctuated between 7.22 and 7.85 hours per vessel per day.

by harvesters for hake (p), measured by price per unit; and the discount rate (δ). As is the case with the biological parameters, the dynamic bioeconomic model assumes that the values of the parameters δ, c and p are fixed. This section concerns itself with determining the values of these parameters for the west coast and south coast hake fisheries.

Turning first to examine price and cost functions, it should be recalled that the assumption is made that prices are autonomous and that the cost function is linear. To put matters differently, the assumption is made that unit prices do not change with output and that costs, calculated as cost per unit of catcher effort, increase proportionately (constant marginal cost) with effort and catch. In other words, the supply curve is assumed to be perfectly elastic and unit (or marginal) costs are assumed to be constant. Accordingly, the problem of estimating price and cost functions becomes one of estimating operating margins, that is, the difference between revenue and costs per unit of output (or price and unit costs), before deducting for interest and taxation charges. Given South Africa's inflationary environment, it is arguably more useful to generate price and cost figures based on operating margins – reflecting the real price:cost ratio – as opposed to nominal price and cost data.

The task of establishing operating profit margins is complicated by a number of factors. To start with, although accurate data on the landed price of hake are readily available,[13] reliable information on trawling costs is extremely difficult to access. In fact, the only information acquired from the various companies involved in the fishery was from those listed on the Johannesburg Stock Exchange, namely Irvin and Johnson Limited and Sea Harvest Corporation Limited (and its parent, ICS Holdings Limited). However, given that Irvin and Johnson Limited and Sea Harvest Corporation Limited collectively account for four-fifths of the hake catch, the data provided by these two companies are assumed to be highly representative of the hake fishery as a whole. Unfortunately, the only information supplied by those companies is the same as that made available to ordinary shareholders, as

[13] See, for example, Van D. Boonstra (1992, 1993 & 1994) or Stuttaford (1994 & 1995).

presented in the companies' annual financial statements. Moreover, the only available data on operating margins are those for the company as a whole, and not for specific operations, such as the trawl fleet. Perhaps the low level of disclosure is not surprising given the highly concentrated structure of the industry. None the less, when coupled with other industry information, company level operating margins do provide a first guide to operating margins of the hake trawling operation. This information, in turn, provides the basis for establishing what would seem to be relatively reliable estimates of operating margins in the trawl fishery. The arguments are detailed below.

Table 5.3 sets out the operating profit margins for all companies listed on the Johannesburg Stock Exchange that are involved directly in the South African fishing industry for the period 1984-95. The set of companies includes Irvin and Johnson Limited (I&J); Sea Harvest Corporation Limited (Sea Harvest), and its parent, ICS Holdings Limited (ICS); Oceana Fishing Group Limited (Ocfish); and Natal Ocean Trawling Limited (Natrawl). For comparative purposes, Table 5.3 also includes the two listed Namibian fishing operations, Namibian Fishing Industries Limited (Namfish) and Namibian Sea Products Limited (Namsea).

The data presented in Table 5.3 allow for a number of observations. First, margins vary substantially between companies (as well as within companies across years). The large degree of variation in margins within companies is evidenced by the wide range between the minimum and maximum operating margins, as well as the high coefficients of variation. Indeed, with the exception of Natrawl, the minimum and maximum values of operating margins lie outside of one standard deviation of the mean; and the substantial differences that exist between companies' operating margins is evidenced by large differences in the minimum, maximum and mean values of operating margins between the companies. More specifically, minimum values range between -4.10 (Namfish) and 17.49 (Sea Harvest), while maximum values range between 6.15 (ICS) and 60.50 (Namsea). Similarly, the means range between a low of 2.78 (ICS) and a high of 26.51 (Namsea). In short, the high level of variability in margins between and within fishing companies listed on the Johannesburg Stock Exchange would seem to complicate the task of arriving at a reliable and representative estimate of operating margins in the hake fishery.

Table 5.3: Operating Profit Margins of Fishing Companies Listed on the Johannesburg Stock Exchange (1984-95)

	ICS	I&J	Namfish	Namsea	Natrawl [14]	Ocfish	Sea Harvest [15]
1984	3.64	5.69	25.77	24.90	8.84	18.26	-
1985	2.24	6.45	43.78	60.50	13.61	15.18	-
1986	1.78	8.11	39.40	56.19	24.96	18.07	-
1987	2.65	9.61	38.00	37.31	29.79	19.03	-
1988	2.95	9.43	34.23	38.97	21.66	21.23	-
1989	2.98	7.09	28.09	25.22	13.70	12.62	22.90
1990	1.25	6.91	21.52	20.54	1.59	7.70	22.00
1991	1.33	6.87	15.09	6.65	11.91	6.50	24.20
1992	1.24	6.55	-4.10	7.54	9.98	9.66	28.80
1993	2.76	5.10	6.66	15.62	13.02	11.59	18.90
1994	4.40	4.97	8.85	11.70	9.40	10.70	17.49
1995	6.15	5.23	1.04	12.95	-	11.46	20.77
Mean	2.78	6.83	21.53	26.51	14.41	13.50	22.15
Minimum	1.24	4.97	-4.10	6.65	1.59	6.50	17.49
Maximum	6.15	9.61	43.78	60.50	29.79	21.23	28.80
Standard Deviation	1.44	1.56	15.98	18.51	8.06	4.77	3.72
Coefficient of Variation (%)	51.92	22.84	74.22	68.47	55.96	35.33	16.81

Source: Adapted from McGregor (1988, 1992 & 1996).

This point could not be better illustrated than by the fact that, of the listed companies operating in South African waters (excluding ICS which holds 72 percent of Sea Harvest), the lowest and highest values for the mean

[14] Natrawl was first listed in 1987, but data are available from 1984 onwards. However, the company was placed in provisional liquidation toward the end of 1994. As a result, there is no figure for 1995, and the figure for 1994 is for the first half of the financial year, and not the full financial year.
[15] Sea Harvest was first listed in 1994, data are only available from 1989 onwards.

operating margins are recorded by the two dominant operators in the hake fishery, namely I&J and Sea Harvest. It should be noted, however, that I&J is a relatively diverse company, with just 15 percent of turnover coming from its seafoods division (Irvin and Johnson Limited, 1994 & 1995). Sea Harvest, on the other hand, generates its turnover almost exclusively from fishing. Even more important is the fact that the principal business of Sea Harvest is deep-sea trawling (and the processing and marketing of its products). Rough estimates suggest that at least 50 percent of Sea Harvest's turnover is derived from the hake fishery.[16] Accordingly, although estimates of operating margins vary substantially between companies, it is apparent that the estimates of highly diversified operators (I&J) are of little value in estimating operating margins in the hake fishery, and that a better idea of the size of operating margins in the hake fishery is given by more concentrated operators, such as Sea Harvest.

This argument is given further support by the size of operating margins of ICS. Similarly to I&J, ICS is a diversified investment holding company that owns, *inter alia*, 72 percent of Sea Harvest and derives approximately one-third of its operating profit from that company. More important, however, is the fact that, for the period under consideration, ICS has a mean operating profit margin of 2.78 percent, equal to just one-eighth of Sea Harvest's mean operating margin. Furthermore, margins for Namfish, whose principal activities involve trawling for hake, although in Namibian waters, fall within the upper end of the spectrum of the range of mean operating margins recorded for the various companies. Specifically, while margins for the Namibian operation are more volatile, the mean operating margin for the company over the period 1984-95 of 21.53 percent, is almost equal to the 22.15 percent recorded by Sea Harvest. As an aside, it is perhaps useful to note that the lower volatility in Sea Harvest's operating margins is probably attributable to

[16] As noted earlier, companies involved in the hake fishery provide almost no information on the details of their operations. Accordingly, this estimate is derived from various sources. These sources include Japp (1996), where accurate estimates of catch by company are given, Stuttaford (1995), which provides information on landings, landed values and wholesale values of catch; and the annual reports of Sea Harvest (Sea Harvest Corporation Limited, 1995).

the fact that the company also operates in the more stable inshore fisheries (which include the inshore trawl fisheries for hake and sole, as well as harvesting operations which direct their efforts at mussels, oysters and lobster). In summary, the above analysis suggests that operating margins for fishing companies that are involved primarily in the hake fishery are of the order of approximately 20-22 percent.

There are a number of possible reasons for questioning the accuracy of this estimate. To start with, the suggestion that the operating profit margins of Sea Harvest and Namfish provide a good guide to the margins in the hake fishery must be questioned on the grounds that both companies operate in fisheries other than hake. Sea Harvest operates in inshore fisheries, where, apart from hake, it is involved in harvesting sole, mussels, oysters and lobster. It is argued elsewhere that these inshore resources typically produce higher operating profit margins than offshore resources (Bergh and Barkai, 1996, 14). Accordingly, it could be concluded that using Sea Harvest's mean operating profit margin as a proxy for operating margins in the hake fishery is misleading as the estimate is biased upwards. However, this argument is rejected on the grounds that not only are there insufficient data for distinguishing between the margins in the hake fishery and other operations, but, further, the figure for Namfish is derived from an operation which is devoted almost exclusively to trawling, deriving only a small portion of its income (and costs) from a holding in United Fisheries Limited, which is involved in the pelagic fishery.

However, there are two further possible reasons, relating to price and cost data, for questioning the accuracy of the above estimate of operating margins. With regard to price, the operating margins quoted above are based on wholesale (free on board processed) prices – I&J and Sea Harvest both process their entire catch – and not on landed prices. This is contrary to the specifications of the bioeconomic model. However, accurate figures for landed prices are available (Stuttaford, 1995 & 1996), and the data show that wholesale prices are of the order of twice as great as landed prices. This suggests that the quoted operating margins are inflated by a factor of two. However, the cost figures used to generate the above estimates of operating margins are also distorted: the figures include processing as well as harvesting costs. By stripping processing costs out of total company costs, it is possible to arrive at harvesting costs, as required by the dynamic biological model. Once again, accurate disaggregated cost data are generally not available. Casual empiricism, however, suggests that the differential between total costs and harvesting costs is considerable. For example, in the case of Sea Harvest, depreciation charges on fixed assets other than fishing trawlers (which includes buildings, plant, machinery, equipment and motor vehicles) are as great as depreciation charges on fishing vessels. Based on this, it is assumed that trawling costs are only half as great as trawling plus processing costs. In short, if the above arguments are taken into account, whilst prices are argued to halve, so are costs, leaving the estimates of operating profit margins unaffected.

As a final point, it should be noted that confidence in the accuracy of the above estimate of operating margins is bolstered by the estimate of operating margins in the trawl fishery provided in the *Report of the Multi-Disciplinary Task Group on the First Phase of the Longline Experiment* (Van Zyl, Undated, 11).[17] There, it was estimated (for 1993) that the average costs of trawling are R1.68 per kilogram.[18] Based on the average landed price of hake of R2.18 per kilogram, it can be concluded that the mean operating profit margin in the hake fishery for that year was in the region of 22.94 percent. Based on the above arguments, it is assumed that the mean operating profit margin of 22.15 percent, as calculated for Sea Harvest, provides a reasonably accurate

[17] In spite of an extensive literature search and extensive discussions with various industry players, this is the only other estimate of operating profit margins in the hake trawl fishery known to the current study.

[18] The paper also provides an extensive review of a host of problems encountered in attempting to arrive at accurate estimates of costs in fisheries, including the valuation of vessels and equipment; the allocation of overheads between harvesting and processing operations; and valuing salaries and wages where owners of smaller operations often form part of the crew. Because the current study employs a 'top down' approach to estimate margins, as opposed to the 'bottom up' approach used by the Multi-Disciplinary Task Group on the first Phase of the Longline Experiment (Van Zyl, Undated), these issues are not considered in detail here.

guide to operating margins in the South African hake fishery. For convenience, a figure of 20 percent is initially assumed in the estimation procedure below.

Before proceeding, it should be stressed that excessive emphasis should not be placed on the figure of 20 percent for at least three reasons. First, in the absence of more accurate data, the figure of 20 percent is intended as a proxy and not as a precise estimate. Second, the figure is assumed to be an industry average, and margins are likely to vary considerably between companies (as well as across years) due to changing costs and prices. On the cost side, per kilogram costs are typically higher for hake harvested by wetfish trawlers as opposed to factory freezers, and even higher for longliners (Van Zyl, Undated). Reasons for this include economies of scale enjoyed by processing at sea in the case of factory freezers, and the substantially higher labour costs and absence of economies of scale in the case of labour-intensive longlining operators (Van Zyl, Undated). Thus, operating margins are influenced by the proportion of catch taken by the various techniques. On the price side, margins are influenced by the portion of output sold to export markets, where prices are as much as 10 percent higher than in domestic markets (Van Zyl, Undated, 12), as well as the quality and size of individual fish (Stuttaford, 1995, 29). Again, in the virtual absence of reliable information, it is impossible to provide even crude estimates of the impact on margins for individual companies of these various influences. Third, the above estimates of operating margins are based on accounting costs and not economic costs. In the economist's definition, total cost includes the opportunity cost of any capital, labour, or other inputs supplied by the owners of the firm. Accounting costs, however, are calculated entirely on the basis of historic costs. For this reason, accounting costs are typically lower than economic costs. This suggests that the operating margin figures set out above are biased upwards. However, in the absence of more reliable estimates, the accounting operating margins must be assumed to be reliable proxies for economic operating margins. This assumption is not out of line with the methodology employed in studies of a similar nature (DuPont, 1991;

Stead *et al*, 1996) and with evidence from fisheries of a similar nature in other countries.[19]

Accepting that operating profit margins are approximated by a figure of 20 percent, the final step in the process involves constructing price and cost indices. To this end, the price per unit (ton) of hake is initially set equal to unity. From this, it is possible to arrive at a cost index per unit of catcher effort. For this task, the average harvest per unit of catcher effort is set equal to the annual average for the period 1984-95. For the west coast, the average annual catch is 93 520 tons per annum with an average effort of 13 320 standard-boat-days per annum (Japp, 1996). This yields an average harvest of 7.02 tons per standard-boat-day per annum. Then, based on a price index of one, and an operating profit margin of 20 percent, it is possible to arrive at a value for the cost index. In this instance, the cost index is calculated as equal to 5.85. Similarly, for the south coast fishery, the average annual catch and effort over the period 1984-95 are 49 410 tons and 79 850 standard-boat-hours respectively (Japp, 1996). This reduces to a figure of 0.62 tons per standard-boat-hour which, in turn, based on a price index of unity, generates a cost index of 0.52. Accordingly, in estimating the model for the west and south coast fisheries, a price index of one is employed in both instances, with the cost index set equal to 5.85 in the case of the west coast fishery and 0.52 in the south coast fishery.

In terms of the bioeconomic model presented in Chapter 3, a primary determinant of the optimal – that is, discounted rent maximising – biomass, catch and effort levels is the discount rate. Conventional discounting converts all future costs and benefits of fishing into net present values using a constant, positive discount rate. The further into the future benefit and cost streams occur, the lower their present value. A crucial issue that arises here is: 'What value should the discount rate take?'. In essence, the answer to this question hinges around the rationale for discounting where, generally

[19] *The Economist* (1997) estimates that profit margins in the Icelandic cod fishery are of the order of 20-25 percent.

speaking, neo-classical economists offer two main motivations for discounting: consumers' time preference and the opportunity cost of capital. These are represented by the consumption rate of interest and investment rate of interest respectively, although with perfectly free capital markets, no taxation and no uncertainty, these 'rates of interest' would be equal. Of course, such conditions are never extant, which means that policy makers are generally forced to make a choice between these two rates (Lumby and Saville, 1995, 5).

In practice, adopting either the consumption rate of interest or investment rate of interest as the discount rate is problematic for a number of reasons. To start with, using the consumption rate of interest as the discount rate requires the assumption that individuals' preferences are intransitive across time; that consumption now is always better than consumption later; that all costs and benefits can be converted into consumption streams; and that the economy grows at a constant rate across time (Markandya and Pearce, 1988, 6). Clearly, these assumptions are heroic. Adopting the investment rate of interest as the discount rate is equally problematic. The investment rate of interest reflects the marginal productivity of investment, equal to the real rate of interest as determined by capital markets. However, investment rates of interest are distorted by progressive tax structures and the presence of differentials between individual and corporate tax rates. More seriously, using the investment rate of interest as the discount rate requires that only investment resources are displaced and that all future costs and benefits are consumed. Moreover, from a practical perspective, there is no single rate of return on investments, but rather a wide spread of rates that are influenced by, *inter alia*, the risk and maturity structures of investments (Lumby and Saville, 1996, 325).

In short, employing either the investment rate of interest or the consumption rate of interest as the discount rate is generally considered to be highly problematic, and this is no different in the case of the dynamic bioeconomic model applied to South Africa's hake fishery. While a possible solution might involve adopting a weighted average of the two rates – the accounting rate of interest – as the discount rate (Markandya and Pearce, 1988, 6), the problems that arise out of employing either the consumption rate of interest or the investment rate of interest as the discount rate are not dissipated by employing some weighted average of these market rates. Indeed, as an aggregated measure, the weighted average is likely to involve more problems than employing either of the rates independently. Given the problems that exist with adopting a market-determined discount rate, it is necessary to examine alternatives to the so-called conventional discount approach. Three main alternatives exist: zero-rate discounting; discounting at the internal rate of return; and quasi-discounting. These options are explored below.

Zero-rate discounting is synonymous with believing that future costs and benefits should be weighted equally to current costs and benefits. However, just as there is no reason to expect benefits, for example, to fall exponentially with time, so there is no reason to expect benefits to remain constant with time. Indeed, zero-rate discounting is nothing more than a special case of discounting at a positive constant rate. Furthermore, as contended by Common (1983, 93-103), adopting a zero rate results in meaningless outcomes under certain conditions. For example, where non-renewable resources exist in a 'technological vacuum', adopting a discount rate of zero implies equal shares for all generations of these finite resources over infinite time, that is, nothing for everyone.[20] Surely this cannot be the outcome desired by advocates of zero-rate discounting.

The internal rate of return is defined as the rate at which discounted benefits are equal to discounted costs. As such, the internal rate of return provides the weighting that should be placed on future values, and represents the highest level of interest with which the investment can compete. Therefore, the internal rate of return can be compared with some 'cut-off rate', and in this way facilitates decision making *vis-à-vis* resource allocation. This method of

[20] This result, however, only holds in this simple 'cake-eating' model and does not apply in the instance of renewable resources.

establishing intertemporal resource allocation is, however, flawed in three important respects (Price, 1993, 41). First, employing the internal rate of return to discount future values is based on the assumption of continuous reinvestment of revenues. Yet it is difficult to assume that all revenues will be in reinvestible form (Price, 1993, 75). Second, the net cash flow of ar. investment may change from positive to negative, and vice versa, over the lifespan of a project. If this occurs, then there may no longer be a single internal rate of return, but rather multiple rates at which discounted costs will equal discounted benefits. This creates uncertainty as to which is the correct rate to employ in evaluating the project.[21] Third, selecting projects in descending order of internal rate of return (up to some 'cut-off rate') may favour projects with late costs, which makes the internal rate of return an 'unsafe' decision criterion (Price, 1993, 49).[22] In short, although the internal rate of return is easily calculable, and is thus desirable from a practical point of view, adopting the internal rate of return involves a number of problems which undermine the efficacy of this particular discounting technique.

In considering the above arguments, Price (1993, 291) suggested that no universal discount rate can be defined, and argued that, ideally, project evaluation should impute utility directly to each act. In other words, discount rates are 'project-specific' or, in the current context, 'resource-specific'. Thus, discounting involves the adoption of a quasi-discount rate, which reflects the expected trend of utility under a projected steady trend of circumstances. However, as Price is at pains to point out, this quasi-discount rate is a parameter that is calculated – on the basis of the expected rate of growth of consumption for a specific resource and the marginal utility of consumption of that resource – to represent the trend of utility and not, as with conventional discounting, an arbitrarily selected rate used to determine the trend of that utility.

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[21] Wright (1963) suggests that the correct rate to employ is the lowest rate.

[22] This result is at odds with Common's (1988) argument regarding 'late costs' which holds that there can be no general presumption that the use of lower interest rates in cost benefit analysis will reject more projects with long-run environmental effects (Lumby and Saville, 1995, 12).

There is nothing novel about this approach. Marchand and Pestiau (1984) suggest that different discount rates should be adopted for each public sector enterprise, depending on its system of financing. Ray (1984) accepts the use of different discount rates for different income groups; and Lind (1990) suggests that the discount rate selected for a project may depend upon, *inter alia*, the method of financing; the nature and arrangement of expenditures and profits; project risk; market imperfections (particularly those resulting from tax differentials); the structure and politics of the government; and the overall composition of the economy. Weitzman (1994) goes even further by suggesting that the discount rate employed in the evaluation of any project should vary over time. Howe (1990, S1) summarises this set of views well: 'discount rates, like all other prices, must be tailored to particular times, locations, types of projects and methods of financing'.

Rather than adopting a conventional constant-positive discount rate which is (arbitrarily) linked to some market rate of interest, the quasi-discount rate approach allows for the use of variable quasi-discount rates, which may produce results that are intuitively more appealing than those produced by conventional discounting. By taking into account the probabilities of discoveries of new resources, the possibility of substitution of one resource for another and the likely trajectory of technological progress, Price (1995, 226-87) concluded that extrapolating reasonable values for rates of growth of consumption and taking a moderate elasticity of marginal utility of consumption would produce a quasi-discount rate for a reasonably slow growing renewable resource, such as hake, of between 3-6 percent. Based on this result, and for simplicity, a mean rate of 4.5 percent is assumed as a reasonable value for the quasi-discount rate in the estimation procedure.[23] By coincidence, Andrew and Butterworth (1987) employ a rate of 5 percent in modelling the west coast stock.

[23] For completeness, the model is also run for quasi-discount rates of 3 percent and 6 percent, as well as zero percent. The latter number effectively amounts to reducing the dynamic bioeconomic model to a static model.

5.3.3 SUMMATION

The task of establishing accurate and reliable estimates of the various biological and economic parameters required to run the dynamic bioeconomic model, as set out in Chapter 4, is plagued by problems of data accuracy and/or availability. To be more specific, whilst obtaining estimates of the various biological parameters, r, K and q is relatively unproblematic, the accuracy of these estimates is questionable. The immediate reason for this is that the estimates of the various biological parameters generated by the surplus production model and available catch per unit effort data estimates behave 'badly' as they do not remain constant across time. There are two possible explanations as to why parameter estimates vary across time: model misspecification and data problems.

With regard to model misspecification, it has been suggested elsewhere that the Schaefer form of the surplus production model may not provide the most accurate specification of the biological features of the hake resource. This, however, is a moot point, and given the robustness of estimates produced by the Schaefer form of the biological model in the past, it would be hasty to replace the Schaefer form with an untested (or at least unproved) alternative; more work needs to be done here. The issue of data accuracy, however, is less contentious. It is widely accepted that catch per unit effort data are biased and inaccurate because of, amongst other things, technological advancement and changes in industry structure. Furthermore, while efforts have been made to redress the problems of bias and inaccuracy, more work needs to be done before a revised data set is available. In light of this, in order to cater for data bias and inaccuracy, the biological estimates used in the estimation procedure are range estimates (based on a 95 percent confidence interval) as opposed to point estimates, as it is argued that the confidence intervals generated above have an extremely high probability of containing the true values of the various biological parameters.

Estimation of values for economic parameters also involves a number of problems. First, given the assumptions of the dynamic bioeconomic model, the problem of estimating cost and price values becomes – at its simplest level – one of establishing operating profit margins, that is, profit before interest and taxation charges. However, the oligopolistic structure of the trawl fishery for hake has resulted in trawling companies being extremely reluctant to disclose any information regarding the profitability of their trawling operations. Indeed, the only information that is available is at the company level as opposed to the fishery level. This is problematic in that company data are aggregated across a variety of operations, which include the harvesting of hake as well as other stocks. In addition, over and above harvesting, companies are involved in the processing and distributing of products. The matter is compounded by difficulties involved in estimating price and cost. However, by taking into account the nature of operations, and by making various assumptions based on available price and cost data, as well as by assuming that accounting costs are a satisfactory proxy for economic costs, it is possible to generate what is, arguably, a relatively reliable estimate of operating profit margins in the hake fishery. Importantly, the accuracy of this estimate is supported by other available estimates.

With regard to the discount rate, two points should be noted. First, there is considerable controversy as to what discount rate should be employed. While the solution is far from clear-cut, it is suggested here that the discount rate is best represented by a discount rate that is adjusted to reflect the peculiarities of each resource: the so-called quasi-discount rate. As such, quasi-discounting marks a sharp break with the notion that the discount rate should be given by market rates, or some derivative thereof. Rather, quasi-discounting employs a 'resource-specific' discount rate. It is argued elsewhere, and accepted here, that a reasonable value for the hake resource, a relatively slow growing stock, lies in the range 3-6 percent. For simplicity, a mean rate of 4.5 percent is assumed below. The following section turns to estimating the optimal biomass, catch and effort figures for the South African hake fisheries.

5.4 Model Estimation: Optimal Biomass, Catch and Effort in the West and South Coast Hake Fisheries

Based on the set of parameter values generated in Section 5.3 above, the concern of this section is with estimating the dynamic rent maximising biomass, catch and effort levels for the west and south coast hake fisheries as per the estimatable form of the dynamic bioeconomic model. The estimation procedure is carried out by reproducing the model, as set out in Section 4.2, in spreadsheet format. By entering input values for each of the biological and economic parameters, the spreadsheet model calculates the dynamic rent maximising biomass, X^{*}_{MEY}, as per Equation 3.19.[24] For comparative purposes, results are also generated for the dynamic maximum sustainable yield biomass, X^{*}_{MSY}, as per Equation 5.3; as well as the static (zero discount rate) maximum economic yield biomass, X_{MEY}, as per Equation 5.2. In addition, the model calculates the total catch and effort levels associated with each biomass outcome. Catch is given by:

$$C_t = qKE_t\left(1 - \frac{qE_t}{r}\right); \qquad \text{(Equation 3.9)}$$

that is, catch is a function of the catchability coefficient (q), the carrying capacity of the fishery (K), the level of effort expended (E_t), and the intrinsic growth rate of the fish stock (r). Furthermore, the level of effort required to take any given catch is given by:

$$E_t = \frac{C_t}{qX_t}; \qquad \text{(Equation 3.7)}$$

[24] The spreadsheet software package Microsoft Excel Version 5.0 was employed for this purpose.

that is, total effort is a function of the total catch (C_t), the catchability coefficient (q) and stock size (X_t). All else equal, the larger the stock size and the greater the catchability coefficient, the lower the level of effort required to take a given catch.

As outlined above, the estimation procedure assumes range values for each of the biological parameters. Accordingly, the model simulates the optimal outcome for a range of parameter values, rather than point estimates. In this regard, the range of estimates selected for the simulation is given by the parameter sets discussed in Section 5.3 and set out in Table 5.2. The parameter sets are made up of the minimum, maximum and mean values generated by the Schaefer form of the biological model for each of the parameters, as well as the two, more extreme, estimates given by the lower and upper limits of the confidence intervals calculated for each of the parameters in Section 5.3. The values taken by the economic parameters, cost (c) and price (p), are approximated by an operating profit margin figure, which is argued above to be (of the order of) 20 percent. The margin is represented by an index, whereby the cost parameter, as noted, takes a value of 5.85 and 0.52 in the west and south coast fisheries respectively, and the price parameter takes the value 1.0. A mean value of 4.5 percent is assumed in the modelling process for the quasi-discount rate. (For completeness, the spreadsheet model also simulates the outcomes for rates of 3 and 6 percent, as well as for the static [zero discount rate] model.)

The detailed results of the model simulation for each scenario are given in Appendix 2, and summarised in Tables 5.4 and 5.5 below. The tables set out the optimal, that is, dynamic rent maximising, biomass for both the west and south coast fisheries. For comparative purposes, Tables 5.4 and 5.5 also set out the maximum sustainable yield biomass. The catch and effort levels associated with each outcome are also shown.

Table 5.4: Biomass, Catch and Effort Levels under the Dynamic Bioeconomic Model

West Coast Hake Fishery						
Maximum Sustainable Yield			Scenario	Maximum Economic Yield		
Biomass (Tons)	Catch (Tons)	Effort (Days)		Biomass (Tons)	Catch (Tons)	Effort (Days)
744 140	112 626	14 553	Lower Confidence Limit	1 105 493	110 225	9 587
716 895	126 962	14 882	Minimum	1 021 458	123 575	10 166
630 025	137 188	15 122	Mean	871 406	132 074	10 525
540 195	141 963	15 368	Maximum	737 146	135 186	10 725
477 654	135 725	15 443	Upper Confidence Limit	658 324	127 512	10 527
621 782	130 893	15 074	Sample Mean	878 765	125 714	10 305
113 485	11 561	365	Sample Standard Deviation	187 397	9 719	450
18.25	8.83	2.42	Sample Coefficient of Variation (%)	21.33	7.73	4.37

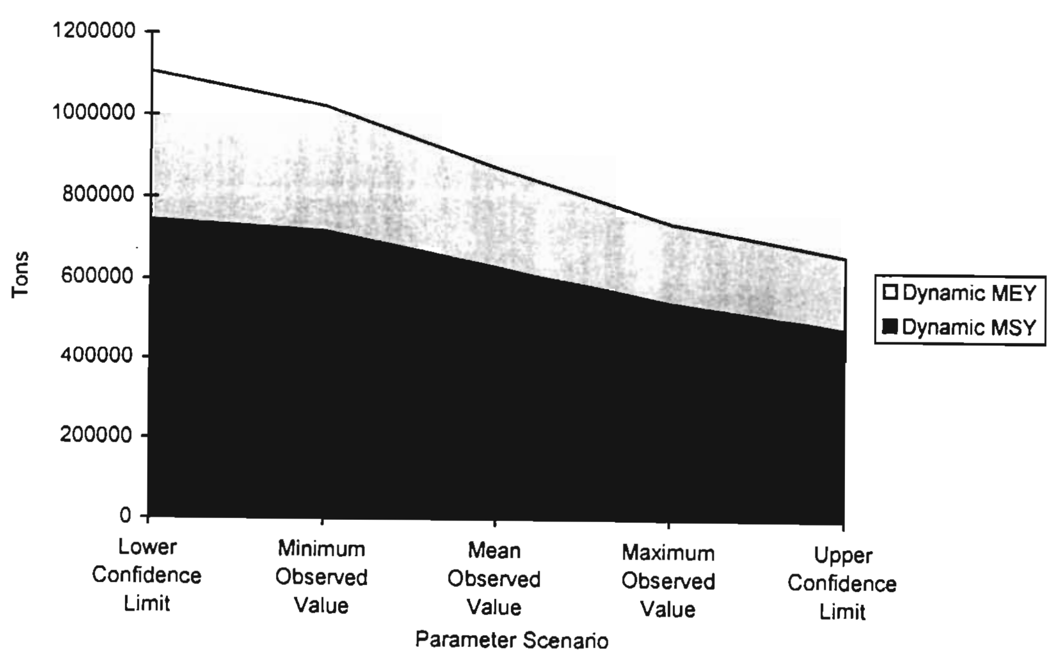
Table 5.5: Biomass, Catch and Effort Levels under the Dynamic Bioeconomic Model

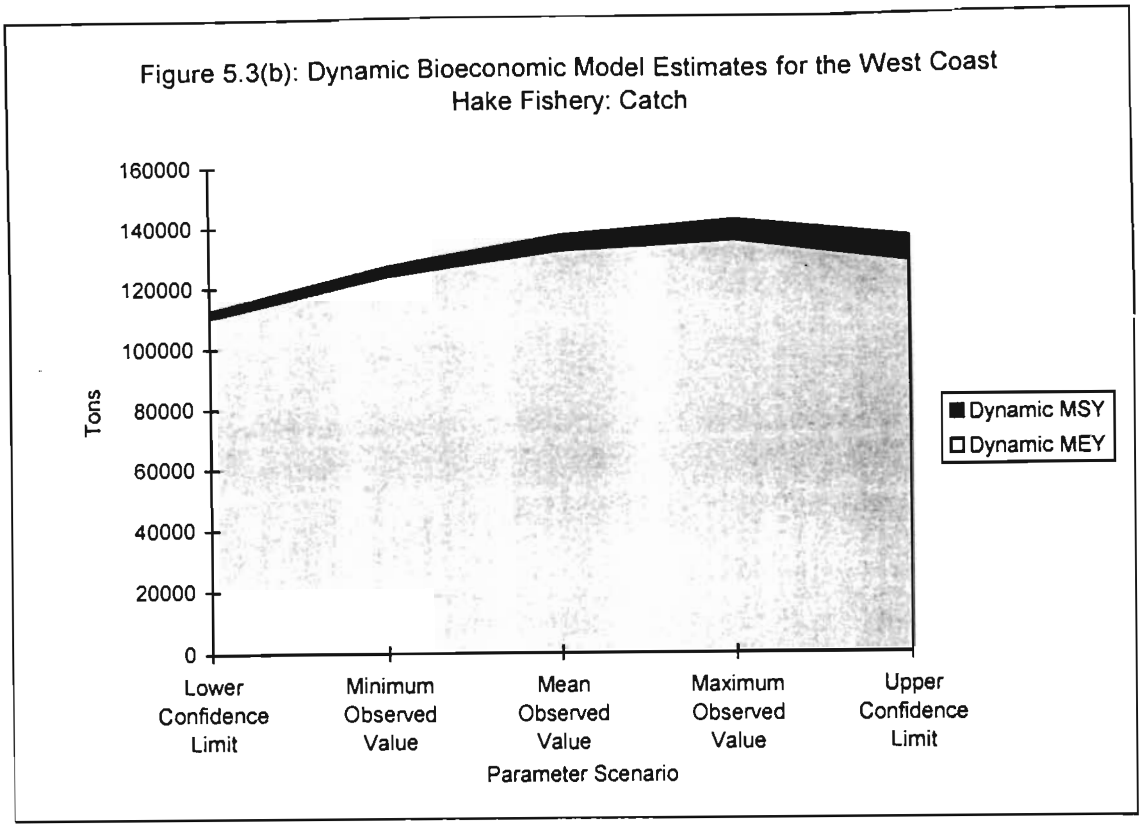
South Coast Hake Fishery						
Maximum Sustainable Yield			Scenario	Maximum Economic Yield		
Biomass (Tons)	Catch (Tons)	Effort (Hours)		Biomass (Tons)	Catch (Tons)	Effort (Hours)
144 727	50 618	97 153	Lower Confidence Limit	223 844	40 998	50 876
141 314	51 580	98 649	Minimum	217 990	41 717	51 722
134 204	52 709	98 188	Mean	204 718	42 998	52 509
127 220	53 770	98 291	Maximum	192 448	44 075	53 261
123 265	53 713	99 034	Upper Confidence Limit	186 846	43 759	53 227
134 146	52 478	98 263	Sample Mean	205 169	42 708	52 319
9 079	1 371	704	Sample Standard Deviation	15 897	1 321	1 023
6.77	2.61	0.72	Sample Coefficient of Variation (%)	7.75	3.09	1.96

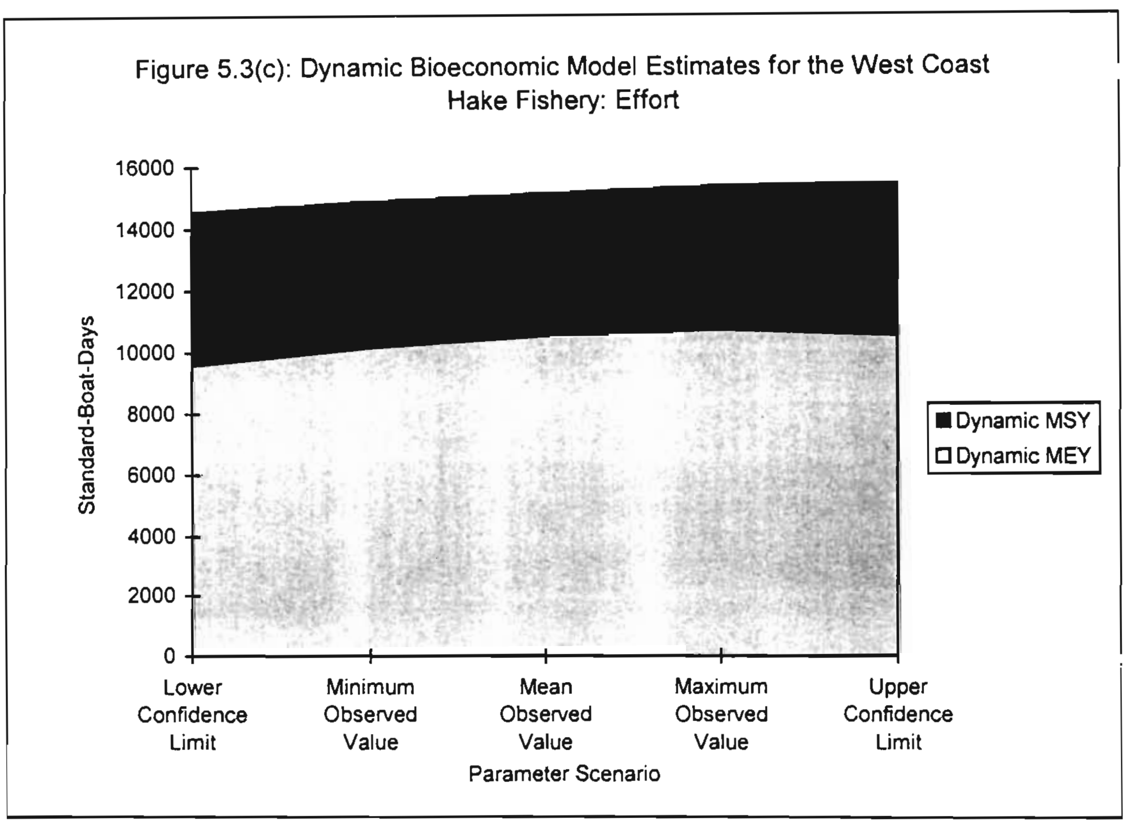
5.4.1 The West Coast Fishery: Dynamic Maximum Economic Yield versus Dynamic Maximum Sustainable Yield

The results of the bioeconomic model have implications of varying proportions for the west and south coast fisheries. Considering the west coast fishery first, the dynamic bioeconomic model calculates that, for the various biological parameter values and given economic parameter values, the dynamic rent maximising biomass ranges between 658 324 tons and 1 105 493 tons, and that the sustainable catch associated with this biomass lies between 110 225 tons and 135 186 tons per annum. The level of effort required to take the optimal catch is calculated to be of the order of 9 587-10 725 standard-boat-days per year. In contrast, the model calculates that the dynamic maximum sustainable yield biomass ranges between 477 654 tons and 744 140 tons which generates an annual catch of 112 626 - 141 963 tons requiring an effort level of 14 553 - 15 443 standard-boat-days. These results are summarised in Figure 5.3 below.

Figure 5.3(a): Dynamic Bioeconomic Model Estimates for the West Coast Hake Fishery: Biomass







To put matters differently, the dynamic bioeconomic model calculates that the average dynamic rent maximising biomass of 878 765 tons is approximately 40 percent greater than the average discounted maximum sustainable yield biomass of 621 782 tons.[25] In turn, the average maximum economic yield biomass produces a sustainable catch of 125 714 tons per annum, which is 3.96 percent less than the average discounted maximum sustainable yield of 130 893 tons per annum. However, the average level of effort of 10 305 standard-boat-days per annum required to take the dynamic rent maximising catch is approximately 30 percent less than the 15 074 standard-boat-days required to take the dynamic sustainable yield. In short, the modelling procedure calculates that shifting the west coast hake fishery from the dynamic biological optimum to the dynamic bioeconomic optimum produces a lower sustainable catch but, due to the higher biomass levels, requires substantially lower levels of fishing effort which, in turn, precipitates significant gains in economic efficiency.[26]

Specifically, catch per unit effort increases from an average 8.68 tons per standard-boat-day under the dynamic maximum sustainable yield scenario to an average 12.20 tons per standard-boat-day under the dynamic rent maximising scenario, an increase of more than 40 percent in fishing efficiency. It is interesting to note that between 1990 and 1995, catch per unit effort in the west coast hake fishery averaged 7.21 tons per standard-boat-day, and that a catch per unit effort ratio in excess of 12 tons per standard-boat-day has not been observed in the west coast hake fishery since 1964. The welfare implications of shifting the fishery from the dynamic yield

[25] Average values are calculated as the arithmetic mean of the output values generated by the dynamic bioeconomic model for each of the five parameter (input) scenarios, as summarised in Tables 5.4 and 5.5.

[26] As an aside, the dynamic rent maximising biomass for the five scenarios is, on average, 23.03 percent greater than the static maximum sustainable yield biomass. In other words, the discounted rent maximising strategy would be considered relatively conservative, as compared to a static yield maximising strategy. However, as the discount rate rises, the dynamic rent maximising biomass falls below the static yield maximising biomass, resulting in 'biological over-fishing' at discount rates of between 21.5 and 40.5 percent (depending upon the parameter scenario). Furthermore, as noted in Chapter 3, in the extreme, high discount rates render extinction of the resource the optimal harvest strategy.[28] That said, depending on the values taken by the biological parameters, a discount rate of between 4.93×10^{16} and 7.72×10^{16} percent per annum is required to render extinction of the west coast hake stock optimal under a dynamic rent maximising strategy. It is difficult to imagine circumstance that would justify a discount rate of this order. In any event, harvesting the resource to extinction is likely to be practically unfeasible (and, perhaps, impossible).

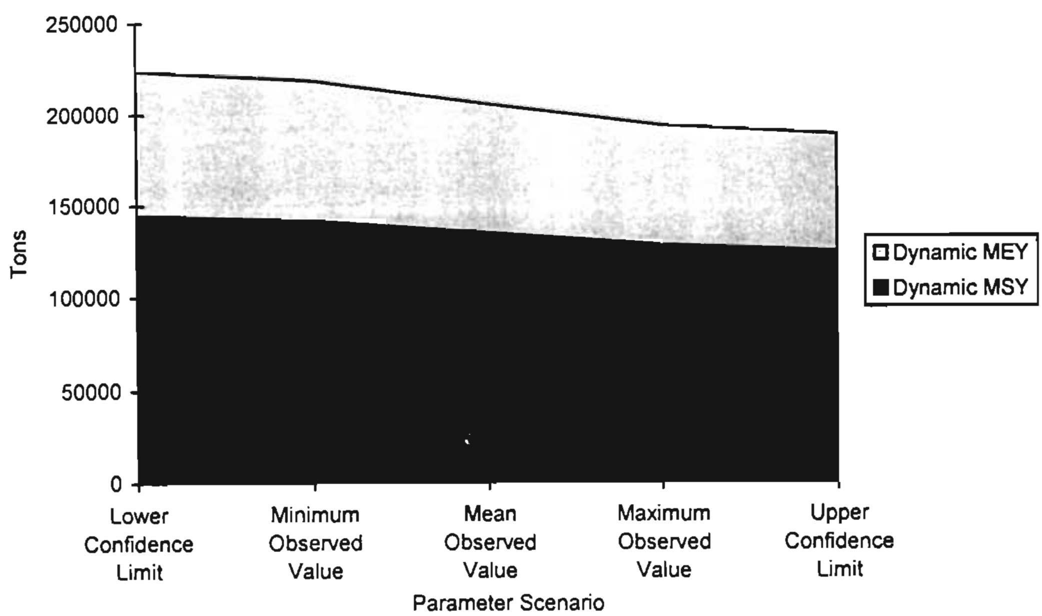
maximising position to the dynamic rent maximising position are considered more fully in Section 5.5.

5.4.2 The South Coast Fishery: Dynamic Maximum Economic Yield versus Dynamic Maximum Sustainable Yield

In the case of the south coast hake fishery, the dynamic bioeconomic model calculates the dynamic rent maximising biomass ranges between 186 846 tons and 223 844 tons depending on the values taken by the biological parameters, as opposed to the dynamic yield maximising biomass which ranges between 123 265 tons and 144 727 tons. In turn, the catch associated with the dynamic bioeconomic optimum ranges between 40 998 tons and 44 075 tons per annum which is, on average, four-fifths the size of the dynamic maximum sustainable yield, which ranges between 50 618 tons and 53 770 tons per annum. However, the level of effort required to take the dynamic maximum economic yield is just one-half of that required to take the dynamic maximum sustainable yield, which is of the order of 50 876 - 53 261 standard-boat-hours, as opposed to the 97 153 - 99 291 standard-boat-hours required to take the dynamic maximum sustainable yield. These results are summarised in Figure 5.4 below.

To put matters slightly differently, for the range of biological parameters employed, the model calculates that the average of dynamic rent maximising biomass in the south coast fishery of 205 169 tons is approximately 50 percent greater than the average discounted maximum sustainable yield biomass of 134 146 tons. In turn, the average dynamic maximum economic yield biomass produces a sustainable catch of 42 708 tons per annum, which is four-fifths the size of the discounted maximum sustainable yield of 52 478 tons per annum. However, the level of effort required to take the average rent maximising catch is approximately one-half the 98 263 standard-boat-hours required to take the average dynamic sustainable yield. In short, the modelling procedure calculates that shifting the south coast hake fishery from the dynamic biological optimum to the dynamic bioeconomic optimum produces a lower sustainable catch but, due to the higher biomass levels, requires considerably lower levels of fishing effort which, in turn, precipitates substantial gains in economic efficiency.[27]

Figure 5.4(a): Dynamic Bioeconomic Model Estimates for the South Coast Hake Fishery: Biomass



[27] As an aside, the average dynamic rent maximising biomass for the five scenarios of 205 169 tons is approximately 45 percent greater than the average static maximum sustainable yield biomass of 142 943 tons. Accordingly, the discounted rent maximising strategy would be considered conservative, as compared to a static yield maximising strategy. However, as the discount rate rises, the dynamic rent maximising biomass falls below the static maximum sustainable yield biomass, rendering 'biological over-fishing' the optimal harvesting strategy at discount rates of 350-450 percent per annum (depending on the biological parameter scenario). Furthermore, with sufficiently high discount rates, the dynamic rent maximising strategy involves extinction of the resource. In the case of the south coast fishery, a discount rate of the order of between 1.05×10^{17} and 2.81×10^{17} percent per annum is required to render extinction optimal. As in the case of the west coast, it is difficult to think of a set of circumstances that would justify a discount rate of this order.

Figure 5.4(b): Dynamic Bioeconomic Model Estimates for the South Coast Hake Fishery: Catch

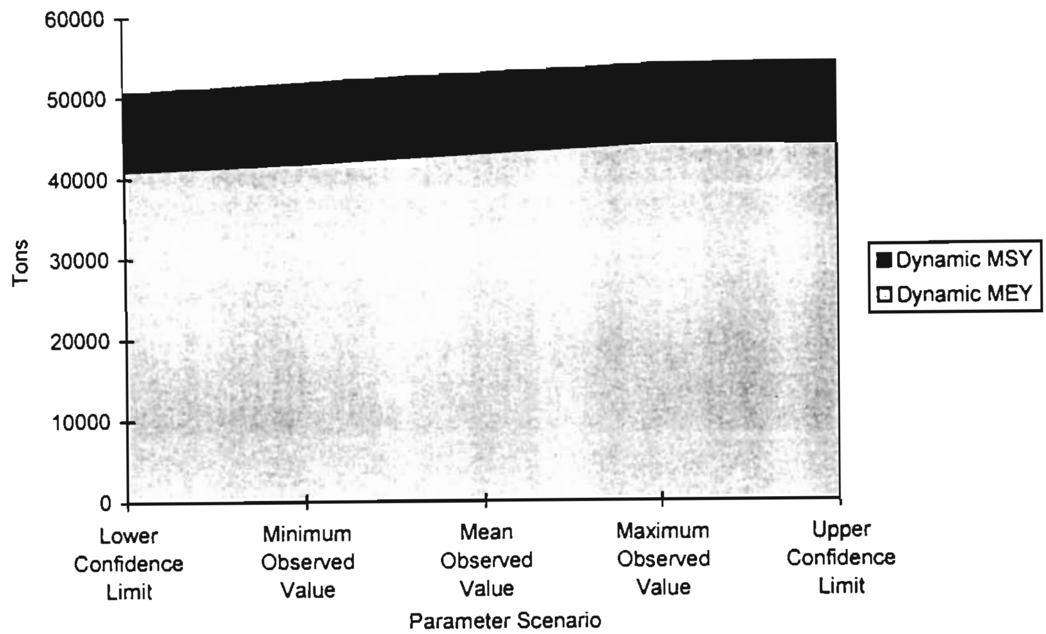
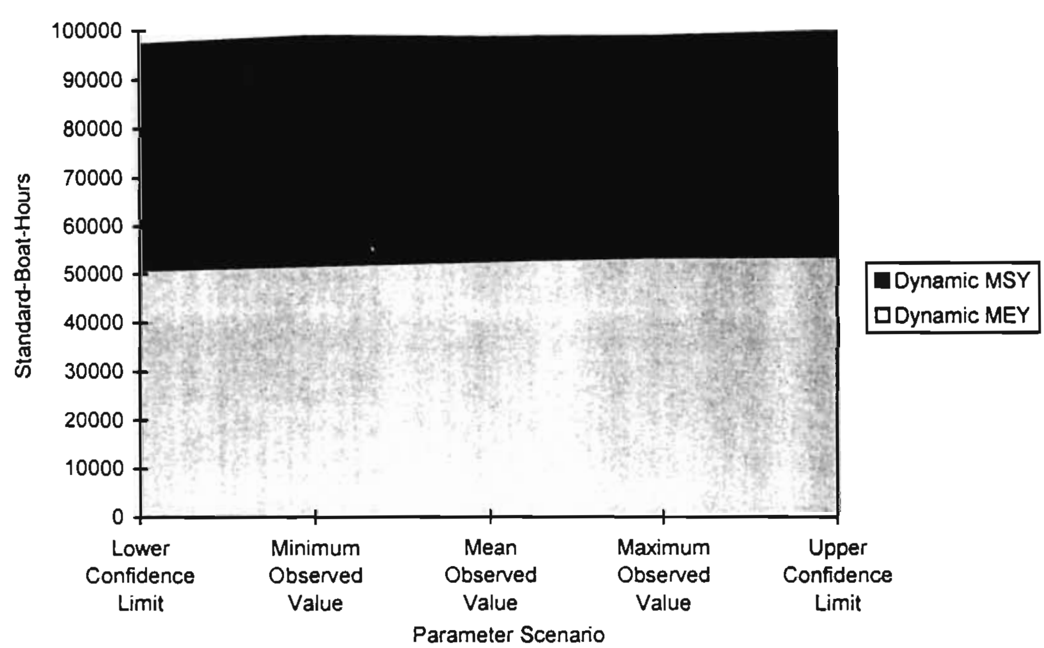


Figure 5.4(c): Dynamic Bioeconomic Model Estimates for the South Coast Hake Fishery: Effort



Specifically, catch per unit effort in the south coast fishery increases from an average 0.53 tons per standard-boat-hour under the dynamic maximum sustainable yield scenario to an average 0.82 tons per standard-boat-hour under the dynamic rent maximising scenario. This represents an increase of almost 55 percent in fishing efficiency in the south coast hake fishery. It is worthwhile noting that between 1990 and 1995, catch per unit effort in the south coast hake fishery averaged 0.68 tons per standard-boat-hour, and that a catch per unit effort ratio in excess of 0.8 tons per standard-boat-day has not been observed in the south coast hake fishery since 1971. The welfare implications of shifting the south coast hake fishery from the dynamic yield maximising position to the dynamic rent maximising position are considered more fully in Section 5.5.

5.5 Welfare Implications Under the Dynamic Bioeconomic Model

Having noted the implications for biomass, catch and effort of adopting a dynamic yield or rent maximising strategy, the problem then becomes a matter of establishing the welfare implications of moving the fishery from an optimal biological outcome (dynamic maximum sustainable yield) to an optimal bioeconomic outcome (dynamic maximum economic yield).[28] In this regard, in line with the theory presented in Chapter 3, it is accepted, *a priori*, that a shift from the dynamic maximum sustainable yield to the dynamic maximum economic yield position, by definition, involves an improvement in social welfare. The concern of this section is with attempting to quantify the extent of the net gain in economic rents, that is, total social welfare.

At the outset, it should be recalled that, in terms of the dynamic surplus production model, social welfare is defined as economic rent. In other words, social welfare is measured as the surplus of revenue over economic costs. Accordingly, the task of measuring social welfare requires the conversion of catch and effort data into revenue and cost data. This is achieved by re-

[28] Of course, in reality, the matter revolves around driving the fishery from its extant position (which, depending on existing policy, may or may not coincide with the dynamic maximum sustainable yield outcome) to the dynamic maximum economic yield outcome. Chapter 7 deals with this issue in greater detail.

employing the price-per-ton:cost-per-standard-boat-day ratios as derived in Section 5.3 above. There, the cost:price ratio in the west coast fishery was argued to be of the order of 1.00:5.85, where the unit of measurement is the landed price per ton of hake. By multiplying catch and effort data by the price and cost figures of 1.00 and 5.85 respectively, it is possible to arrive at total revenue and economic cost figures for any given scenario (for example, dynamic maximum sustainable yield or dynamic maximum economic yield). The difference between these two values is the size of the economic rent generated by the fishery under the given scenario. Similarly, multiplying catch and effort data for the south coast fishery by 1.00 and 0.52 respectively, generates revenue and economic cost data for that fishery which allows fo the calculation of the size of the economic rent under any scenario for the south coast hake fishery.

The full calculations required to arrive at revenue and economic cost figures for the west and south coast fishery under the assumption of dynamic maximum sustainable yield and dynamic maximum economic yield strategies for each of the five biological parameter scenarios are set out in Appendix 3. Table 5.6 summarises the results and shows the net gain in total social welfare achieved by shifting the fishery from the optimal biological to optimal bioeconomic outcome under the dynamic surplus production model. Before examining the results, it should be emphasised again that, in the absence of accurate price and cost data, the figures quoted in Table 5.6 should not be read as monetary values, but rather as relative values. This has no influence on the significance of the results.

The outcome of the above results is unambiguous: shifting the west and south coast fisheries from a dynamic maximum sustainable yield – or biological – outcome to a dynamic maximum economic yield – or bioeconomic – outcome would generate considerable gains in net social welfare. In the case of the west coast hake fishery, such a shift would yield an increase in net social welfare of between 39.16 and 96.93 percent, depending on the biological parameter scenario. The average gain in social welfare over all five scenarios in the west coast fishery is 57.34 percent. In the case of the south coast hake fishery, shifting the fishery from the dynamic biological optimum to the dynamic bioeconomic optimum would produce a gain in social welfare of between 516.07 and 14 672.94 percent, depending on the biological parameter scenario. The average gain in social welfare over all five scenarios in the south coast fishery is 4 362.30 percent.

Table 5.6: Economic Rent under Dynamic Maximum Sustainable Yield and Dynamic Maximum Economic Yield Strategies in the West and South Coast Fisheries

	Dynamic MSY (1)	Dynamic MEY (2)	Percentage Increase of (2) over (1)
Parameter scenario	West Coast Fishery: Economic Rent (Index Points)		
Lower Confidence Limit	27 491.31	54 140.01	96.93
Minimum Observed Value	39 902.30	64 103.90	60.65
Mean	48 724.30	70 502.75	44.70
Maximum Observed Value	52 060.20	72 444.75	39.16
Upper Confidence Limit	45 382.45	65 929.05	45.27
Sample Average	42 712.11	65 424.09	57.34
	South Coast Fishery: Economic Rent (Index Points)		
Lower Confidence Limit	98.44	14 542.48	14 672.94
Minimum Observed Value	282.52	14 821.56	5 146.20
Mean	1 651.24	15 693.32	850.40
Maximum Observed Value	2 658.68	16 379.28	516.07
Upper Confidence Limit	2 215.32	16 080.96	625.90
Sample Average	1 381.24	15 503.52	4 362.30
Combined Average	44 093.35	80 927.61	83.54

In short, shifting the fisheries from a dynamic maximum sustainable yield to dynamic maximum economic yield outcome would generate considerable gains in social welfare. Perhaps, the best picture of the extent of potential gain in social welfare is given by summing the figures for both fisheries. This is done in the last line of Table 5.6, which shows the combined average rent for the west and south coast hake fisheries. From that result, it can be concluded that shifting the fishery from the dynamic maximum sustainable yield to the dynamic maximum economic yield outcome would approximately double the rent generated by the hake resource.

5.6 Implications: Bioeconomic versus Biological Management

The implications of the above results for management are clear-cut: shifting management of the fishery from a biological to bioeconomic basis will, without exception, produce improvements in social welfare. Furthermore, the potential gains in social welfare are considerable. By the same token, the implications for the fishery are equally notable.

Shifting the west coast hake fishery from the dynamic maximum sustainable yield position to the dynamic rent maximising position involves a reduction in catch of the order of 4 percent. At the same time, however, the biomass under the dynamic rent maximising strategy is substantially (on average 41 percent) higher than the dynamic yield maximising biomass. The effect of this is that the level of effort required to take the dynamic rent maximising catch is a little more than two-thirds that required to take the dynamic maximum sustainable yield. The net result of the fall in effort being more than offset by the fall in catch is an increase in rents (and therefore social welfare). Indeed, under the dynamic rent maximising strategy rents are of the order of one-and-a-half times the size of rents generated under the dynamic sustainable yield strategy.

Shifting the south coast hake fishery from the dynamic maximum sustainable yield position to the dynamic rent maximising position involves a reduction in catch of the order of 20 percent. At the same time, however, the biomass under the dynamic rent maximising strategy is, on average, more than 50 percent greater than the dynamic yield maximising biomass. As a result, the level of effort required to take the dynamic rent maximising catch is approximately half that required to take the dynamic maximum sustainable catch. The net effect of the fall in effort being more than offset by the fall in catch is a substantial increase in rents (and therefore social welfare) generated by the fishery. To be more specific, under the dynamic rent maximising strategy rents are, on average, forty times the size of rents generated under the dynamic sustainable yield strategy.

It is evident that shifting the west and south coast fisheries from the dynamic maximum sustainable yield position to the dynamic maximum economic yield position produces considerable increases in rent. However, these increases in rents are achieved on the back of (in instances considerable) adjustments to the level of catch (biomass) and effort in the fishery. In turn, the changes in catch and effort levels required to achieve higher rents implies alterations to the structure of the fishery, especially in terms of the rates of employment of labour and capital. This issue, however, is complex, and fuller comment is reserved for Chapter 7 where the implications of adopting bioeconomic, as opposed to biological, management principles for the industrial structure of the fishery are considered in closer detail.

At this point, however, several caveats should be noted with regard to the implications, as set out above, of adopting bioeconomic management principles. First, the above arguments are based on the assumption that the fishery is at the dynamic maximum sustainable yield outcome: this is unlikely given that South Africa's fisheries managers have adopted a management strategy different to the yield maximising strategy, namely the $f_{0.2}$ strategy.[29] Accordingly, the welfare implications – and implications for effort and catch in

[29] As noted in Chapter 4, the extant 'biologically conservative' $f_{0.2}$ management strategy involves driving the fishery to the so-called $B_{0.2}$ biomass, equal to approximately 1 000 000 tons on the west coast and 165 000 tons on the south coast.

the fishery – are likely to be different to those spelled out above. The implications of this argument are explored more fully in Chapter 7.

Furthermore, if the above model is to serve as the basis for policy formulation, it is vital, from a management perspective, that the sensitivity of the model to fluctuations in biological and economic parameters is first considered. The reasons behind this are readily apparent. Given the difficulties involved in arriving at accurate estimates of biological parameters, and evidence to suggest that parameter values change across time, it is vital that fisheries managers have a solid grasp of the nature of the relationship between biological parameter values and the dynamic rent maximising outcome. Similarly, estimation problems exist in the case of economic parameter values, and there is also evidence to suggest that the assumptions of autonomy in prices, costs and the quasi-discount rate are 'unrealistic' in the context of the hake fishery.

Given these concerns, Chapter 6 turns to consider the sensitivity of the dynamic rent maximising outcome (and the bioeconomic model more generally) to changes in biological and economic parameter values, and explores the implications for management and policy formulation.

CHAPTER 6: EXTENDING THE DYNAMIC BIOECONOMIC MODEL INTO A VARIABLE PARAMETER SETTING: THEORY AND EVIDENCE

6.1 INTRODUCTION

A primary assumption of the dynamic surplus production model, as presented in Chapters 3 and 5, is that the value of each of the economic and biological parameters is constant across time, that is, autonomous. There is evidence to suggest that this assumption is flawed. Furthermore, as outlined in Chapter 5, there are considerable difficulties involved in arriving at accurate and unbiased estimates of biological and economic parameter values. However, at least from a management perspective, it is vital that the results produced by the dynamic bioeconomic model are robust, that is, relatively insensitive to changes in parameter values or, if not robust, that managers have a solid grasp of the relationship between parameter values and model estimates. The reasons for this are four-fold. First, if the model is insensitive to changes in parameter values, the problems involved in generating accurate estimates of parameter values, as outlined in Chapter 5, become relatively insignificant. Second, if managers are to adopt goals which require considerable adjustments to biomass, catch and effort levels which, in turn, impact upon, *inter alia*, levels of investment and employment in the fishery, then it is desirable that policy targets are 'stable' or, at least, 'predictable'. Third, fluctuating policy targets require on-going adjustments to employment and investment levels; elsewhere in this study (Chapter 7), it is argued that such an approach is likely to undermine the goals of economic efficiency set by the principle of bioeconomic management. Fourth, whilst model stability is desirable, there is no way of ensuring that output estimates are robust. Indeed, if model estimates are found to be sensitive to changes in input values, it becomes vital that managers should be capable of providing stability through their knowledge of the nature and magnitude of input-output relationships.

Accordingly, the primary objective of this chapter is to provide an analysis of the sensitivity of the dynamic bioeconomic model to variations in biological and economic parameter values. Section 6.2 gives consideration to the evidence regarding the variability of biological and economic parameter values; and Section 6.3 is devoted to extending the dynamic bioeconomic model into a variable parameter setting. In turn, Sections 6.4 and 6.5 are concerned with establishing the relationship between variable input values and output values; and the implications of variable parameter values for social welfare levels are also given consideration.

6.2 Variability of Parameter Values: A Review of the Evidence

6.2.1 Variable Biological Parameters: Assessing the Evidence

Given the problems (as set out in detail in Chapter 5) involved in arriving at accurate and unbiased estimates of biological parameters, it is argued that it is more appropriate, at least from a modelling perspective, to assume range values, as opposed to point values, for these parameter values. However, whilst perhaps considered sound from a modelling perspective, the adoption of range values is problematic from a management perspective, as variations in input values result in variable output values. More to the point, if the model is particularly sensitive to variations in input values, the output values – or management targets – are likely to be vague and 'unreliable'. Accordingly, it is desirable that model estimates should be reasonably robust.

In addition to estimation problems, a second reason exists for testing the sensitivity of the model to changes in biological parameters. It has been suggested by some writers that biological parameter values may vary across time. For example, a temporal trend in the estimates of the biological parameters r and K was noted by Punt (1989) who showed that, for the west coast hake stock, the trend was toward falling estimates of r and rising estimates of K across time. Later studies have confirmed this result (Punt 1990, 1991 & 1992a; Punt and Leslie, 1991; Leslie 1993 & 1996). Furthermore, there is evidence to show that the opposite temporal trend exists in the south coast fishery, that is, rising estimates of r and falling estimates of K across time (Punt 1991 & 1992a; Punt and Leslie, 1991; Leslie, 1993 & 1996). Over the period 1984-95, the temporal trends in growth rates and biomass have been particularly strong on the west coast, with r_w falling by an average 3.34 percent per annum and K_w rising by an average 3.08 percent per annum. Trends on the south coast are considerably weaker, with r_s rising by an average 0.52 percent per annum and K_s falling by an average 0.27 percent per annum. This evidence, drawn from Table 5.1 of Chapter 5, is reproduced in Table 6.1 below.

It has also been argued by some writers that the catchability coefficient, q, might vary across time. For example, Punt *et al* (1992) suggested that q may be rising across time as higher power factors (which have served to improve harvesting efficiencies) have increased catchability, although the trend has probably been partially (or even completely) offset by trawlers directing what is recorded as 'hake-directed-effort' at other species with increasing frequency. Evidence of temporal trends in q is drawn from Chapter 5 and reproduced in Table 6.1, where it is shown that the temporal trend in catchability has been particularly marked in the west coast, with q_w falling by an average 2.78 percent per annum over the period 1984-95. The trend in q_s has been somewhat weaker, with catchability on the south coast increasing by 0.55 percent per annum over the period 1984-94.

Table 6.1: Estimates Generated by the Schaefer Form of the Dynamic Surplus Production Model for the West and South Coast Hake Stocks for the Parameters r, K and q

	West Coast				South Coast		
	r_w	K_w	$q_w \times 10^{-3}$		r_s	K_s	$q_s \times 10^{-3}$
1984	0.4643	1 226 000	0.0166	1984	0.7275	288 700	0.0039
1985	0.4480	1 262 000	0.0161	1985	0.7834	273 800	0.0042
1986	0.4806	1 192 000	0.0171	1986	0.8003	269 600	0.0043
1987	0.4085	1 357 000	0.0150	1987	0.7617	279 600	0.0041
1988	0.3871	1 415 000	0.0144	1988	0.7248	290 100	0.0039
1989	0.4383	1 282 000	0.0160	1989	0.6850	302 500	0.0037
1990	0.4061	1 362 000	0.0150	1990	0.6852	302 400	0.0037
1991	0.3557	1 510 000	0.0134	1991	0.6955	299 100	0.0037
1992	0.3417	1 557 000	0.0129	1992	0.7289	288 700	0.0039
1993	0.3269	1 610 000	0.0125	1993	0.7578	280 100	0.0041
1994	0.3191	1 639 000	0.0122	1994	0.7724	276 000	0.0042
1995	0.3092	1 678 000	0.0119	1995	0.7630	278 700	0.0041
Mean	0.3905	1 424 167	0.0144		0.7405	285 775	0.0040
Minimum	0.3092	1 192 000	0.0119		0.6850	269 600	0.0037
Maximum	0.4806	1 678 000	0.0171		0.8003	302 500	0.0043
Standard Deviation	0.0596	170 099	0.0018		0.0386	11 243	0.0002
Coefficient of Variation (%)	15.26	11.94	12.50		5.21	3.93	5.34

Source: Adapted from pers. comm. (Leslie, Sea Fisheries Research Institute).

In short, given the problems involved in arriving at accurate estimates of r, K and q, it follows that it is vital – at least from a management perspective - that the model estimates be tested for sensitivity to fluctuations in values of the biological parameters. It should be noted here that because range values, as opposed to point values, are employed as the basis for biological parameter estimates, it follows that the issue of the impact of variable biological parameter values has already been touched on (in Chapter 5). However, more than simply testing the sensitivity of model estimates to fluctuations in biological parameter values, the task in this chapter shifts to one of extending the model into a variable parameter setting and, in so doing, providing a more detailed consideration of the impact of variable biological parameter values on the dynamic bioeconomic outcome. These tasks are undertaken in Section 6.3.

6.2.2 Variable Economic Parameters: Assessing the Evidence

There are at least two reasons for testing the sensitivity of the dynamic bioeconomic model to fluctuations in economic parameter values. As noted in Chapter 5, in the absence of accurate data, the price and cost estimates are represented by an operating profit margin which is argued to be a satisfactory proxy for the 'true' price:cost difference. However, while every effort is made to ensure that the proxy provides a 'reasonable guide' to operating profit margins, there is no basis for assuming that the figure of 20 percent employed in the modelling procedure is entirely accurate. Similarly, while the estimation procedure employs a discount rate of 4.5 percent, the figure is adopted by virtue of the fact that it is the mean of the proposed feasible range of 3-6 percent. Thus the figure of 4.5 percent is somewhat arbitrarily selected, and there is no reason to suppose that some other rate in the 3-6 percent range, say 3.5 percent or 5.5 percent, would not provide a more accurate reflection of the expected trend of utility for the hake resource under a projected steady trend of circumstances.

In light of the difficulties involved in arriving at estimates of the economic parameters, p, q and δ, it is important that the sensitivity of model estimates to fluctuations in economic parameter values should also be examined in closer detail. However, a more fundamental reason exists for examining the relationship between economic parameters and model estimates. Gordon's (1954) model makes a number of key assumptions about the value of economic parameters: the price of the output of the fishery – in this instance hake – is autonomous (or fixed); the unit (or marginal) costs of fishing (trawling) are constant; and the discount rate is invariable across time. It is argued below that these assumptions are unrealistic, with the implication being that even if it were possible to arrive at accurate estimates of the economic parameter values, these values are expected to change across time.

Turning first to examine the issue of prices and costs, studies which have attempted to model fisheries along bioeconomic lines have, in the main, retained the assumptions of autonomous prices and constant costs.[1] For example, Copes (1970, 69) noted:

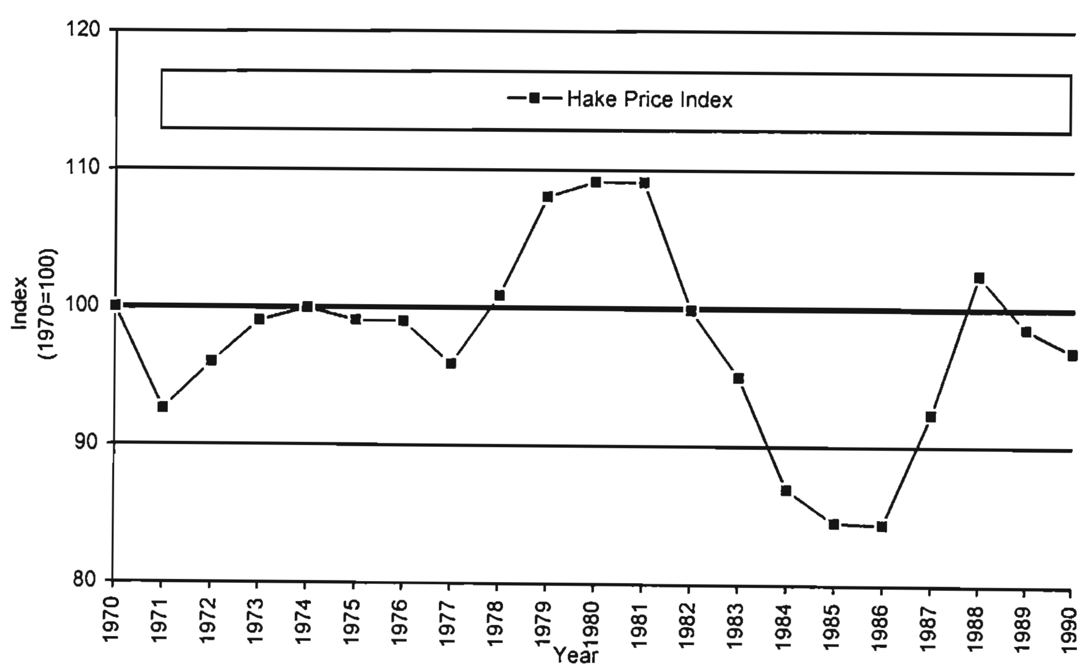
> The literature stresses, on the supply side, the relationship of output to the amount of fishing effort – the latter, in turn, providing a link with cost. But the consequences of these and other relationships in terms of conventional supply and demand analysis – as may be traced in simple price/output graphs – do not appear to have been treated exhaustively.

The case of South African commercial fisheries is no exception. Indeed, while the issue of bioeconomic modelling has received very little attention, the matter of extending the dynamic bioeconomic model into a variable price and/or cost setting has gone virtually unconsidered. For example, in modelling the west coast hake fishery along bioeconomic lines, Andrew and Butterworth (1987, 931) assumed that, in real terms, prices and costs remain

[1] See, for example, Bell (1972), Agnello and Donnelley (1975), Anderson (1976), Clark (1976), Henderson and Tugwell (1979), Morey (1980), Gallastegui (1983), Munro and Scott (1985), Townsend (1986), Bjørndal (1988), Squires (1988), Muraoka (1990), Conrad and Bjørndal (1991), Bjørndal and Salvanes (1993) and Opsomer and Conrad (1994). A limited number of writers have attempted to extend the modelling and estimation procedures to allow for variable prices and/or variable costs, for example, Smith (1968), Levhari *et al* (1981) and Anderson (1985a).

unchanged with time.[2] They suggested that, in the context of the South African hake fishery, the assumption of autonomous prices is reasonable because short-term price fluctuations tend to 'average out' over the medium term (five to ten years). This view is not fundamentally wrong. Indeed, price data show that, in real terms, hake prices have tended to remain constant over the medium to long term. Figure 6.1 shows trends (or, rather, changes) in the real price of hake (in the form of a real price index) over the period 1970-90.

Figure 6.1: Trends in the Real Price of Hake (1970-90)
(1970=100)



Source: Central Statistical Services (1993).

The above evidence leads to two main conclusions. First, in support of Andrew and Butterworth (1987), it would appear that, over the longer term, price fluctuations 'tend to average out'. This result is supported by the fact that the hake price index tends towards the consumer price index base of 100 (which represents constant real hake prices) on a number of occasions

[2] An exhaustive review of the fisheries literature reveals only one attempt – namely that of Andrew and Butterworth (1987) – to model fish stocks in South Africa on the basis of bioeconomic principles.

(1973-76, 1978, 1982 & 1989) over the period 1970-90.[3] The strength of this point is reinforced by the extremely high correlation coefficient between the consumer price index and hake price index of 0.9926 for the period 1970-90. Accordingly, it is accepted that it is reasonable to suggest that, over the longer term, the real price of hake is constant. Second, however, this conclusion does not hold over the short term. There are periods during which hake prices deviate – often by a considerable margin – from the long-term average. For example, over the five-year period 1978-82, the price of hake exceeded the constant real price by an average 10 percent; and over the period 1982-88, the price of hake fell short of the constant real price by as much as 18 percent. To put the matter differently, over the twelve years 1978-89, although the real price returned to approximately the long-term average on three occasions (1978, 1982 & 1988), for the remainder of the period the price of hake was either greater than or less than the constant (1970) real price. This conclusion is supported by more recent evidence (*Cape Business News*, 1997a), which shows that landed prices in the trawl sector climbed by 20 percent over 1995 (with the landed price of hake rising to R3 000 per ton).

With regard to costs, while accurate cost data for the hake trawl fishery are unavailable, there is sufficiently broad evidence to suggest that costs are not autonomous. Evidence of non-autonomy in costs is drawn from two sources. First, Table 6.2 below reproduces the operating margins for the two companies – Sea Harvest and Namfish – which are argued, in Chapter 5, to provide good proxies for operating margins in the hake trawl fishery. However, the margins have been revised, using producer price indices, to reflect a fixed real price scenario. In other words, the fluctuations in the margins presented below are argued to be primarily due to changes in cost and not changes in price.[4]

[3] 1970 is selected as the base year as this is the first year for which accurate figures are available. It is worth noting that any other year could have been chosen and would have served equally well as a base year.

[4] The technique of simply rebasing margins to reflect price changes is accepted as being crude and simplistic as it ignores, for example, changes in cost or price structures that might arise out of different volumes harvested or sold in any one year. It also ignores the differences that exist between the companies' financial year and the calendar year. However,

Table 6.2: Operating Profit Margins of Fishing Companies Listed on the Johannesburg Stock Exchange (1984-95) for Variable and Constant Real Prices

	Variable Real Price Scenario		Constant Real Price Scenario	
	Namfish	Sea Harvest [5]	Namfish	Sea Harvest
1984	25.77	-	25.77	-
1985	43.78	-	47.35	-
1986	39.40	-	45.00	-
1987	38.00	-	41.49	-
1988	34.23	-	32.50	-
1989	28.09	22.90	27.61	22.51
1990	21.52	22.00	21.02	21.49
1991	15.09	24.20	14.02	22.48
1992	-4.10	28.80	-3.54	24.88
1993	6.66	18.90	5.62	15.95
1994	8.85	17.49	7.63	15.09
1995	1.04	20.77	0.90	18.00
Mean	21.53	22.15	22.11	20.06
Minimum	-4.10	17.49	-3.54	15.09
Maximum	43.78	28.80	47.35	24.88
Range	47.88	11.31	50.89	9.79
Standard Deviation	15.98	3.72	17.44	3.72
Coefficient of Variation (%)	74.22	16.81	78.88	18.54

Source: Adapted from McGregor (1988, 1992 & 1996).

Before commenting on the results of the recalculation of operating profit margins, it should be noted that these revised estimates suffer from the same weaknesses as the profit margin figures cited in Chapter 5. Nevertheless, the object of the exercise is not to provide accurate estimates of the impact of fluctuations in cost levels on operating profit margins. Rather, the aim is to provide evidence to suggest that, just as it is unreasonable to assume that

the differences in margins between years are accepted as being sufficiently large to allow for the conclusion that changes in costs must at least play some part in determining operating profit margins in the hake trawl fishery.

[5] Sea Harvest was first listed in 1994; data are available from 1989 onwards.

prices are autonomous, so it is unreasonable to assume that costs are autonomous; and this argument is lent strong support by the above evidence. More to the point, in controlling for price fluctuations, the margins for both companies are found to fluctuate across time. Indeed, in controlling for price variations, margins are found to fluctuate more widely than in the case of variable real prices. This is evidenced by higher coefficients of variation for both companies' profit margins under the constant price scenario compared to the variable price scenario.

Further evidence to suggest that costs fluctuate across time is provided by data on input costs in the fishing industry (Industrial Development Corporation, 1992; Central Statistical Services, 1993). In this regard, while the data are not specific to the hake fishery, the evidence is sufficient to suggest that input costs across the entire fishing industry – including the hake fishery – fluctuate across time. Figure 6.2 shows the trends in input costs in the fishing industry over the period 1972-90 versus the fishing industry producer price index over the same period.

The evidence presented above is unambiguous: relative to output prices, costs – and therefore operating profit margins – have not remained constant. Rather, without exception, costs have varied from year to year, often deviating considerably (for example, 1978-79 and 1985-87) from real prices.[6] Given that the cost level is a determinant of the dynamic outcome, it is argued here that a necessary aspect of modelling the hake resource involves considering the impact of variable costs on the dynamic rent maximising outcome in the hake fishery and the implications for welfare.

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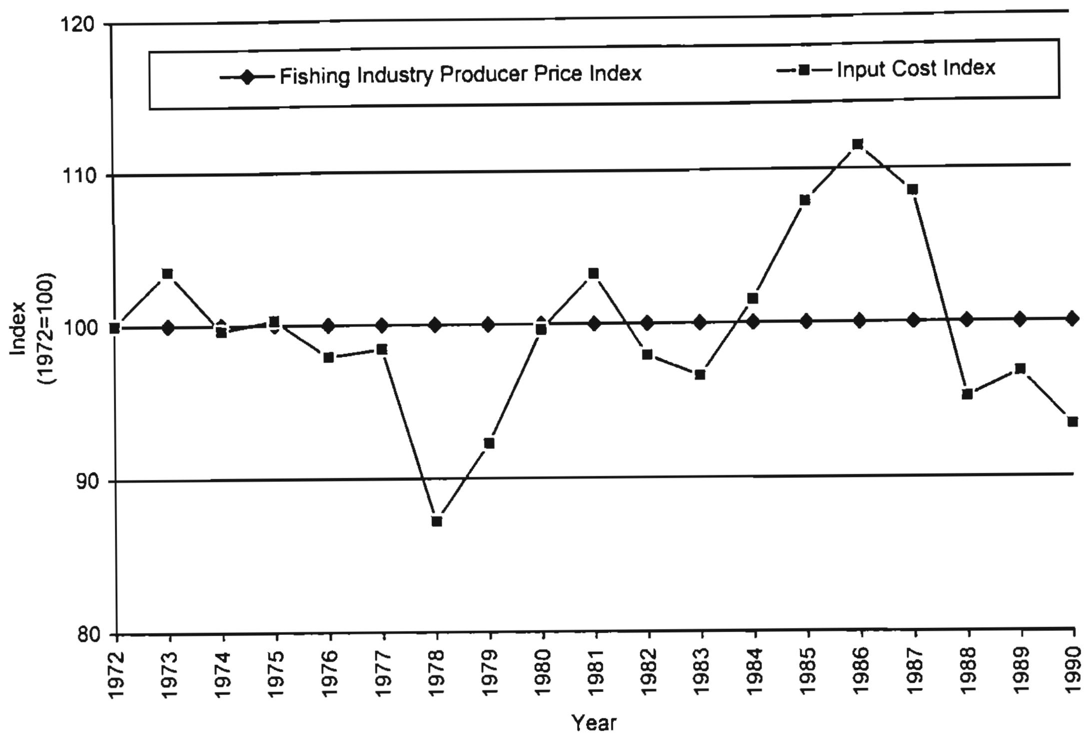
[6] It should be noted that the fluctuation of costs across time does not imply that the cost function is no longer linear. Rather, the implication is that the slope of the linear cost function changes from year to year. The point is important because, although extending the model by relaxing the linearity assumption is, mathematically, a straightforward task (Munro and Scott, 1985, 642-3; Conrad and Bjørndal, 1991, 167), estimation of the non-linear cost model is a problem of considerable proportions in the South African context, where accurate, detailed data on costs structures of the hake fishery (or, for that matter, any of the other major commercial fisheries) are generally not available. Moreover, as argued previously, given that hakes do not group like pelagics, for example, catchability tends to remain constant (Pitcher and Alheit, 1995, 12), which is suggestive of a linear cost function, rather than a non-linear cost function.

Figure 6.2: Trends in Fishing Industry Input Costs Relative to the Fishing Industry Producer Price Index (1972-90)



Source: Adapted from Industrial Development Corporation (1992); Central Statistical Services (1993).

Finally, with regard to the discount rate, there are at least two reasons for expecting the quasi-discount rate to fluctuate across time. First, given that the quasi-discount rate reflects the expected trend of the utility of a resource under an assumed steady trend of circumstances, it follows that any changes in the expected trend in utility will result in changes in the quasi-discount rate. At the same time, it is reasonable to expect changes in the expected trend in utility to occur, with the most obvious causes being unexpected or random changes in technology, incomes or tastes (Lewis and Neher, 1982). Furthermore, changes in growth rates in the resource under consideration would also precipitate fluctuations in projected trends in utility. It was argued above that there is sufficient evidence to conclude that, at least in the instance of hake stocks, growth rates fluctuate across time, with the model parameter r falling in the west coast fishery and rising in the south coast fishery (Punt 1991 & 1992a; Punt and Leslie, 1991; Leslie, 1993 & 1996). The upshot of the above is that, as with cost and price estimates, the value of the discount rate parameter varies (subject to a trend) across time. Accordingly, an important aspect of the modelling and estimation procedure involves extending the dynamic bioeconomic model into a variable discount rate setting.[7]

6.3 Extending the Dynamic Bioeconomic Model into a Variable Parameter Setting

In extending the dynamic bioeconomic model to allow for variable parameters, it should be noted that a number of assumptions are retained (Copes 1970 & 1972; Clark 1976 & 1985). Specifically, it is assumed that the fishery is a single-species fishery with an unchanging matrix of natural conditions and biological relationships. The assumption that the fishery is characterised by fixed factor portions is also retained. However, by allowing the catchability coefficient to vary, it follows, by implication, that technology is no longer assumed to be constant. As in the model developed in Chapter 5, externalities are not considered, except where they follow directly from bioeconomic interaction through variation in the level of fishing effort. Finally, the assumption that the demand for fish is perfectly elastic is retained.[8]

Extending the dynamic bioeconomic model to allow for non-autonomy in parameters requires returning to the dynamic bioeconomic model developed in Chapters 3 and 5. There, it was noted that the size of the dynamic rent maximising fish stock (X^*_{MEY}) is determined by the fundamental equation:

$$G'(X^*_{MEY}) - \frac{c'(X^*_{MEY})G(X^*_{MEY})}{p - c(X^*_{MEY})} = \delta; \qquad \text{(Equation 3.18)}$$

[7] This result is by no means novel; indeed, some writers suggest that variable discount rates should serve as the starting point in resource modelling (Weitzman, 1994).
[8] This assumption is not unreasonable given the relative availability of substitutes for hake and given that hake accounts for a considerable portion of income spent on fish.

and that in the event that $G(X)$ is the Schaefer logistic function, it is possible to restate Equation 3.18 as:

$$X^{*}_{MEY} = \frac{1}{4}\left\{\bar{X}_t + K(1-\frac{\delta}{r}) + \sqrt{\left[\left(\bar{X}_t + K\left(1-\frac{\delta}{r}\right)\right)^2 + \frac{8K\bar{X}_t\delta}{r}\right]}\right\}. \quad \text{(Equation 5.1)}$$

As it stands, Equation 5.1 provides a clear guide to the relationship between the dynamic rent maximising biomass (X^{*}_{MEY}) and the biological variables r and K, as well as δ (the quasi-discount rate). But, the equation provides no explicit guide to the relationship between the biological variable q, the economic variables p and c and the dynamic rent maximising outcome. However, given that $\bar{X}_t = \frac{c}{pq}$, it is possible to expand Equation 5.1 to explicitly include the catchability coefficient, price and cost as determining variables of the dynamic rent maximising biomass:

$$X^{*}_{MEY} = \frac{1}{4}\left\{\frac{c}{pq} + K(1-\frac{\delta}{r}) + \sqrt{\left[\left(\frac{c}{pq} + K\left(1-\frac{\delta}{r}\right)\right)^2 + \frac{8K\frac{c}{pq}\delta}{r}\right]}\right\}. \quad \text{(Equation 6.1)}^9$$

Based on this equation, it follows that by varying the value of the price, cost and discount rate parameters, it is possible to establish the dynamic rent maximising biomass for any parameter value. In addition, given the formulae for catch ($C_t = qE_tX_t$) and the associated effort level ($E_t = \frac{C_t}{qX_t}$) for any given biomass, it is possible to establish the nature of the relationship between the dynamic rent maximising catch and effort and each of the parameters. Furthermore, by employing the methodology set out in Chapter 5, it is possible to establish the impact of variable parameter values on

[9] For simplicity, the subscript on each of the biological and economic parameters, to reflect parameter fluctuations across time (t), is ignored.

welfare levels. The theoretical and empirical relationship between variable biological and economic parameters and the dynamic rent maximising outcome is reviewed in Sections 6.4 and 6.5. Consideration is also given to the implications of variable parameter values for social welfare levels and, in turn, for fisheries management.

6.4 The Relationship between Biological Parameters and the Dynamic Rent Maximising Biomass, Catch and Effort

The aim of this section is to provide a review of the nature of the relationship between each of the biological parameters and the dynamic rent maximising outcome. However, rather than examining the responsiveness of the outcome to variations on a parameter-by-parameter basis, the assumption of biological parameter 'scenarios' is retained. The reason for this is that, based on the evidence presented in Section 6.2.1, and as per the parameter scenarios presented in Chapter 5, the variations in biological parameters display clear relationships, with the value of r being negatively correlated with K and positively correlated with q.[10] The analysis of the impact of variable biological parameter values on the dynamic rent maximising outcome is made up of three parts. Section 6.4.1 provides a review of the nature of the relationship between the biological parameters and the dynamic rent maximising outcome. Section 6.4.2 is concerned with estimating the dynamic rent maximising outcome under variable biological parameter conditions. Finally, Section 6.4.3 considers the impact of variable biological parameter values on welfare levels under the dynamic bioeconomic model.

[10] No assumption is made with regard to economic parameter scenarios as there is no obvious basis for assuming the existence of relationships between economic parameters.

6.4.1 A Review of the Relationship between Biological Parameters and the Dynamic Rent Maximising Outcome

The extended model, as set out in Equation 6.1, provides for a number of results apropos the dynamic rent maximising biomass and biological parameter values. First, with regard to the intrinsic growth rate of the biomass, r, holding all other parameter values constant, from Equation 6.1 it follows for all $r \geq 0$:

$$\frac{dX^{*}_{MEY}}{dr} > 0 \text{ and } \frac{d^2 X^{*}_{MEY}}{dr^2} < 0;$$

with $\lim_{r \to \infty} X^{*}_{MEY} = X_{MEY}$ and $\lim_{r \to 0} X^{*}_{MEY} = \overline{X}_t$. That is to say, the dynamic rent maximising biomass increases with the intrinsic growth rate, although at a decreasing rate. At sufficiently high growth rates, the growth effect swamps the discounting effect, such that the dynamic rent maximising biomass asymptotically approaches the static maximum economic yield biomass. At the other extreme, the dynamic rent maximising biomass falls to equal the static zero-rent (bionomic) biomass, $\overline{X}_t$, for a growth rate of zero.

From the above, it is also possible to derive the dynamic rent maximising catch and effort levels associated with the dynamic rent maximising biomass under variable growth rates. Formally, given that:

$$C_t = qE_t X_t \qquad \text{(from Equation 3.3)}$$

it follows, from Equation 6.1, that for all $r \geq 0$:

$$\frac{dC^{*}_{MEY}}{dr} > 0 \text{ and } \frac{d^2 C^{*}_{MEY}}{dr^2} > 0;$$

with $\lim_{r\to\infty} C^*_{MEY} = +\infty$ and $\lim_{r\to 0} C^*_{MEY} = 0$. In other words, the dynamic rent maximising catch increases at an increasing rate with the intrinsic growth rate, approaching infinity as the growth rate approaches infinity. At the other extreme, under a zero growth scenario, the dynamic rent maximising catch falls to zero.

Finally, with regard to the dynamic rent maximising level of effort under variable growth rates, by rearranging Equation 3.3, such that:

$$E_t = \frac{C_t}{qX_t};$$

from Equation 6.1, it follows, for all $r \geq 0$, that:

$$\frac{dE^*_{MEY}}{dr} > 0 \text{ and } \frac{d^2 E^*_{MEY}}{dr^2} > 0,$$

with $\lim_{r\to\infty} E^*_{MEY} = +\infty$ and $\lim_{r\to 0} E^*_{MEY} = 0$. That is, in line with dynamic rent maximising catch, the dynamic rent maximising effort increases at an increasing rate with the growth rate, approaching infinity as the growth rate approaches infinity. At the other extreme, under a zero growth scenario, with the dynamic rent maximising catch equal to zero, it follows that the dynamic rent maximising effort level is zero.

With regard to the carrying capacity of the environment, K, holding all other biological and economic parameter values equal, it follows, from Equation 6.1, that for all $K > 0$:

$$\frac{dX^*_{MEY}}{dK} > 0 \text{ and } \frac{d^2 X^*_{MEY}}{dK^2} > 0$$

That is to say, the dynamic rent maximising biomass rises, at an increasing rate, with the carrying capacity of the environment, for all non-negative levels of K. Furthermore, from Equations 3.3 and 6.1, it follows, for all $K > 0$, that:

$$\frac{dC^*_{MEY}}{dK} > 0 \text{ and } \frac{d^2C^*_{MEY}}{dK^2} < 0 \text{ where } K > \overline{X}_t,$$

with $\lim_{K\to\infty} C^*_{MEY} = +\infty$ and $\lim_{K\to\overline{X}} C^*_{MEY} = 0$ for all $K \le \overline{X}_t$. That is to say, for all instances in which the carrying capacity exceeds the zero-rent biomass, the dynamic rent maximising catch increases, albeit at a decreasing rate, with the carrying capacity of the environment, approaching infinity as the carrying capacity approaches infinity. At the other extreme, the dynamic rent maximising catch falls to zero for all carrying capacities less than or equal to the bionomic – or zero rent – biomass.

Finally, with regard to the dynamic rent maximising level of effort under a variable carrying capacity, by rearranging Equation 3.3, it follows from Equation 6.1, that, for all $K > 0$:

$$\frac{dE^*_{MEY}}{dK} > 0 \text{ and } \frac{d^2E^*_{MEY}}{dK^2} < 0, \text{ for all } K > \overline{X}_t,$$

with $\lim_{K\to\infty} C^*_{MEY} = C^*_{MSY}$ and $\lim_{K\to\overline{X}} C^*_{MEY} = 0$ for all $K \le \overline{X}_t$. In other words, in line with dynamic rent maximising catch rates, the dynamic rent maximising effort increases, although at a decreasing rate, with the carrying capacity of the environment, asymptotically approaching the (fixed) level of effort required to take the static dynamic maximum sustainable yield as the carrying capacity approaches infinity. At the other extreme, the dynamic rent maximising effort level falls to zero for all carrying capacities less than or equal to the bionomic – or zero rent – biomass.

With regard to the catchability coefficient, q, holding all other parameter values constant, it follows from Equation 6.1 that, for all $q > 0$:

$$\frac{dK^*_{MEY}}{dq} < 0 \text{ and } \frac{d^2K^*_{MEY}}{dq^2} < 0 \text{ for all } q \text{ yielding } C^*_{MEY} \geq 0;$$

with $\lim_{q \to 1} X^*_{MEY} = X^*_{MSY}$ and $\lim_{q \to 0} X^*_{MEY} = K$. That is to say, for all coefficients yielding a non-negative catch, the dynamic rent maximising biomass falls as the catchability coefficient rises, asymptotically approaching the dynamic maximum sustainable yield biomass as the catchability coefficient approaches unity. At the other extreme, the dynamic rent maximising biomass approaches the carrying capacity as fishing becomes uneconomic for sufficiently low values of q.

With regard to dynamic rent maximising catch, it follows, from Equations 3.3 and 6.1, that, for all $q > 0$:

$$\frac{dC^*_{MEY}}{dq} > 0 \text{ and } \frac{d^2C^*_{MEY}}{dq^2} > 0, \text{ for all for all } q \text{ yielding } C^*_{MEY} \geq 0,$$

with $\lim_{q \to 1} C^*_{MEY} = C^*_{MSY}$ and $\lim_{q \to 0} C^*_{MEY} = 0$. In other words, the dynamic rent maximising catch rises, at an increasing rate, approaching the dynamic maximum sustainable yield as the catchability coefficient rises to unity. Furthermore, for sufficiently low values of q, fishing is rendered uneconomic, yielding a dynamic rent maximising catch of zero.

As far as the dynamic rent maximising level of effort is concerned, by rearranging Equation 3.3, it follows from Equation 6.1, that, for all $q > 0$:

$$\frac{dE^*_{MEY}}{dq} > 0 \text{ and } \frac{d^2E^*_{MEY}}{dq^2} < 0 \text{ for all } \overline{X}_t > X_{MSY}; \text{ and}$$

$$\frac{dE^*_{MEY}}{dq} < 0 \text{ and } \frac{d^2E^*_{MEY}}{dq^2} > 0 \text{ for all } \overline{X}_t < X_{MSY};$$

with $\lim_{q \to 1} E^*_{MEY} = 0$ and $\lim_{q \to 0} E^*_{MEY} = 0$. That is, at sufficiently low levels of q, the dynamic rent maximising level of effort is zero because fishing is not economically viable. As q rises, the dynamic rent maximising level of effort increases, at a decreasing rate, reaching a maximum where the bionomic biomass is equal to the static maximum sustainable yield biomass. However, as q continues to rise, the level of effort required to take the dynamic rent maximising catch falls, at an increasing rate, as the bionomic biomass falls below the static maximum sustainable yield biomass. In the extreme, the dynamic rent maximising level of effort approaches zero as the catchability coefficient approaches unity.

Having established the theoretical relationship between each of the parameters and the dynamic rent maximising biomass, catch and effort levels, the following section serves to provide estimates of the impact of variable biological parameters on the dynamic rent maximising bioeconomic outcome in the west and south coast hake fisheries. Before proceeding, it should be recalled that the analysis is not undertaken on a parameter-by-parameter basis. Rather, the assumption of parameter 'scenarios' is retained in the sensitivity analysis. As explained above, the reason for using parameter scenarios rests on the fact that biological parameters fluctuate according to clearly identifiable patterns. In the west coast fishery, the relationship between parameters has been characterised by the values of r_w and q_w falling across time, with K_w rising across time. In the south coast fishery, the relationship between parameters has been characterised by the values of r_s and q_s rising across time, with K_s falling across time. In line with this, the sensitivity of the model to fluctuations in biological parameter trends is examined below. For completeness, the estimates of the dynamic rent maximising outcome and dynamic maximum sustainable yield outcome are given. For convenience, the biological parameter scenarios assumed in Chapter 5 are retained here.

6.4.2 The Impact of Variable Biological Parameter Values on the Dynamic Rent Maximising Biomass, Catch and Effort, and the Implications for Social Welfare

Results generated by the dynamic bioeconomic model reveal that the model is highly insensitive to variations in biological parameter values. Evidence of this is provided by two main findings. First, while the values taken by the biological parameters are modelled as deviating by just less than 4.5 standard deviations from the observed mean (see Table 5.2 of Chapter 5), the dynamic rent maximising biomass, catch and effort values generated by the model vary by considerably smaller margins. The results, as set out in Table 6.3, show that estimates of dynamic rent maximising biomass, catch and effort range by between 1.78 and 2.38 standard deviations. To put the matter differently, model estimates of dynamic rent maximising biomass, catch and effort for the west and south coast hake fisheries, vary by less than half the deviation in biological parameter values. Similarly, the range for dynamic maximum sustainable yield biomass, catch and effort estimates of 2.00 and 2.67 standard deviations suggest that the model is relatively insensitive to fluctuations in biological parameter values.

Supporting evidence of the robustness of the dynamic bioeconomic model is provided by the values recorded for the coefficients of variation – the standard deviation of the parameter value expressed as a percentage of the mean – of the input and output values. In this regard, the coefficients of variation of the input parameters on the west coast fishery of 15.19 percent, 14.51 percent and 12.50 percent for r_w, K_w and q_w respectively (see Table 5.2 of Chapter 5), are, on average, approximately 35 percent greater than the coefficients of variation of the model outputs, as reported in Table 6.3. Similarly, the coefficients of variation of the input parameters on the south coast fishery of 5.21 percent, 3.68 percent and 5.00 percent for r_s, K_s and q_s

respectively, are, on average, approximately 21 percent greater than the coefficients of variation of the model outputs, that is, estimates of biomass, catch and effort levels under dynamic maximum sustainable yield and dynamic maximum economic yield scenarios. In short, the estimates produced by the dynamic bioeconomic model are found to be inelastic, that is, unresponsive, to changes in biological parameter values. The implications for management in the face of welfare maximisation goals are considered in Section 6.4.3 below.

Table 6.3: The Impact of Variable Biological Parameter Values on the Estimates of Dynamic Maximum Economic Yield and Dynamic Maximum Sustainable Yield Outcomes in the West and South Coast Hake Fisheries

West Coast				South Coast		
Biomass (tons)	Catch (tons)	Effort (days)	Parameter Scenario	Biomass (tons)	Catch (tons)	Effort (hours)
			Dynamic Maximum Economic Yield Estimates			
658 324	127 512	10 527	Lower Confidence Limit	186 846	43 759	53 227
737 146	135 186	10 725	Minimum	192 448	44 075	53 261
871 406	132 074	10 525	Mean	204 718	42 998	52 509
1 021 458	123 575	10 166	Maximum	217 990	41 717	51 722
1 105 493	110 225	9 587	Upper Confidence Limit	223 844	40 998	50 876
447 169	17 287	940	Range	36 998	2 761	2 351
187 397	9 720	450	Standard Deviation	15 897	1 319	1 023
2.38	1.78	2.09	Range/Standard Deviation	2.31	2.09	2.30
21.33	7.73	4.36	Coefficient of Variation	7.75	3.09	1.96
			Dynamic Maximum Sustainable Yield Estimates			
477 654	135 724	15 443	Lower Confidence Limit	123 265	53 713	99 034
540 195	141 963	15 368	Minimum	127 220	53 770	98 291
630 025	137 188	15 122	Mean	134 204	52 709	98 188
716 895	126 962	14 882	Maximum	141 314	51 580	98 649
744 140	112 626	14 553	Upper Confidence Limit	144 727	50 618	97 153
296 486	23 098	890	Range	21 462	3 095	1 881
113 485	11 561	365	Standard Deviation	9 079	1 371	704
2.61	2.00	2.44	Range/Standard Deviation	2.36	2.26	2.67
18.25	8.83	2.42	Coefficient of Variation	6.77	2.61	0.72

6.4.3 Sensitivity Analysis: The Impact of Variable Biological Parameter Estimates on Welfare Levels

Results generated by the dynamic bioeconomic model reveal that variable biological parameter values have a muted impact upon welfare levels. Evidence of this, albeit somewhat mixed, is provided by two main findings. First, as noted above, while the values taken by the biological parameters are modelled as deviating by just less than 4.5 standard deviations from the observed mean (see Table 5.2 of Chapter 5), the estimates of welfare levels under the dynamic rent maximising strategy (and, for that matter, dynamic yield maximising strategy) are found to vary by considerably smaller margins. In this regard, the results drawn from Appendix 3, and set out in Table 6.4, reveal that welfare levels vary by 1.86 and 1.65 standard deviations in the west coast under dynamic maximum sustainable yield and maximum economic yield strategies respectively. The figures for the south coast fishery are 1.85 and 1.94 standard deviations respectively. Based on this evidence, the estimates of welfare levels are argued to be highly insensitive to fluctuations in biological parameter values, with the implication being that the model provides an excellent basis for setting management targets.

However, the evidence regarding model stability is somewhat more mixed when assessed on the basis of coefficient of variation values. Thus, the coefficient of variation for biological parameters ranges between an average of 13.19 percent on the west coast to 4.71 percent on the south coast. The coefficient of variation of welfare levels are considerably higher, measuring 22.52 percent and 10.93 percent on the west coast and 82.97 percent and 5.12 percent on the south coast under dynamic maximum sustainable yield and dynamic maximum economic yield outcomes respectively. Measured by this standard, the estimates of welfare levels generated by the model would seem to be relatively sensitive to biological parameter values, which is problematic from a management perspective.

Table 6.4: The Impact of Variable Biological Parameter Values on Welfare Levels under Dynamic Maximum Economic Yield and Dynamic Maximum Sustainable Yield in the West and South Coast Hake Fisheries

West Coast (Index Points) Maximum Sustainable Yield	West Coast (Index Points) Maximum Economic Yield	Parameter Scenario	South Coast (Index Points) Maximum Sustainable Yield	South Coast (Index Points) Maximum Economic Yield
45 382.45	65 929.05	Lower Confidence Limit	2 215.32	16 080.96
52 060.20	72 444.75	Minimum	2 658.68	16 379.28
48 724.30	70 502.75	Mean	1 651.24	15 693.32
39 902.30	64 103.90	Maximum	282.52	14 821.56
27 491.31	54 140.01	Upper Confidence Limit	98.44	14 542.48
17 891.14	11 789.04	Range	2 116.88	1 538.48
42 712.11	65 424.09	Mean	1381.24	15 503.52
9 620.70	7 148.62	Standard Deviation	1 145.99	794.53
1.86	1.65	Range/Standard Deviation	1.85	1.94
22.52	10.93	Coefficient of Variation	82.97	5.12

Nevertheless, in both the west and south coast fisheries, it is the estimates of welfare levels under the dynamic maximum sustainable yield scenario that are relatively elastic, that is, sensitive, to biological parameter values. The estimates of welfare levels under the dynamic maximum economic yield scenario – arguably the basis for setting management targets – are considerably less sensitive to biological parameter estimates, with a coefficient of variation some 17.13 percent less than the coefficient of variation for biological parameters in the west coast and 8.01 percent greater than the coefficient of variation of biological parameter values in the south coast. Moreover, in both the west and south coast fisheries, the coefficients of variation under the dynamic maximum economic yield scenario are low in absolute terms – 10.13 percent and 5.12 percent respectively – which provides for a high degree of confidence in the stability of the model and, accordingly, in the robustness of the management targets (that is, dynamic rent maximising biomass, catch and effort levels) provided by the model.

Caveats aside, a number of points should be made regarding the management implications of the above results. First, irrespective of the values assumed for the biological parameters, the increase in rents achieved by shifting the fishery from the dynamic maximum sustainable yield outcome to the dynamic maximum economic yield outcome is considerable. In the case of the west coast fishery, the average increase in rents is 53.17 percent. In the south coast fishery, the impact upon rents is more dramatic, with rents increasing by an average 1 022.43 percent. It should also be noted that while the potential increase in rents in the south coast fishery is considerably more dramatic than that for the west coast fishery, the south coast fishery accounts for a significantly smaller portion of total rents (between 3.5 and 22.5 percent under the dynamic maximum sustainable yield and dynamic maximum economic yield strategies respectively). None the less, the increase in total rents achieved by shifting the west and south coast fisheries from biological to bioeconomic outcomes is considerable, amounting to an average 83.54 percent across the two fisheries. Second, the rent estimates generated under the dynamic maximum economic yield scenario are relatively stable under variable biological parameter conditions, evidenced by coefficients of variation of rents of 10.93 percent and 5.12 percent versus average coefficients of variation on biological parameters of 14.07 percent and 4.63 percent in the west and south coast fisheries respectively (as per Table 5.2 of Chapter 5 and Table 6.4).

The conclusion to be drawn from the above is unambiguous: in spite of biological parameter fluctuations, the management implications of the dynamic bioeconomic model remain unaltered. Specifically, the rents generated under the dynamic maximum economic yield strategy are, without exception, substantially greater than those generated under the dynamic maximum sustainable yield strategy. Moreover, the bioeconomic rent estimates generated by the dynamic bioeconomic model are found to be reasonably stable which, while not a necessary outcome, is highly desirable from a management perspective.

In summation, while variations in biological parameter values are reflected in output values generated by the dynamic bioeconomic model, as well as estimates of welfare levels, the extent of the variation in output values is found to be considerably less than that of input values. Similarly, estimates of welfare levels are found to be relatively unresponsive (albeit less so than output estimates) to variations in biological parameter values. Moreover, the results reveal that the increases in rents achieved by shifting the fishery from the dynamic maximum sustainable yield strategy to the dynamic maximum economic yield strategy remain considerable over the entire range of biological parameter values adopted. Accordingly, it is concluded that the dynamic bioeconomic model is sufficiently robust to provide resource managers with management targets in which they are able to place a high degree of confidence.

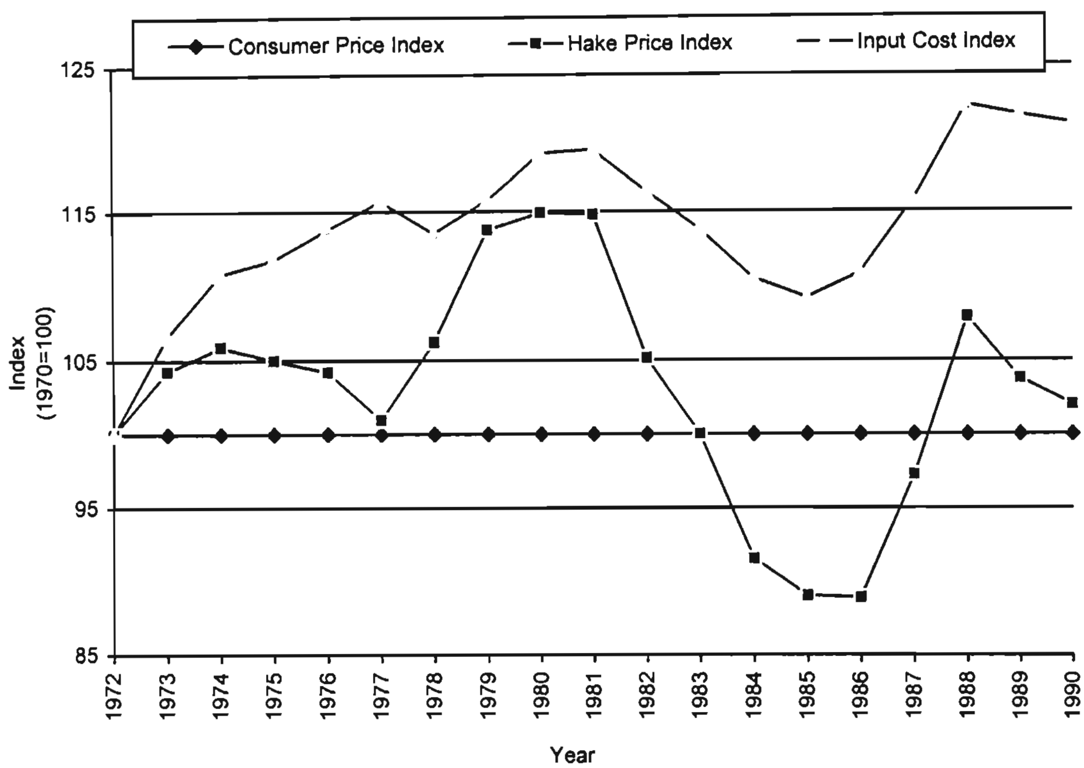
Before proceeding, it should be noted that it is assumed in the remainder of this study that biological parameter values are given by the mean of the observed values of the biological parameters, r, K and q, as set out in Table 5.2 of Chapter 5. Two points should be made in this regard. First, allowing for variable biological parameters under a variable economic parameter scenario generates a substantial number of potential outcomes. Yet, as demonstrated in Sections 6.4.2 and 6.4.3 above, the model estimates are highly insensitive to variations in biological parameter values. Thus, assuming constant values for biological parameters considerably reduces the number of possible outcomes (in this case by a factor of five) without violating the integrity of the model or its results. Second, it should be emphasised that any other set of biological parameter values, for example, the set of lower confidence interval values, could be adopted in place of the set of mean biological parameter values. However, to reiterate, assuming a different set of values for biological parameters would have an insignificant impact on the estimates produced by the dynamic bioeconomic model.

6.5 The Impact of Variable Economic Parameter Values on the Dynamic Rent Maximising Biomass, Catch and Effort, and the Implications for Social Welfare

In contrast to the sensitivity analysis conducted with regard to biological parameter values, sensitivity of the dynamic bioeconomic model to economic parameters is conducted on a parameter-by-parameter basis. The reason for this is that, unlike the case of biological parameters, there are no obvious reasons for expecting relationships to exist between the various economic parameters. This is supported by anecdotal evidence presented below. First, with regard to cost and price levels, while the trend is for prices and costs to increase across time, their movements relative to one another are uncorrelated. For instance, using the cost and price data cited above, the correlation coefficient between the hake price index and the index of fishing industry input costs is relatively low, measuring 0.45 for the period 1972-90. The correlation coefficient between hake price inflation and fishing cost inflation, that is, the change in indices across time, is somewhat higher, measuring 0.6248 over the period 1972-90. This result is suggestive of a stronger relationship between prices and costs than that revealed by the first result. However, the relationship is too weak to warrant the conclusion that changes in cost and price levels take place on the basis of a well-defined, strong relationship.

A similar conclusion is reached with regard to the possible existence of a relationship between the quasi-discount rate and prices or costs. There are two reasons for this. First, the available evidence suggests that movements in cost and (particularly) price levels are relatively unsystematic, failing to observe any particular trend (Figure 6.3). Furthermore, although it is argued in Section 6.2 that the temporal trend in growth rates of hake stocks might influence discount rates, it is highly unlikely that these trends will not be disturbed, for example, by exogenous changes in tastes, income levels or the state of technology. Accordingly, it is concluded that, while quasi-discount rates are likely to vary across time, there is no basis for suggesting that the trend in quasi-discount rates might be either upwards or downwards across time or, for that matter, whether there is any trend in the movement of quasi-discount rates across time.

Figure 6.3: Trends in the Hake Price Index and Fishing Industry Input Cost Index Relative to the Consumer Price Index (1970-90)



Source: Adapted from Industrial Development Corporation (1992).

Based on these arguments, it is concluded that, in contrast to the case of biological parameter movements, there is no obvious basis or evidence for suggesting that economic parameters are likely to vary according to a predetermined pattern. Accordingly, the sensitivity of the model to changes in economic parameter values is undertaken on a parameter-by-parameter basis. The analysis of the impact of variable economic parameter values on the dynamic rent maximising outcome is made up of three parts. Section 6.5.1 provides a review of the nature of the relationship between each of the economic parameters and the dynamic rent maximising outcome. Section 6.5.2 is concerned with estimating the dynamic rent maximising outcome under variable economic parameter conditions. Finally, Section 6.5.3 considers the impact of variable economic parameter values on welfare levels under the dynamic bioeconomic model.

6.5.1 A Review of the Relationship between Economic Parameters and the Dynamic Rent Maximising Outcome

Turning first to examine the nature of the relationship between variable prices and the dynamic rent maximising outcome, the extended model, as set out in Equation 6.1, provides for a number of results. First, from Equation 6.1, it follows that for all $p \geq p_{\min}$:

$$\frac{dX^*_{MEY}}{dp} < 0 \text{ and } \frac{d^2 X^*_{MEY}}{d^2 p} < 0;$$

with $\lim_{p \to \infty} X^*_{MEY} = X^*_{MSY}$ and $\lim_{p \to p_{\min}} X^*_{MEY} = K$, where $p_{\min}$ represents the lowest price at which harvesting is economically viable. In other words, the dynamic rent maximising biomass falls below the carrying capacity biomass (K), although at a decreasing rate, as price rises above the minimum feasible harvest price ($p_{\min}$). Furthermore, in the extreme, the dynamic rent maximising biomass asymptotically approaches the discounted maximum sustainable yield biomass as price approaches infinity.

With regard to dynamic rent maximising catch, it follows from Equations 3.3 and 6.1 that for all $p \geq p_{\min}$:

$$\frac{dC^*_{MEY}}{dp} > 0 \text{ and } \frac{d^2 C^*_{MEY}}{dp^2} < 0 \text{ for } p \text{ such that } X^*_{MEY} > X_{MSY}\text{; and}$$

$$\frac{dC^*_{MEY}}{dp} < 0 \text{ with, initially, } \frac{d^2 C^*_{MEY}}{dp^2} > 0 \text{ and then } \frac{d^2 C^*_{MEY}}{dp^2} < 0 \text{ for } p \text{ such that}$$

$$X^*_{MEY} < X_{MSY};$$

with $\lim_{p \to \infty} C^*_{MEY} = C^*_{MSY}$ and $\lim_{p \to p_{\min}} C^*_{MEY} = 0$. That is to say, the dynamic rent maximising catch rises with prices, approaching the static maximum sustainable yield. However, once prices have risen sufficiently to reduce the dynamic rent maximising biomass to below the static maximum sustainable yield biomass, the dynamic rent maximising catch falls (initially, at an increasing rate, and then at a decreasing rate) to approach the dynamic maximum sustainable yield as prices approach infinity. At the other extreme, at sufficiently low prices, fishing is not economically viable, rendering a dynamic rent maximising catch of zero. The net result is that the price:catch relationship produces a 'backward-bending' supply of output curve, a feature of the bioeconomic model which is relatively well documented in the fisheries economics literature (Copes, 1970 & 1972; Tisdell, 1972; Clark 1976 & 1985). The implications of the backward-bending supply of output curve are discussed more fully in Section 6.5.2.[11]

As far as the dynamic rent maximising level of effort is concerned, it follows from Equations 3.3 and 6.1 that for all $p \geq p_{\min}$:

$$\frac{dE^*_{MEY}}{dp} > 0 \text{ and } \frac{d^2 E^*_{MEY}}{dp^2} < 0 \text{ for all } p > p_{\min};$$

with $\lim_{p \to \infty} E^*_{MEY} = E^*_{MSY}$ and $\lim_{p \to p_{\min}} E^*_{MEY} = 0$. That is, as price rises from $p_{\min}$, effort increases, from zero, at a decreasing rate, asymptotically approaching that level of effort required to take the dynamic maximum sustainable catch (C^*_{MSY}) as price approaches infinity.

With regard to the impact of costs on the dynamic rent maximising outcome, it follows from Equation 6.1 that, for all $0 \leq c \leq c_{\max}$:

[11] More recently, Gautam *et al* (1996) have extended this analysis to explain a backward-bending supply of labour curve in the fishing industry.

$$\frac{dX^*_{MEY}}{dc} > 0 \text{ and } \frac{d^2X^*_{MEY}}{dc^2} > 0;$$

with $\lim_{c \to c_{\max}} X^*_{MEY} = K$ and $\lim_{c \to 0} X^*_{MEY} = X^*_{MSY}$, where $c_{\max}$ represents the cost level at which the price:cost ratio renders harvesting economically unfeasible. In other words, the dynamic rent maximising biomass rises, at an increasing rate, from the dynamic maximum sustainable yield biomass as the cost level rises from zero. Once the cost level reaches $c_{\max}$, however, fishing is no longer economically viable, such that the dynamic rent maximising biomass is given by the carrying capacity of the environment.

With regard to dynamic rent maximising catch, it follows from Equations 3.3 and 6.1 that for all $0 \le c \le c_{\max}$:

$$\frac{dC^*_{MEY}}{dc} > 0 \text{ and } \frac{d^2C^*_{MEY}}{dc^2} < 0 \text{ for all } c \text{ such that } X^*_{MEY} < X_{MSY} \text{; and}$$

$$\frac{dC^*_{MEY}}{dc} < 0 \text{ and } \frac{d^2C^*_{MEY}}{dc^2} > 0 \text{ for all } c \text{ such that } X^*_{MEY} > X_{MSY};$$

with $\lim_{c \to c_{\max}} C^*_{MEY} = 0$ and $\lim_{c \to 0} C^*_{MEY} = C^*_{MSY}$. That is to say, for cost levels which produce a dynamic rent maximising biomass less than the static maximum sustainable yield biomass, catch rises, at a decreasing rate, with cost (as the dynamic rent maximising biomass falls to approach the static maximum sustainable yield biomass). However, once cost has risen sufficiently to produce a dynamic rent maximising biomass greater then the static maximum sustainable yield biomass, further cost increases result in the dynamic rent maximising catch falling at an increasing rate to approach zero as cost approaches $c_{\max}$.

As far as the dynamic rent maximising level of effort is concerned, it follows from Equations 3.3 and 6.1 that for all $0 \le c \le c_{\max}$:

$$\frac{dE^*_{MEY}}{dc} < 0 \text{ and } \frac{d^2E^*_{MEY}}{dc^2} > 0;$$

with $\lim_{c \to c_{max}} E^*_{MEY} = 0$ and $\lim_{c \to 0} E^*_{MEY} = E^*_{MSY}$. That is to say, as cost rises from zero to c_{max}, effort decreases, at an increasing rate, from that level required to take the dynamic maximum sustainable catch (C^*_{MSY}), approaching zero as cost approaches c_{max}.

Finally, with regard to the impact of variable discount rates on the dynamic rent maximising outcome, it follows from Equation 6.1 that, for all $\delta \geq 0$:

$$\frac{dX^*_{MEY}}{d\delta} < 0;$$[12]

with $\lim_{\delta \to \infty} X^*_{MEY} = \overline{X}_t$ and $\lim_{\delta \to 0} X^*_{MEY} = X_{MEY}$. That is, the dynamic rent maximising biomass decreases monotonically from the static maximum economic yield level to the zero-rent (bionomic) level as the quasi-discount rate increases from zero to infinity.

With regard to dynamic rent maximising catch, it follows from Equation 3.3 and Equation 6.1 that, for all $\delta \geq 0$:

$$\frac{dC^*_{MEY}}{d\delta} > 0 \text{ for all } \delta \text{ such that } X^*_{MEY} > X_{MSY}; \text{ and}$$

$$\frac{dC^*_{MEY}}{d\delta} < 0 \text{ for all } \delta \text{ such that } X^*_{MEY} < X_{MSY};$$

[12] No comment is made on the rate of increase or decrease in the state variable, that is, dynamic rent maximising biomass, under variable discount rates or, for that matter, on either of the derived outcomes, that is, dynamic rent maximising catch and effort levels. The reason for this is that the rate of change is highly variable, 'switching' by as many as four times (in the case of dynamic rent maximising catch) over a discount rate range of $0 \leq \delta \leq +\infty$.

with $\lim_{\delta \to \infty} C^*_{MEY} = \overline{C}_t$ and $\lim_{\delta \to 0} C^*_{MEY} = C_{MEY}$. In other words, as the discount rate rises from zero, the dynamic rent maximising catch rises as the biomass is reduced to approach the static maximum sustainable yield level. However, once the dynamic rent maximising biomass has been reduced below the static maximum sustainable yield biomass, further increases in the discount rate result in the dynamic rent maximising catch falling, approaching the bionomic yield outcome as the discount rate rises to infinity.

As far as the dynamic rent maximising level of effort is concerned, it follows from Equations 3.3 and 6.1 that, for all $\delta \geq 0$:

$$\frac{dE^*_{MEY}}{d\delta} > 0;$$

with $\lim_{\delta \to \infty} E^*_{MEY} = \overline{E}_t$ and $\lim_{\delta \to 0} E^*_{MEY} = E_{MEY}$. That is to say, the dynamic rent maximising level of effort monotonically increases with the discount rate, rising from the static maximum economic yield level under a zero discount rate scenario, and approaching the bionomic level of effort as the discount rate rises to infinity.

Having established the nature of the relationship between each of the economic parameters and the dynamic rent maximising biomass, catch and effort levels, the section below is concerned with providing estimates of the impact of variable economic parameters on the dynamic rent maximising outcome in the west and south coast hake fisheries. However, before proceeding, two points should be noted. First, in line with the conclusion arrived at in Section 6.2.2 above, biological parameter values are assumed to be given by the mean of the observed values of r, K and q over the period 1985-95, as set out in Chapter 5. Second, given that economic parameter values are not fixed across time, it is useful to provide range values for the parameters p, c and q, to represent the likely extent of fluctuations in parameter values.

The evidence presented in Section 6.2.2 provides a close guide to the extent to which economic parameter values can 'reasonably' be expected to fluctuate across time. The price data contained in Section 6.2.2 reveal that, over the period 1970-90, real prices fluctuated in a range of 25 percentage points (Figure 6.1) to 30 percentage points (Figure 6.3) of the mean price. In contrast, the evidence pertaining to real costs reveal that, over the period 1970-90, costs fluctuated in a range of 25 percentage points (Figure 6.2) to 50 percentage points (Table 6.1) of the mean real cost. Based on these figures, and given the earlier argument that (especially cost) data contain possible errors, the widest cited range of a 50 percentage point fluctuation is assumed to apply to both cost and price data, with the fluctuation measured by a 25 percentage point variation in real price and cost levels either side of the mean cost and price levels. In other words, the hake price index is assumed to vary within a range of 0.7500-1.2500 (per ton) while the index measuring the cost of trawling for hake is assumed to range between 4.3875-7.3125 per standard-boat-day on the west coast and 0.3900-0.6500 per standard-boat-hour on the south coast.

Furthermore, in the absence of any evidence as to the likely extent of fluctuations in the quasi-discount rate, the same level of variation as for cost and price data is assumed to apply in the case of the discount rate, that is, a 25 percent fluctuation either side of the mean value of 4.5 percent, producing a discount rate range of 3.375-5.625 percent per annum which, by coincidence, closely reflects the range of 3 to 6 percent initially suggested in Chapter 5. Finally, while the focus of the analysis conducted below lies with establishing the sensitivity of the dynamic bioeconomic model estimates to fluctuations in economic parameter values across the so-called 'reasonable range', for completeness, the section does consider the impact of variations in parameter estimates across the entire range of possible outcomes.

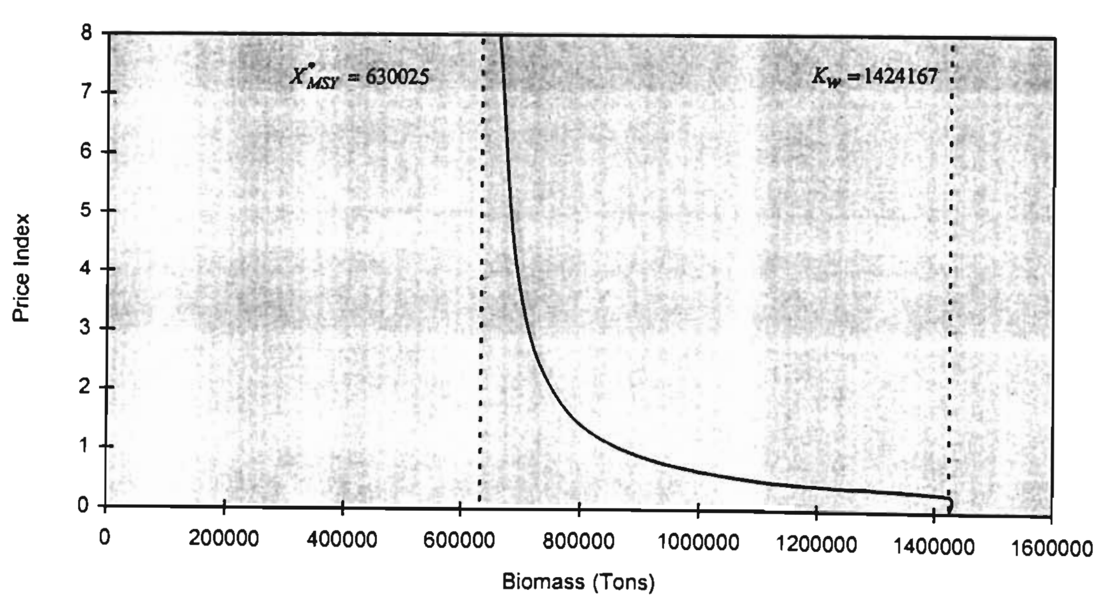
6.5.2 Sensitivity Analysis: The Responsiveness of Dynamic Rent Maximising Biomass, Catch and Effort Levels to Changes in Economic Parameter Estimates

6.5.2.1 The West Coast Hake Fishery

(a) Changes in Price

The dynamic rent maximising biomass estimates produced by the dynamic bioeconomic model confirm the arguments set out in Section 6.5.1, namely, that the dynamic rent maximising biomass falls below the carrying capacity biomass ($K_w = 1424167$ tons), although at a decreasing rate, as price rises above the minimum feasible harvest price, where $p_{\min} = 0.2582$ in the west coast fishery. Furthermore, in the extreme, the dynamic rent maximising biomass asymptotically approaches the discounted maximum sustainable yield biomass as price approaches infinity ($X^*_{MSY} = 630025$ tons). These results are summarised in Figure 6.4.

Figure 6.4: Response of Dynamic Rent Maximising Biomass in the West Coast Hake Fishery to Price Variations



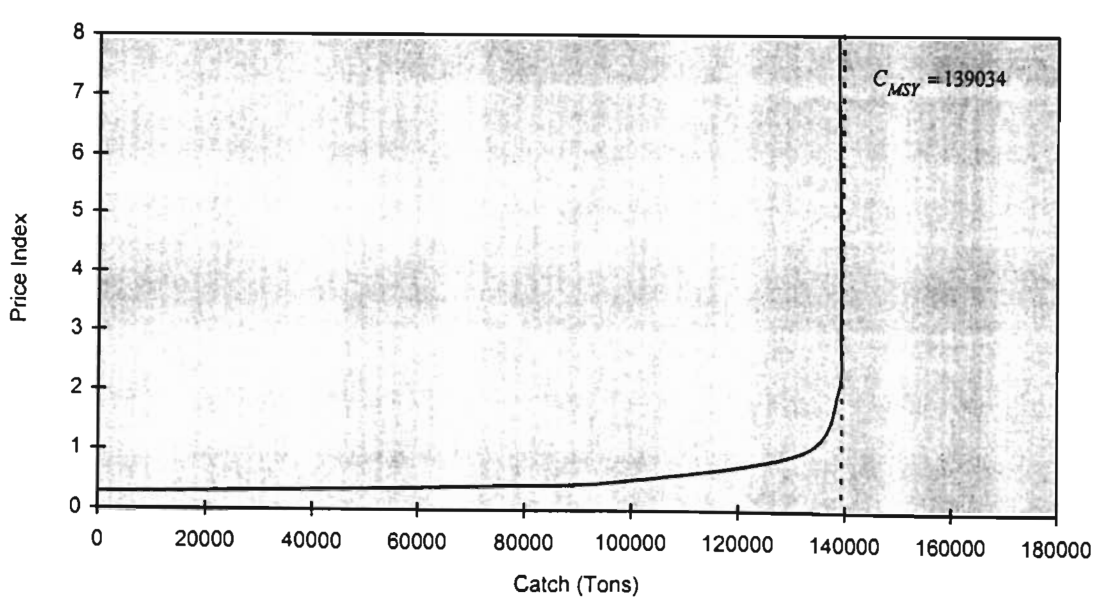
More important, however, is the result that, over the feasible price range, $0.75 \leq p_w \leq 1.25$, the dynamic rent maximising biomass in the west coast fishery is highly unresponsive to changes in the price level. Indeed, the results of the estimation procedure, set out in Table 6.5, reveal that over the feasible price range, a 1 percent change in the price level results, on average, in a 0.55 percent change in the dynamic rent maximising biomass. Further evidence of the insensitivity of the dynamic rent maximising biomass to changes in the price level is given by the coefficient of variation of the dynamic rent maximising biomass of 4.61 percent, which is approximately one-quarter the coefficient of variation of the price level. As an aside, the dynamic maximum sustainable biomass is unaffected by the price level: $X^{*}_{MSY} = 630025$ tons for all p_w.

Table 6.5: Dynamic Rent Maximising Biomass, Catch and Effort Levels in the West Coast Hake Fishery over the Assumed Feasible Price Range

Price (p_w)		Biomass (X^{*}_{MEY})	Catch (C^{*}_{MEY})	Effort (E^{*}_{MEY})
0.7500		947 757	123 805	9 071
0.8000		928 796	126 157	9 433
0.8500		911 999	128 076	9 752
0.9000		897 013	129 657	10 038
0.9500		883 556	130 972	10 294
1.0000		871 406	132 074	10 525
1.0500		860 379	133 004	10 735
1.1000		850 324	133 794	10 927
1.1500		841 119	134 469	11 102
1.2000		832 659	135 048	11 263
1.2500		824 857	135 547	11 412
0.50	Range	122 900.08	11 742.18	2 340.18
1.00	Mean	877 260.47	131 145.79	10 413.84
0.16	Standard Error	40 433.49	3 839.54	769.91
16.58	Coefficient of Variation (%)	4.61	2.93	7.39

With regard to catch, the estimates produced by the dynamic bioeconomic model confirm the arguments set out in Section 6.5.1, namely, the dynamic rent maximising catch rises at a decreasing rate as price rises above, $p_{\min} = 0.2582$, approaching the static maximum sustainable yield ($C_{MSY} = 139034$ tons). However, once prices have risen sufficiently to reduce the dynamic rent maximising biomass below the static maximum sustainable yield biomass ($X_{MSY} = 712084$ tons), the dynamic rent maximising catch falls, asymptotically approaching the dynamic maximum sustainable yield ($C^*_{MSY} = 137188$ tons) as price approaches infinity. As noted, the net result is that the price-catch relationship produces a 'backward-bending' supply of output curve. The implications for welfare are commented on below. Here, however, it should be said that the price level at which the price-output relationship becomes negative ($p = 3.00$) is sufficiently high to render the backward-bending supply of output curve in the west coast hake fishery useless for all practical purposes. The above results are summarised in Figure 6.5.

Figure 6.5: Response of Dynamic Rent Maximising Catch in the West Coast Hake Fishery to Price Variations

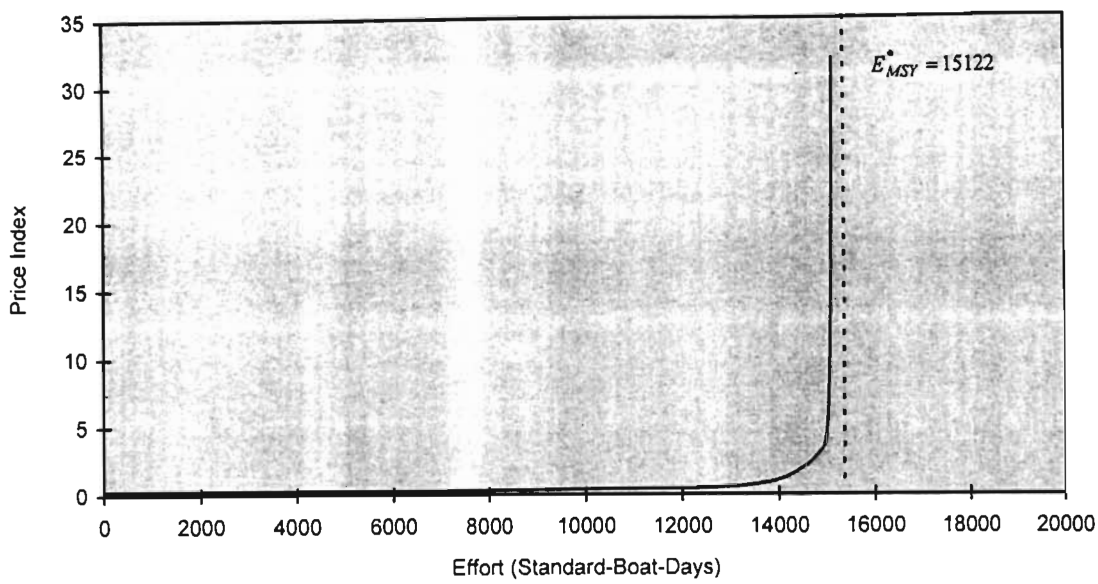


Over the feasible price range, the dynamic rent maximising catch in the west coast fishery is highly unresponsive to changes in the price level. Returning to Table 6.5, the results of the estimation procedure reveal that, over the feasible price range, $0.75 \leq p_W \leq 1.25$, a 1 percent change in the price level produces an average 0.18 percent change in the dynamic rent maximising catch. Further evidence of the insensitivity of the dynamic rent maximising catch to changes in the price level is given by the coefficient of variation of the dynamic rent maximising catch of 2.93 percent, which is approximately one-sixth the size of the coefficient of variation of 16.58 percent for the price level. It may also be noted that, as is the case with biomass, the dynamic maximum sustainable yield is unaffected by the price level (C^*_{MSY} tons for all p_w).

With regard to the impact of variable prices on the dynamic rent maximising effort, the estimates reveal that as price rises from $p_{\min} = 0.2582$, the dynamic rent maximising level of effort in the west coast hake fishery increases from $E^* = 0$ standard-boat-days, asymptotically approaching that level of effort required to take the dynamic maximum sustainable catch ($E^*_{MSY} = 15122$ standard-boat-days) as price approaches infinity. These results are summarised in Figure 6.6.

The dynamic rent maximising effort level in the west coast fishery is highly unresponsive to changes in price over the feasible price range, $0.75 \leq p_w \leq 1.25$. Returning to Table 6.5, the results of the estimation procedure reveal that, over the feasible price range, a 1 percent change in the price level produces an average 0.46 percent change in the dynamic rent maximising effort. The relative insensitivity of the dynamic rent maximising effort level to variations in price is supported by the coefficient of variation of the dynamic effort level (7.49 percent) being approximately half as great as that recorded for price (16.58 percent). As is the case with biomass and catch, the level of effort required to take the dynamic maximum sustainable yield is unaffected by the price level ($E^*_{MSY} = 15122$ standard-boat-days for all p_w).

Figure 6.6: Response of Dynamic Rent Maximising Effort in the West Coast Hake Fishery to Price Variations

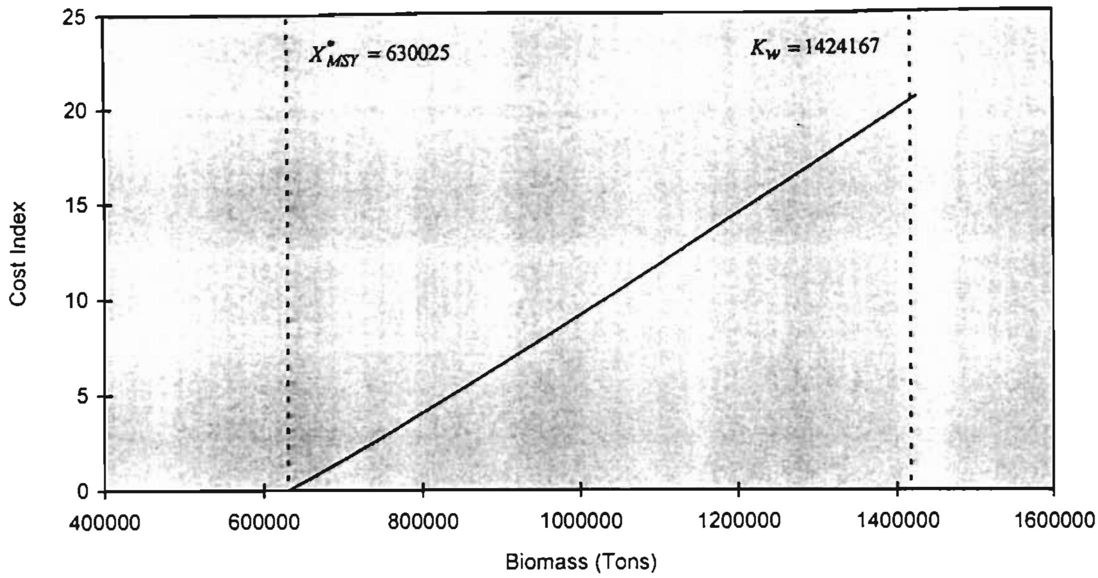


The conclusion to the above is inescapable: while price fluctuations result in changes in the dynamic rent maximising biomass, catch and effort levels, over the so-called feasible price range, the model estimates are highly insensitive to price fluctuations. The implications of this result for welfare outcomes and for management policies are examined in Section 6.5.3.

(b) Changes in Cost

The dynamic rent maximising biomass estimates produced by the dynamic bioeconomic model confirm the arguments set out in Section 6.5.1: the dynamic rent maximising biomass rises from the dynamic maximum sustainable yield biomass ($X^*_{MSY} = 630025$ tons) to the carrying capacity biomass ($K_w = 1424167$ tons) as cost rises from zero ($c = 0$) to infinity, although fishing becomes unfeasible at a cost level of $c = c_{\max} = 20.50$ in the west coast fishery. These results are summarised in Figure 6.7.

Figure 6.7: Response of Dynamic Rent Maximising Biomass in the West Coast Hake Fishery to Cost Variations



Over the feasible cost range, $4.3875 \leq c_w \leq 7.3125$, the dynamic rent maximising biomass in the west coast fishery is highly unresponsive to changes in the price level. The results of the estimation procedure, given in Table 6.6, reveal that over the cost range, $4.3875 \leq c_w \leq 7.3125$, a 1 percent change in the cost index results, on average, in a 0.27 percent change in the dynamic rent maximising biomass. Further evidence of the insensitivity of the dynamic rent maximising biomass to changes in the price level is given by the coefficient of variation of the dynamic rent maximising biomass, which is approximately one-quarter the coefficient of variation of the cost index. As an aside, the dynamic maximum sustainable biomass is unaffected by changes in the cost index (X^*_{MSY} = 630025 tons for all $0 \leq c_w \leq c_{\max}$).

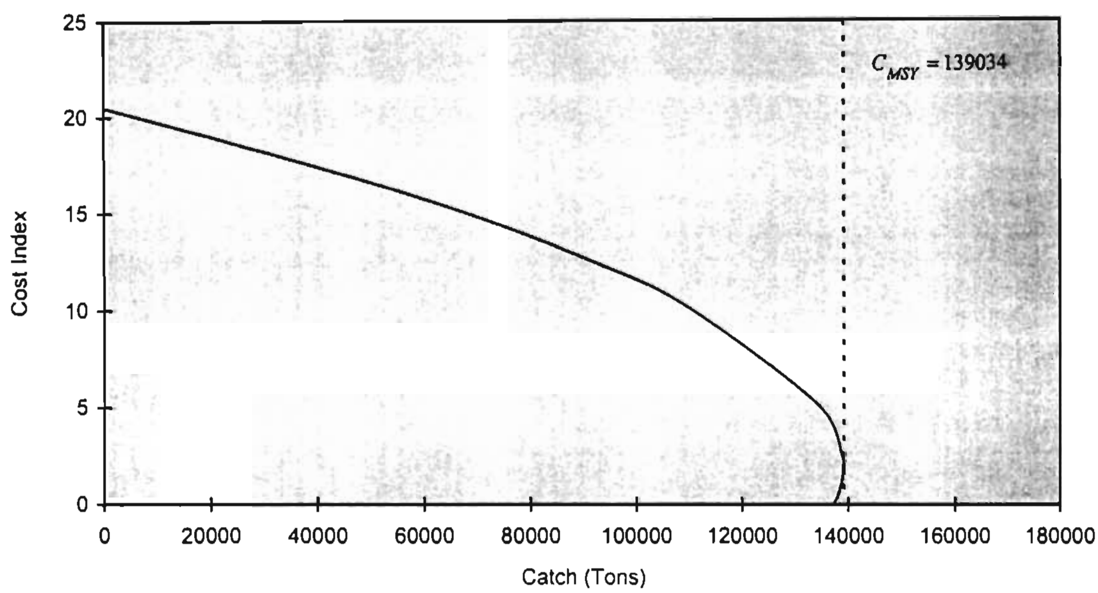
With regard to catch, the dynamic rent maximising catch initially rises, at a decreasing rate, from the dynamic maximum sustainable yield ($C^*_{MSY} = 137188$ tons) under zero cost conditions ($c = 0$), to approach the static maximum sustainable yield $C_{MSY} = 139034$ tons. However, once costs have risen sufficiently ($c \geq 1.9207$) to raise the dynamic biomass above the static maximum sustainable yield biomass ($X_{MSY} = 712084$ tons), further cost increases result in the dynamic optimum catch falling, at an increasing rate, to approach zero as the cost index approaches $c_{\max} = 20.5080$. The above results are summarised in Figure 6.8.

Table 6.6: Dynamic Rent Maximising Biomass, Catch and Effort Levels in the West Coast Hake Fishery over the Assumed Feasible Cost Range

Cost (c_w)		Biomass ($X^{\bullet}_{MEY}$)	Catch ($C^{\bullet}_{MEY}$)	Effort ($E^{\bullet}_{MEY}$)
4.3875		813 117	136 235	11 635
4.6800		824 857	135 547	11 412
4.9725		836 553	134 786	11 189
5.2650		848 209	133 953	10 967
5.5575		859 826	133 049	10 746
5.8500		871 406	132 074	10 525
6.1425		882 950	131 029	10 306
6.4350		894 459	129 914	10 086
6.7275		905 936	128 730	9 868
7.0200		917 381	127 478	9 650
7.3125		928 796	126 157	9 433
2.93	Range	115 678.77	10 078.48	2 202.68
5.85	Mean	871 226.54	131 723.09	10 528.73
0.97	Standard Error	36 360.53	3 360.63	730.44
16.58	Coefficient of Variation (%)	4.17	2.55	6.94

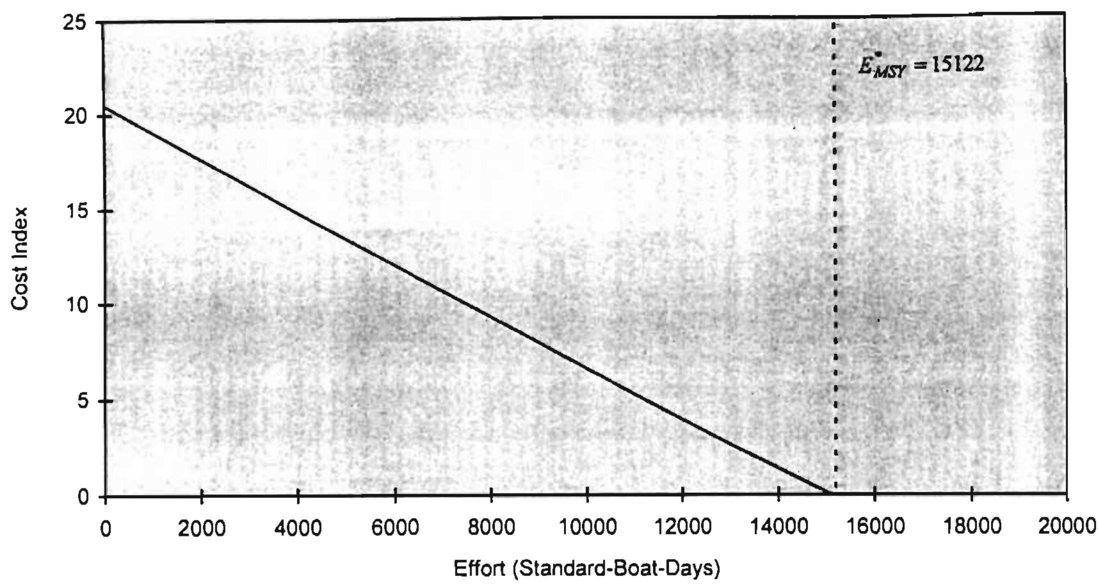
Over the entire feasible cost range, $(4.3875 \leq c_w \leq 7.3125)$, the relationship between the dynamic rent maximising catch and the cost index is negative, that is, higher costs reduce the dynamic rent maximising catch. Second, and of greater interest, the dynamic rent maximising catch in the west coast fishery is highly unresponsive to changes in the cost index over the feasible range. Indeed, over the feasible cost range, a 1 percent change in the cost index produces an average 0.15 percent change in the dynamic rent maximising catch. Further evidence of the insensitivity of the dynamic rent maximising catch to changes in the price level is given by the coefficient of variation of the dynamic rent maximising catch of 2.55 percent, which is approximately one-sixth the size of the coefficient of variation of 16.58 percent for the cost index. As is the case with biomass, the dynamic maximum sustainable yield is unaffected by the cost level ($C^*_{MSY} = 137188$ tons for all c_w).

Figure 6.8: Response of Dynamic Rent Maximising Catch in the West Coast Hake Fishery to Cost Variations



With regard to effort, as costs rise from zero to $c_{\max}$, effort decreases, at an increasing rate, from that level required to take the dynamic maximum sustainable catch ($E^*_{MSY} = 15122$ standard-boat-days), falling to zero as cost rises to $c_{\max}$. These results are summarised in Figure 6.9.

Figure 6.9: Response of Dynamic Rent Maximising Effort in the West Coast Hake Fishery to Cost Variations



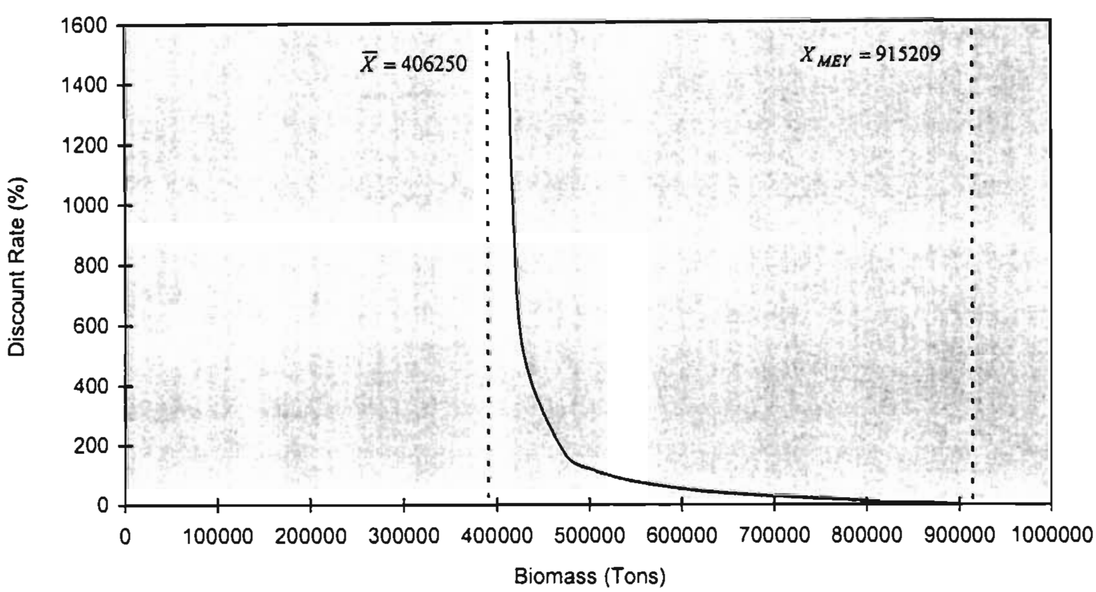
Over the feasible cost range, the dynamic rent maximising effort level in the west coast fishery is highly unresponsive to changes in the cost index. More specifically, the results of the estimation procedure reveal that, over the feasible range, a 1 percent change in the cost index produces an average 0.42 percent change in dynamic rent maximising effort. The relative insensitivity of the dynamic rent maximising effort level to variations in cost is further confirmed by the coefficient of variation of the dynamic effort level (6.94 percent) being less than half as great as that recorded for cost (16.58 percent) over the feasible range. As is the case with biomass and catch, the level of effort required to take the dynamic maximum sustainable yield is unaffected by the cost level ($E^{*}_{MSY} = 15122$ standard-boat-days for all c_w).

It may be concluded, therefore, that while fluctuations in the cost level result in changes in the dynamic rent maximising biomass, catch and effort, over the so-called feasible range, the model estimates are highly insensitive to changes in costs. The implications of this result for welfare outcomes and for management policies are considered in Section 6.5.3.

(c) Changes in the Quasi-Discount Rate

For all $\delta \geq 0$ percent, the dynamic rent maximising biomass decreases as the discount rate rises, falling from a level equal to the static maximum economic yield biomass for $\delta = 0$ percent and approaching the zero rent biomass ($\bar{X} = 406250$ tons) as the discount rate rises to infinity. The impact of changes in the quasi-discount rate on the dynamic rent maximising biomass in the west coast hake fishery are summarised in Figure 6.10.

6.10 Response of Dynamic Rent Maximising Biomass in the West Coast Hake Fishery to Quasi-Discount Rate Variations

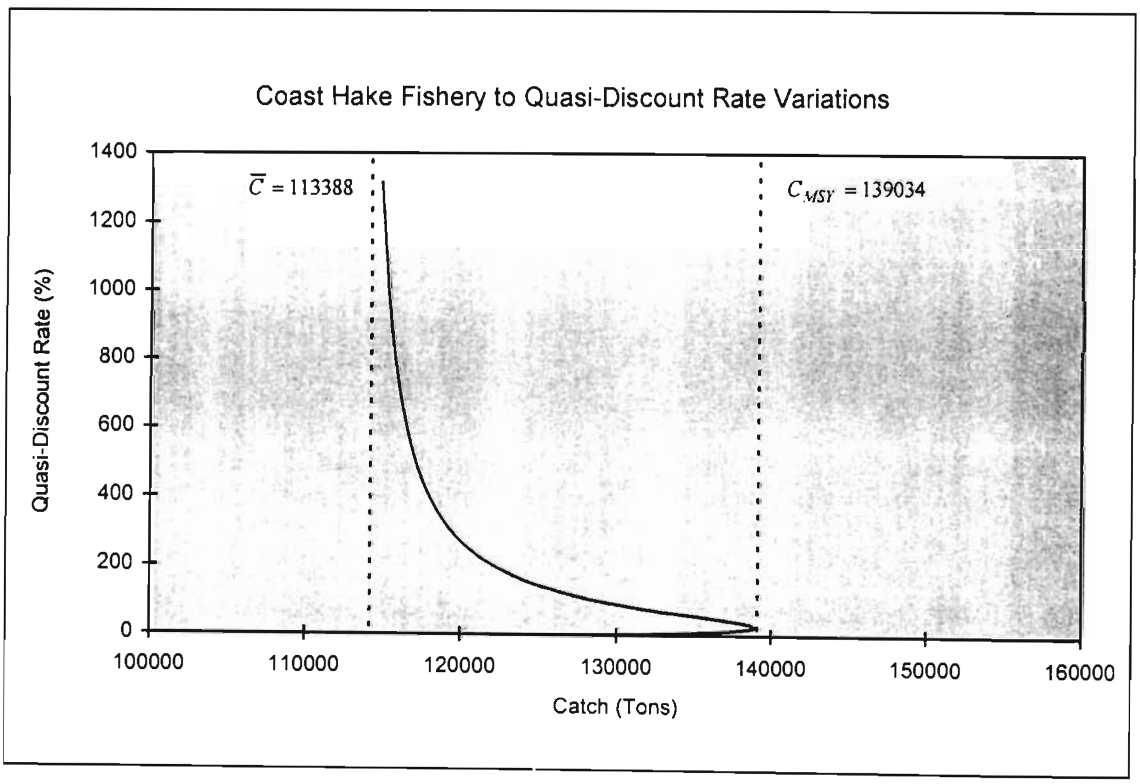


More important, over the feasible quasi-discount rate range $3.3750 \leq \delta_w \leq 5.6250$ percent, the dynamic rent maximising biomass in the west coast fishery is highly unresponsive to changes in the quasi-discount rate. Over the assumed discount rate range, a 1 percent change in the discount rate results, on average, in a 0.05 percent change in the dynamic rent maximising biomass. Further evidence of the insensitivity of the dynamic rent maximising biomass to changes in the discount rate is given by the coefficient of variation of the dynamic rent maximising biomass of 0.80 percent, versus a coefficient of variation of 16.58 percent for the discount rate. As an aside, for the purpose of modelling welfare effects, it should be noted that, unlike the case of price and cost variations, the dynamic maximum sustainable biomass is affected by changes in the quasi-discount rate, with the dynamic maximum sustainable yield biomass ranging between $650540 \geq X^{*}_{MSY} \geq 609511$ tons as the discount rate rises from $\delta_w = 3.3750$ percent to $\delta_w = 5.6260$ percent. However, as is the case with the dynamic maximum economic yield scenario, the dynamic maximum sustainable yield biomass is highly insensitive to fluctuations in the discount rate, evidenced by a responsiveness of 0.13 percent in the dynamic maximum sustainable yield biomass to every 1 percent change in the discount rate. Furthermore, the coefficient of variation of 0.80 percent recorded for X^{*}_{MEY} is substantially smaller than the 16.58 percent coefficient of variation recorded for δ_w.

Table 6.7: Dynamic Rent Maximising Biomass, Catch and Effort Levels in the West Coast Hake Fishery over the Assumed Feasible Quasi-Discount Rate Range

Quasi-Discount Rate (δ_w)		Biomass (X^{*}_{MEY})	Catch (C^{*}_{MEY})	Effort (E^{*}_{MEY})
3.3750		882 012	131 117	10 323
3.6000		879 872	131 315	10 364
3.8250		877 742	131 510	10 405
4.0500		875 620	131 701	10 445
4.2750		873 508	131 889	10 485
4.5000		871 406	132 074	10 525
4.7250		869 312	132 256	10 565
4.9500		867 228	132 434	10 605
5.1750		865 153	132 610	10 644
5.4000		863 088	132 782	10 684
5.6250		861 031	132 951	10 723
2.25	Range	20 980.15	1 834.38	399.49
4.50	Mean	871 452.08	132 058.12	10524.44
0.75	Standard Error	6 958.47	608.45	132.50
16.58	Coefficient of Variation (%)	0.80	0.46	1.26

With regard to catch, the dynamic rent maximising catch initially rises from the static maximum economic yield ($C_{MEY} = 127721$ tons), under a zero discount rate, approaching the static maximum sustainable yield ($C_{MSY} = 139034$ tons) as the dynamic rent maximising biomass falls with higher discount rates. However, once the dynamic rent maximising biomass has been reduced below the static maximum sustainable yield biomass, further discount rate increases result in a reduction in the size of the dynamic rent maximising catch. In the extreme, the dynamic rent maximising catch approaches the zero-rent catch ($\overline{C} = 113388$ tons) as the discount rate approaches infinity. The above results are summarised in Figure 6.11.

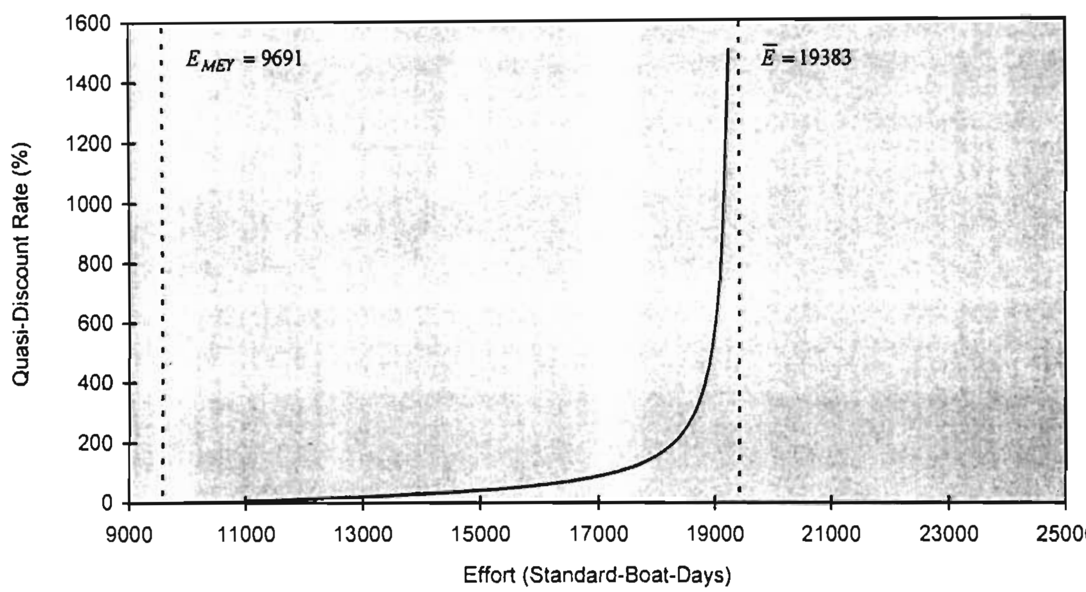


Two points should be noted with regard to the impact of quasi-discount rate changes on the dynamic rent maximising catch over the feasible discount rate range ($3.3750 \le \delta_w \le 5.6250$ percent). First, over the entire feasible discount rate range, the relationship between the dynamic rent maximising catch and the discount rate is positive, that is, higher discount rates increase the dynamic rent maximising catch. Second, the dynamic rent maximising catch in the west coast fishery is highly unresponsive to changes in the quasi-discount rate over the feasible range, with the dynamic rent maximising catch changing by 0.03 percent for each 1 percent change in the quasi-discount rate. Further evidence of the insensitivity of the dynamic rent maximising catch to changes in the discount rate is given by the coefficient of variation of the dynamic rent maximising catch of 0.46 percent, which is approximately one-thirtieth the size of the coefficient of variation of 16.58 percent recorded for the quasi-discount rate. As an aside, as is the case with biomass, the dynamic maximum sustainable yield is negatively related to the quasi-discount rate, ranging between $137996 \leq C^{*}_{MSY} \leq 136149$ tons as the discount rate varies between $3.3750 \leq \delta_{w} \leq 5.6250$ percent. The coefficient of variation on the dynamic maximum sustainable yield estimates of 0.45 percent is considerably lower than the coefficient of variation recorded for the quasi-discount rate of 16.58 percent.

Finally, the estimates produced by the dynamic bioeconomic model confirm the arguments set out in Section 6.5.1 that is, the dynamic rent maximising level of effort increases from the static maximum economic yield leve: $E^{*}_{MEY} = E_{MSY} = 9691$ standard-boat-days) under a zero discount rate ($\delta = 0$ percent), approaching the bionomic outcome ($E^{*}_{MEY} = \overline{E} = 19383$ standard-boat-days) as the discount rate approaches infinity. These results are summarised in Figure 6.12.

Over the feasible discount rate range ($3.3750 \leq \delta_{w} \leq 5.6250$ percent), the dynamic rent maximising effort level is highly unresponsive to changes in the quasi-discount rate: each 1 percent change in the discount rate produces an average 0.08 percent change in the dynamic rent maximising effort level. The relative insensitivity of the dynamic rent maximising effort level to variations in the discount rate is further confirmed by the coefficient of variation of the dynamic effort level (1.26 percent) being less than one-tenth the size of the coefficient of variation on the discount rate (16.58 percent) over the feasible range. It should be noted that, as is the case with biomass and catch, the level of effort required to take the dynamic maximum sustainable yield is influenced by the discount rate, ranging between $14731 \le E^{*}_{MSY} \le 15512$ standard-boat-days as the discount rate rises from $\delta_w = 3.33750$ percent to $\delta_w = 5.6250$ percent. However, the level of effort required to take the dynamic maximum sustainable yield is highly unresponsive to changes in the discount rate, varying by an average 0.1 percent for every 1 percent change in the discount rate.

Figure 6.12: Response of Dynamic Rent Maximising Effort in the West Coast Hake Fishery to Quasi-Discount Rate Variations



Therefore, while fluctuations in the quasi-discount rate result in fluctuations in model estimates of dynamic rent maximising biomass, catch and effort levels, these estimates are relatively unresponsive to changes in the discount rate. The implications for welfare levels and management policies are considered below in Section 6.5.3.

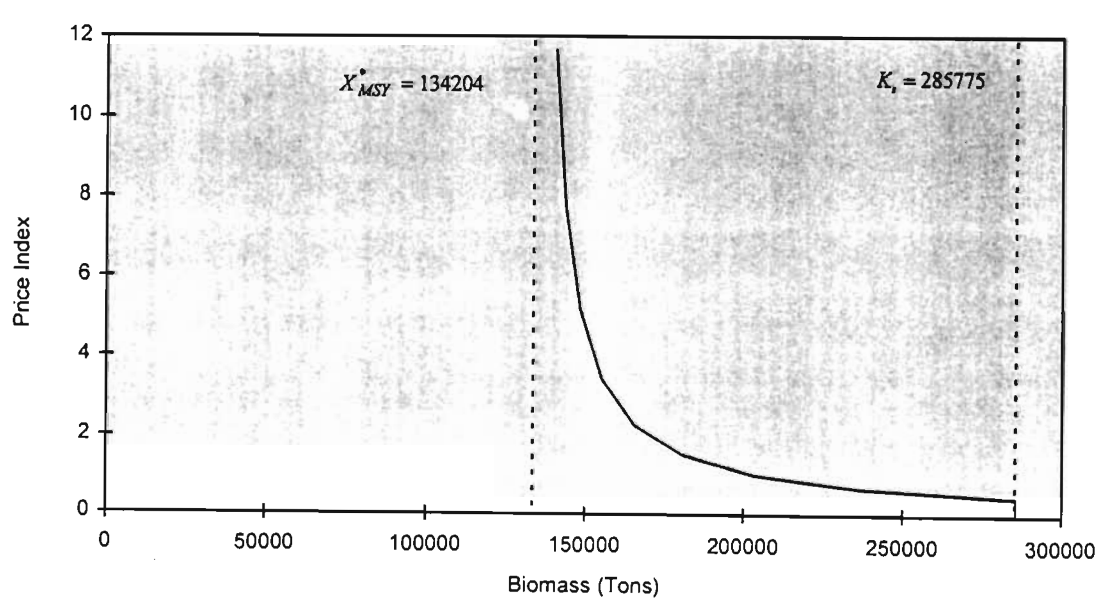
6.5.2.2 The South Coast Hake Fishery

As is the case with the west coast hake fishery, estimates produced by the dynamic bioeconomic model for the south coast fishery confirm the arguments set out in Section 6.5.1 regarding the impact of the price, cost and quasi-discount rate parameters on dynamic rent maximising biomass, catch and effort levels.

(a) Changes in Price

As modelled in Section 6.5.1, the dynamic rent maximising biomass falls, at a decreasing rate, below the carrying capacity biomass ($K_s = 285775$ tons) as price rises above the minimum feasible harvest price, where $p_{\min} = 0.4549$ in the south coast hake fishery. In the extreme, the dynamic rent maximising biomass asymptotically approaches the discounted maximum sustainable yield biomass ($X^*_{MSY} = 134204$ tons) as price approaches infinity. These results are summarised in Figure 6.13.

Figure 6.13: Response of Dynamic Rent Maximising Biomass in the South Coast Hake Fishery to Price Variations



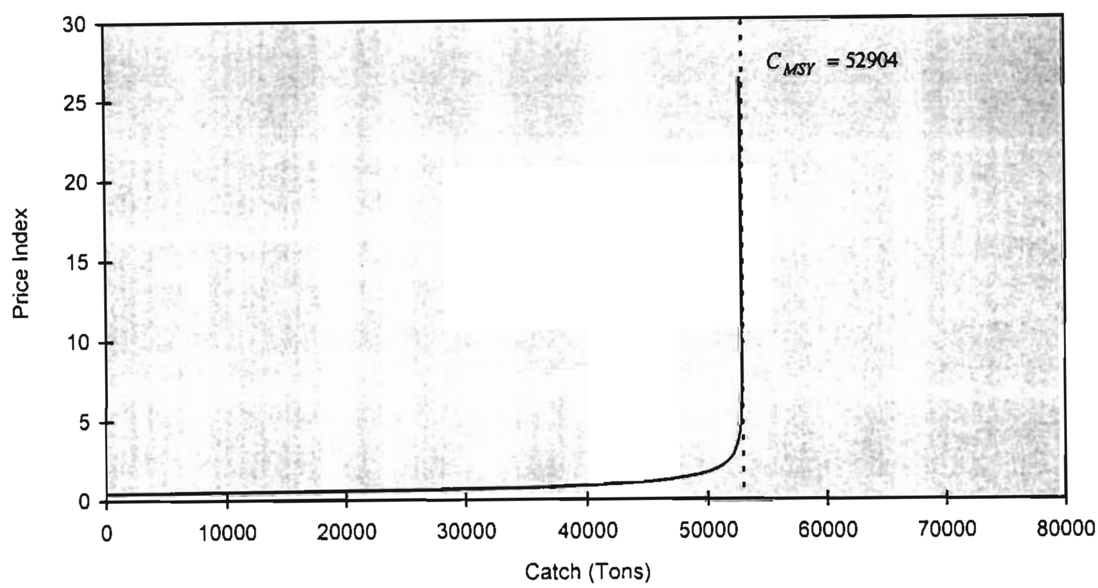
The dynamic rent maximising biomass in the south coast fishery is highly unresponsive to changes in the price level over the feasible price range, $0.75 \le p_s \le 1.25$. As shown in Table 6.8, a 1 percent change in the price level results in an average 0.35 percent change in the dynamic rent maximising biomass. The insensitivity of the dynamic rent maximising biomass to changes in the price level is confirmed by a coefficient of variation of the dynamic rent maximising biomass which is approximately one-third the size of the coefficient of variation (16.58 percent) recorded for the price level. As an aside, it is worthwhile noting that the dynamic maximum sustainable biomass is unaffected by the price level ($X^{*}_{MSY} = 134204$ tons for all p_s).

Table 6.8: Dynamic Rent Maximising Biomass, Catch and Effort Levels in the South Coast Hake Fishery over the Assumed Feasible Price Range

Price (p_s)		Biomass (X^{*}_{MEY})	Catch (C^{*}_{MEY})	Effort (E^{*}_{MEY})
0.7500		227 487	34 359	37 759
0.8000		221 816	36 762	41 433
0.8500		216 800	38 748	44 682
0.9000		212 333	40 407	47 575
0.9500		208 329	41 807	50 170
1.0000		204 718	42 998	52 509
1.0500		201 446	44 019	54 628
1.1000		198 466	44 900	56 559
1.1500		195 741	45 666	58 324
1.2000		193 239	46 335	59 945
1.2500		190 934	46 922	61 438
0.50	Range	36 553	12 564	23 679
1.00	Mean	206 482.63	42 083.85	51 365.59
0.16	Standard Error	12 023.65	4 108.98	7 788.92
16.58	Coefficient of Variation (%)	5.82	9.76	15.16

Initially, for prices above $p_{\min} = 0.4549$, the dynamic rent maximising catch increases with price, approaching the static maximum sustainable yield ($C_{MSY} = 52904$ tons). However, once prices have risen sufficiently to reduce the dynamic rent maximising biomass to below the static maximum sustainable yield biomass ($X_{MSY} = 142888$ tons), the dynamic rent maximising catch falls, asymptotically approaching the dynamic maximum sustainable yield ($C^*_{MSY} = 52709$ tons) as price approaches infinity. The net result is a 'backward-bending' supply of output curve for the south coast fishery. However, the price level at which the price-output relationship becomes negative ($p = 8.00$) is sufficiently high to render the backward-bending supply of output curve practically useless (Figure 6.14).

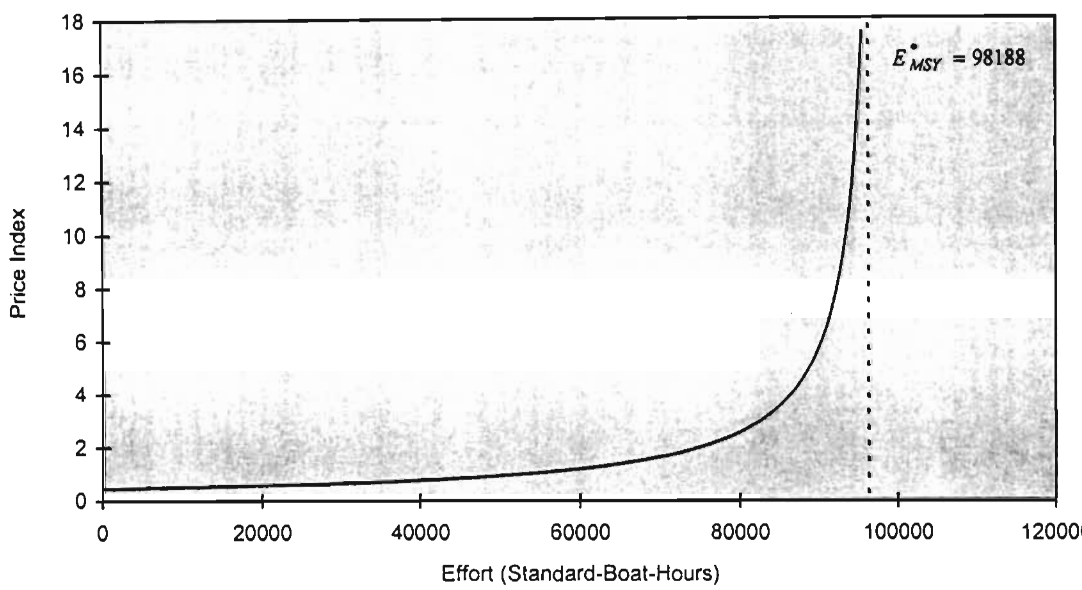
Figure 6.14: Response of Dynamic Rent Maximising Catch in the South Coast Hake Fishery to Price Variations



The dynamic rent maximising catch in the west coast fishery is highly unresponsive to changes in the price level over the feasible price range, $0.75 \leq p_s \leq 1.25$. As presented in Table 6.8, a 1 percent change in the price level produces an average 0.62 percent change in the dynamic rent maximising catch across the feasible price range. The insensitivity of the dynamic rent maximising catch to changes in the price level is further evidenced by a coefficient of variation of the dynamic rent maximising catch of 9.76 percent, which is approximately two-thirds the size of the coefficient of variation of the price level. As is the case with biomass, the dynamic maximum sustainable yield in the south coast fishery is unaffected by the price level ($C^*_{MSY} = 52709$ tons for all p_s).

With regard to effort, as price rises from $p_{\min} = 0.4549$, effort increases from $E^* = 0$ standard-boat-hours, asymptotically approaching that level of effort required to take the dynamic maximum sustainable catch ($E^*_{MSY} = 98188$ standard-boat-hours) as price approaches infinity. These results are summarised in Figure 6.15.

Figure 6.15: Response of Dynamic Rent Maximising Effort in the South Coast Hake Fishery to Price Variations



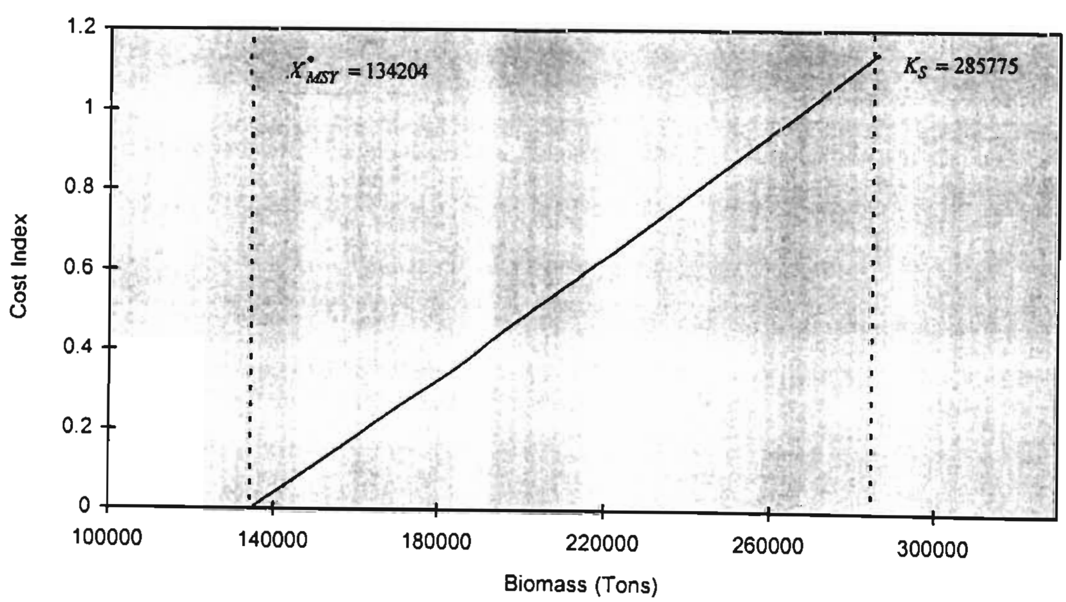
The results of the estimation procedure reveal that, over the feasible price range, a 1 percent change in the price level produces an average 0.95 percent change in the dynamic rent maximising effort (Table 6.8). As such, the dynamic rent maximising effort level is found to vary closely with price. The closeness of this relationship is reinforced by the coefficient of variation of 15.16 percent on the dynamic rent maximising effort level being roughly equal to the coefficient of variation on the price index. In passing, it should be noted that, as is the case with biomass and catch, the level of effort required to take the dynamic maximum sustainable yield is unaffected by the price level ($E^*_{MSY} = 98188$ standard-boat-hours for all p_s.).

Thus, it may be concluded that, while price fluctuations result in changes in the dynamic rent maximising biomass, catch and effort levels over the so-called feasible price range, with the exception of effort levels, the model estimates are highly insensitive to price fluctuations. Fluctuations in the dynamic rent maximising level of effort are closely correlated with changes in price. The implications of this result for welfare outcomes and for management policies are examined in Section 6.5.3.

(b) Changes in Cost

As modelled in Section 6.5.1, the dynamic rent maximising biomass rises from the dynamic maximum sustainable yield biomass ($X^*_{MSY} = 134204$ tons) to the carrying capacity biomass ($K_s = 285775$ tons) as cost rises from zero ($c = 0$) to infinity. However, fishing becomes unfeasible at a cost level of $c = c_{\max} = 1.1431$ in the south coast fishery. These results are summarised in Figure 6.16.

Figure 6.16: Response of Dynamic Rent Maximising Biomass in the South Coast Hake Fishery to Cost Variations



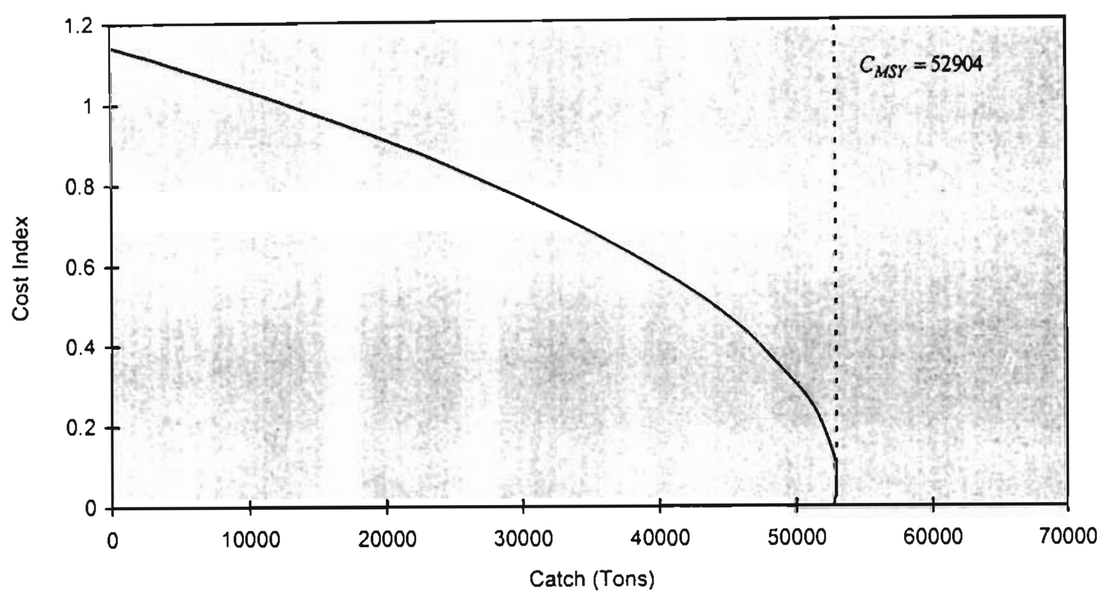
Across the feasible cost range, $0.3900 \le c_s \le 0.6500$, the dynamic rent maximising biomass in the west coast fishery is highly unresponsive to changes in the price, with a 1 percent change in the cost index resulting, on average, in a 0.32 percent change in the dynamic rent maximising biomass. Evidence of the insensitivity of the dynamic rent maximising biomass to changes in the price level is also given by the coefficient of variation of the dynamic rent maximising biomass of 5.56 percent, which is approximately one-third the coefficient of variation of 16.58 percent for the cost index. These figures are given in Table 6.9 below. As an aside, it should be noted that the dynamic maximum sustainable biomass is unaffected by changes in the cost index ($X^*_{MSY} = 134204$ tons for all $0 \le c_s \le c_{\max}$).

Table 6.9: Dynamic Rent Maximising Biomass, Catch and Effort Levels in the South Coast Hake Fishery over the Assumed Feasible Cost Range

Cost (c_s)		Biomass (X^*_{MEY})	Catch (C^*_{MEY})	Effort (E^*_{MEY})
0.3900		187 470	47 754	63 682
0.4160		190 934	46 922	61 438
0.4420		194 390	46 031	59 199
0.4680		197 839	45 079	56 965
0.4940		201 282	44 068	54 735
0.5200		204 718	42 998	52 509
0.5460		208 149	41 868	50 286
0.5720		211 573	40 680	48 068
0.5980		214 992	39 432	45 853
0.6240		218 406	38 126	43 641
0.6500		221 816	36 762	41 433
0.2600	Range	34 345	10 992	2 249
0.5200	Mean	204 688.19	42 701.87	52 528.02
0.09	Standard Error	11 389.82	3 656.78	7 378.33
16.58	Coefficient of Variation (%)	5.56	8.56	14.05

The dynamic rent maximising catch initially rises with cost, although at a decreasing rate, from the dynamic maximum sustainable yield ($C^*_{MSY} = 52709$ tons) under zero cost conditions ($c_s = 0$), to approach the static maximum sustainable yield $C_{MSY} = 52904$ tons. Once costs have risen sufficiently ($c_s \geq 0.0798$) to raise the dynamic rent maximising biomass above the static maximum sustainable yield biomass ($X_{MSY} = 142888$ tons), the dynamic optimum catch falls at an increasing rate to approach zero as the cost index approaches $c_{\max} = 1.1431$ (Figure 6.17).

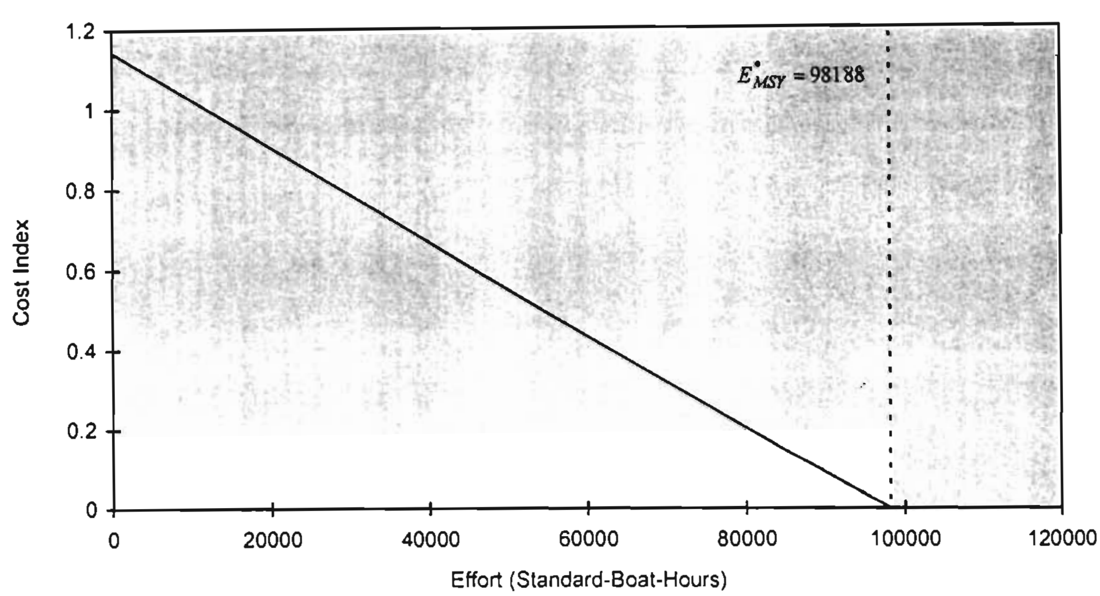
Figure 6.17: Response of Dynamic Rent Maximising Catch in the South Coast Hake Fishery to Cost Variations



Two points should be noted with regard to the impact of costs on the dynamic rent maximising catch in the south coast fishery across the feasible cost range of $0.3900 \leq c_s \leq 0.6500$. First, the relationship between the dynamic rent maximising catch and the cost index is negative over the entire feasible cost range, that is, higher costs reduce the dynamic rent maximising catch. Second, the dynamic rent maximising catch is highly unresponsive to changes in the cost index: a 1 percent change in the cost index produces an average 0.52 percent change in the dynamic rent maximising catch. Further evidence of the insensitivity of the dynamic rent maximising catch to changes in the price level is given by the coefficient of variation of the dynamic rent maximising catch which is approximately half as great as the coefficient of variation of the cost index level. As is the case with biomass, the dynamic maximum sustainable yield is unaffected by the cost level: $C^*_{MSY} = 52904$ tons for all c_s.

With regard to effort, in line with the arguments set out in Section 6.5.1, as costs rise from zero to $c_{\max}$, dynamic rent maximising effort decreases, at an increasing rate, from that level required to take the dynamic maximum sustainable catch ($E^*_{MSY} = 98188$ standard-boat-hours), with fishing becoming unfeasible as cost rises to $c_{\max}$. These results are summarised in Figure 6.18.

Figure 6.18: Response of Dynamic Rent Maximising Effort in the South Coast Hake Fishery to Cost Variations



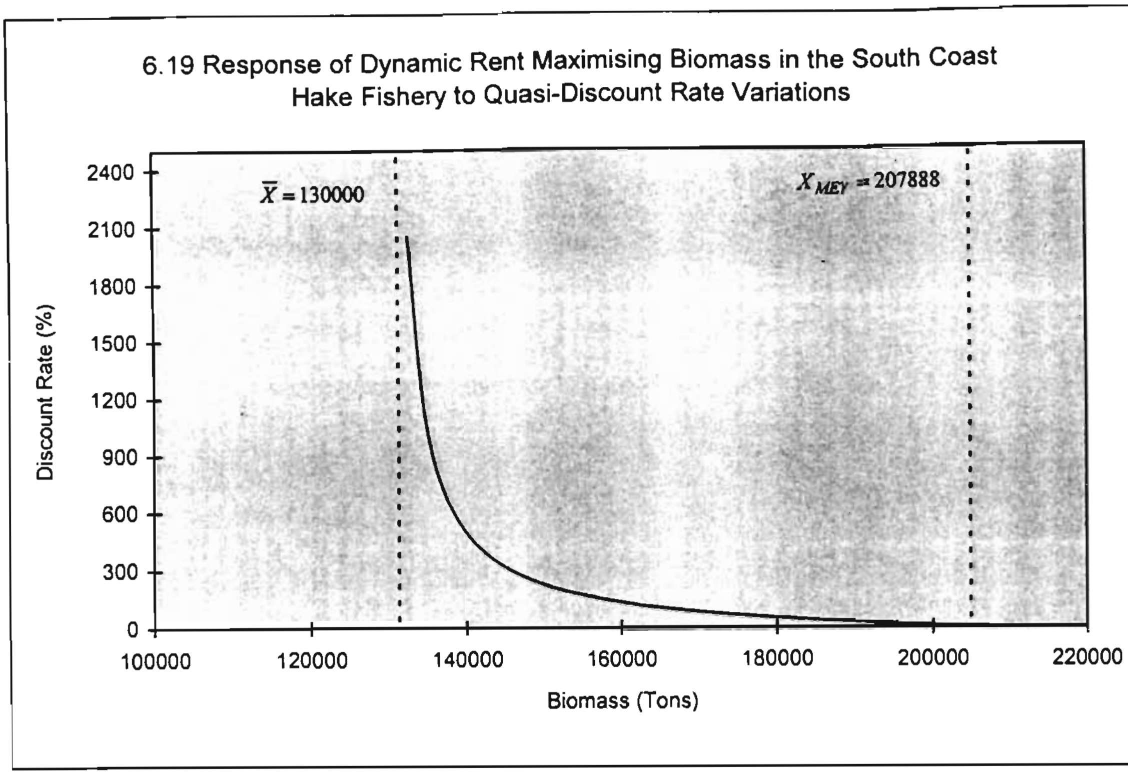
Across the feasible cost range, $0.3900 \le c_s \le 0.6500$, the dynamic rent maximising effort level in the south coast fishery is relatively closely correlated with changes in the cost index: a 1 percent change in the cost index produces an average 0.85 percent change in the dynamic rent maximising effort. This result is confirmed by the coefficient of variation on the dynamic rent maximising effort level (14.05 percent) being marginally less than that recorded for cost (16.58 percent) across the feasible range. It may also be noted that, as is the case with biomass and catch, the level of effort required to take the dynamic maximum sustainable yield in the south coast fishery is unaffected by the cost level ($E^*_{MSY} = 98188$ standard-boat-hours for all c_s).

In summary, fluctuations in the cost level result in extremely modest changes in the dynamic rent maximising biomass and catch over the so-called feasible range. The rent maximising effort level, however, is somewhat more sensitive to changes in cost, displaying a correlation of close to one with the cost index. That aside, it is concluded that, overall, model estimates are relatively insensitive to changes in costs. The implications of this result for welfare outcomes and for management policies are considered in Section 6.5.3.

(c) Changes in the Quasi-Discount Rate

Considering changes in the quasi-discount rate, the dynamic rent maximising biomass in the south coast fishery decreases as the discount rate rises, falling from a level equal to the static maximum economic yield biomass ($X_{MEY} = 207888$ tons) for $\delta_s = 0$ percent, and approaching the zero rent biomass ($\overline{X} = 130000$ tons) as the discount rate rises to infinity (Figure 6.19).

More important, however, is the result that the dynamic rent maximising biomass in the south coast fishery is highly unresponsive to changes in the quasi-discount rate over the feasible range. Specifically, over the range $3.3750 \leq \delta_s \leq 5.6250$ percent, a 1 percent change in the discount rate results, on average, in a 0.02 percent change in the dynamic rent maximising biomass. The insensitivity of the dynamic rent maximising biomass to changes in the discount rate is further evidenced by a coefficient of variation of the discount rate that is over sixty times as great as that for the dynamic rent maximising biomass.



For the purpose of modelling welfare effects, it should be noted that, in contradiction to price and cost variations, the dynamic maximum sustainable biomass is affected by changes in the quasi-discount rate, with the dynamic maximum sustainable yield biomass falling from $136365 \geq X^*_{MSY} \geq 132024$ tons as the discount rate rises from $3.3750 \leq \delta_s \leq 5.6250$ percent. However, as is the case with the dynamic maximum economic yield scenario, the dynamic maximum sustainable yield biomass is highly insensitive to fluctuations in the discount rate, evidenced by a coefficient of variation of 1.07 percent (versus a value of 16.58 percent recorded for the quasi-discount rate), and a responsiveness of 0.06 percent in the dynamic maximum sustainable yield biomass to every 1 percent change in the discount rate.

The dynamic rent maximising catch initially rises from the static maximum economic yield ($C_{MEY} = 207888$ tons), under a zero discount rate, approaching the static maximum sustainable yield ($C_{MSY} = 52904$ tons) as the dynamic rent maximising biomass falls with higher interest rates. However, once the dynamic rent maximising biomass falls below the static maximum sustainable yield biomass ($X_{MSY} = 142888$ tons), further discount rate increases result in a reduction in the size of the dynamic rent maximising catch. In the extreme, the dynamic rent maximising catch approaches the zero-rent yield ($\overline{C} = 52474$ tons) as the discount rate approaches infinity (Figure 6.20).

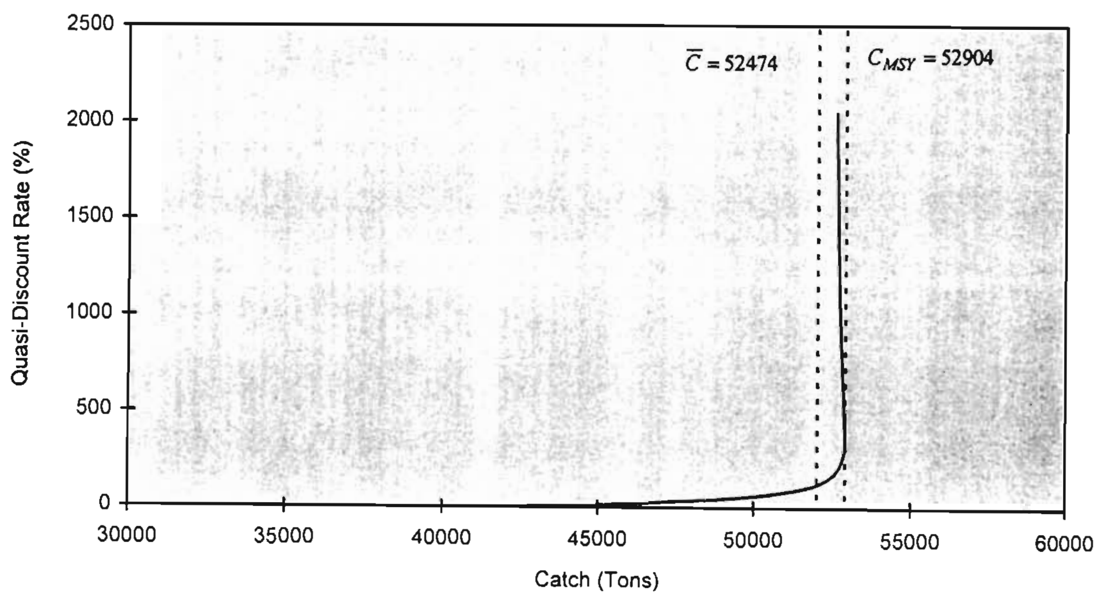
Table 6.10: Dynamic Rent Maximising Biomass, Catch and Effort Levels in the West Coast Hake Fishery over the Assumed Feasible Quasi-Discount Rate Range

Quasi-Discount Rate (δ_s)		Biomass ($X^{\bullet}_{MEY}$)	Catch ($C^{\bullet}_{MEY}$)	Effort ($E^{\bullet}_{MEY}$)
3.3750		205 495	42 747	52 005
3.6000		205 339	42 798	52 107
3.8250		205 183	42 848	52 207
4.0500		205 028	42 898	52 308
4.2750		204 873	42 948	52 408
4.5000		204 718	42 998	52 509
4.7250		204 564	43 047	52 608
4.9500		204 410	43 096	52 708
5.1750		204 257	43 145	52 807
5.4000		204 104	43 194	52 906
5.6250		203 952	43 242	53 005
2.25	Range	1543	494	1000
4.50	Mean	204 720.34	42 996.55	52 507.20
0.75	Standard Error	511.78	164.00	331.53
16.58	Coefficient of Variation (%)	0.25	0.38	0.63

Two points should be noted with regard to the impact of changes in the quasi-discount rate on the dynamic rent maximising catch across the feasible discount rate range ($3.3750 \leq \delta_s \leq 5.6250$ percent). First, over the entire feasible discount rate range, the relationship between the dynamic rent maximising catch and the discount rate in the south coast fishery is positive, with the dynamic rent maximising catch increasing from $C^{*}_{MEY} = 42747$ tons to

$C^{*}_{MEY} = 43242$ tons as the quasi-discount rate rises from $\delta_s = 3.3750$ percent to $\delta_s = 5.6250$ percent. Second, and of greater interest, the dynamic rent maximising catch in the south coast fishery is highly unresponsive to changes in the quasi-discount rate over the feasible range: a 1 percent change in the quasi-discount rate produces an average 0.02 percent change in the dynamic rent maximising catch. This result is supported by a coefficient of variation on the dynamic rent maximising dynamic catch one-fortieth the size of the coefficient of variation recorded for the quasi-discount rate.

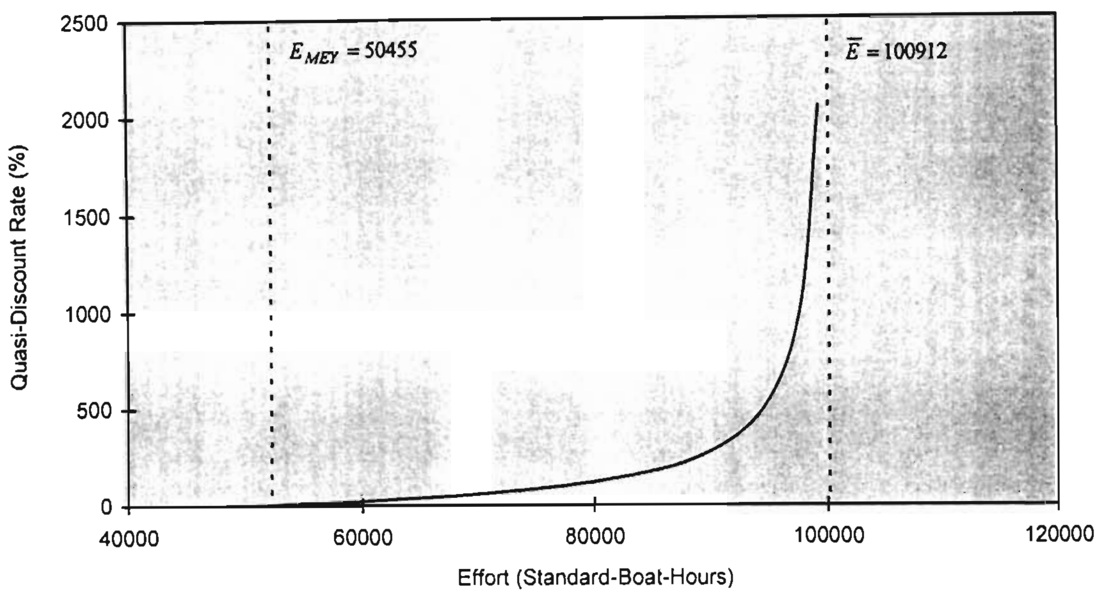
Figure 6.20: Response of Dynamic Rent Maximising Catch in the South Coast Hake Fishery to Quasi-Discount Rate Variations



As is the case with biomass, the dynamic maximum sustainable yield is negatively related to the quasi-discount rate, ranging between $52794 \leq C^{*}_{MSY} \leq 52598$ tons as the discount rate varies between $3.3750 \leq \delta_s \leq 5.6250$ percent. The coefficient of variation on the dynamic maximum sustainable yield estimates of 0.12 percent is considerably lower than the coefficient of variation calculated for the quasi-discount rate of 16.58 percent.

Finally, as modelled in Section 6.5.1, the dynamic rent maximising level of effort in the south coast fishery increases from the static maximum economic yield level ($E^{*}_{MEY} = E_{MSY} = 50455$ standard-boat-hours) under a zero discount rate ($\delta_s = 0$ tons), approaching the bionomic outcome ($E^{*}_{MEY} = \overline{E} = 100912$ standard-boat-hours) as the discount rate approaches infinity. These results are summarised in Figure 6.21.

Figure 6.21: Response of Dynamic Rent Maximising Effort in the South Coast Hake Fishery to Quasi-Discount Rate Variations



Considering the effects of fluctuations in the discount rate over the feasible range ($3.3750 \le \delta_s \le 5.6250$ percent), the results presented in Table 6.10 reveal that the dynamic rent maximising biomass is highly unresponsive to changes in the quasi-discount rate. Specifically, across the feasible range, a 1 percent change in the discount rate produces an average 0.04 percent change in the dynamic rent maximising effort. The relative insensitivity of the dynamic rent maximising effort level to variations in the discount rate is further confirmed by the coefficient of variation of the dynamic effort level (0.63 percent) being considerably smaller than the coefficient of variation on the discount rate (16.58 percent) across the feasible range.

As an aside, as is the case with biomass and catch, the level of effort required to take the dynamic maximum sustainable yield is correlated with the discount rate, ranging between $96788 \leq E^{*}_{MEY} \leq 99600$ standard-boat-hours as the discount rate rises from $\delta_s = 3.33750$ percent to $\delta_s = 5.6250$ percent. However, the level of effort required to take the dynamic maximum sustainable yield is relatively unresponsive to changes in the discount rate, varying by an average 0.06 percent for every 1 percent change in the quasi-discount rate.

In short, while fluctuations in the discount rate influence model estimates of dynamic rent maximising biomass, catch and effort levels, these estimates are relatively unresponsive to changes in the discount rate. The implications for welfare levels and management policies are considered below in Section 6.5.3.

6.5.3 Sensitivity Analysis: The Impact of Variable Economic Parameter Estimates on Welfare Levels

Based on the dynamic rent maximising catch and effort figures set out above, as well as the dynamic maximum sustainable yield estimates, it is possible to arrive at a number of important conclusions about social welfare under the assumption of variable economic parameters. To start with, under each of the variable parameter scenarios, that is, variable price, cost and discount rate scenarios, the rent estimates under the dynamic maximum economic yield strategy are, without exception, considerably higher than those generated under the dynamic maximum sustainable yield strategy. In the west coast fishery, the mean value of rents generated by the dynamic maximum economic yield estimates are, on average, 44.58 percent greater than those generated by the dynamic maximum sustainable yield outcome. In the south coast fishery, the dynamic rent maximising catch and effort levels produce rents which are, on average, 862.99 percent greater than those generated under the dynamic maximum sustainable yield scenario. These results are summarised in Table 6.11. A detailed breakdown of the calculations of rents under the various parameter and management strategy scenarios is given in Appendix 4.

Table 6.11: A Comparison of the Mean Values of Economic Rents Generated under the Dynamic Maximum Economic Yield and Dynamic Maximum Sustainable Yield Strategies under the Assumption of Variable Economic Parameter Values

West Coast Hake Fishery			
Variable Parameter Scenario	Dynamic Maximum Economic Yield ($X^{\bullet}_{MEY}$)	Dynamic Maximum Sustainable Yield ($X^{\bullet}_{MSY}$)	Welfare Gain (%) ($X^{\bullet}_{MEY}$ versus $X^{\bullet}_{MSY}$)
	Index Points		
Price	70 787.43	48 724.30	45.28
Cost	70 773.43	48 724.30	45.26
Quasi-Discount Rate	70 490.56	49 223.90	43.20

South Coast Hake Fishery			
Variable Parameter Scenario	Dynamic Maximum Economic Yield ($X^{\bullet}_{MEY}$)	Dynamic Maximum Sustainable Yield ($X^{\bullet}_{MSY}$)	Welfare Gain (%) ($X^{\bullet}_{MEY}$ versus $X^{\bullet}_{MSY}$)
	Index Points		
Price	15 978.59	1 651.24	867.67
Cost	15 965.63	1 651.24	866.89
Quasi-Discount Rate	15 692.77	1 644.22	854.42

A number of further points should be made with regard to the above results. First, the considerable gains in mean rents achieved by shifting the south coast fishery from the dynamic maximum economic yield outcome to dynamic sustainable yield outcome are, in part, attributable to the fact that, for 'low' price and 'high' cost scenarios, the dynamic maximum sustainable yield rents are negative. Furthermore, while the potential increase in rents in the south coast fishery is considerably more dramatic than that for the west coast fishery, the south coast fishery accounts for a significantly smaller portion of total rents (between approximately 3.5 percent and 22.5 percent under the dynamic maximum sustainable yield and dynamic maximum economic yield strategies respectively). None the less, the increase in total rents achieved by shifting the west and south coast fisheries from biological to bioeconomic outcomes is considerable, amounting to an average 71.28 percent across the two fisheries.

Second, the rent estimates generated under the dynamic maximum economic yield scenario are relatively stable under variable economic parameter conditions, evidenced by coefficients of variation of rents equal to one-quarter and one-half the value of those calculated for economic parameters (as set out in Appendix 4). The importance of this result cannot be over-emphasised given the problems involved in providing accurate values for parameter estimates; the evidence suggesting that parameter values are likely to change across time; and the desirability of employing bioeconomic management targets as the basis for raising the level of social welfare.

Therefore, it can be concluded that, in spite of economic parameter fluctuations, the management implications of the dynamic bioeconomic model remain unaltered. Specifically, the rents generated under the dynamic maximum economic yield strategy are, without exception, substantially greater than those generated under the dynamic maximum sustainable yield strategy. Moreover, the bioeconomic rent estimates generated by the dynamic bioeconomic model are found to be robust which, while not a necessary outcome, is highly desirable from a management perspective. In short, the dynamic rent maximising outcome generates considerable gains in social welfare versus the dynamic maximum sustainable yield outcome. Moreover, in proving relatively stable in the face of economic parameter fluctuations, the dynamic bioeconomic model is capable of providing fisheries managers with a high degree of confidence in the targets that are set as the basis for achieving the above-mentioned gains in economic rents.

CHAPTER 7: POLICY IMPLICATIONS

7.1 INTRODUCTION

The arguments presented in Chapters 5 and 6 allow two important conclusions to be reached in connection with the South African hake fishery. First, harvesting at a biological optimum – such as dynamic maximum sustainable yield – is economically sub-optimal. Second, adopting a dynamic maximum economic yield strategy, consistent with the goal of welfare maximisation, would generate the highest possible rents in the hake fishery. However, South Africa's fisheries managers have generally ignored bioeconomic principles as the basis for management of fish stocks, and the hake fishery is no exception. As noted in Chapter 4, the west and south coast hake fisheries are managed on the basis of an $f_{0.2}$ principle which, although conservative, is none the less heavily geared towards the biological principle of maximum sustainable yield. Nevertheless, this provides for two main arguments. First, given that the hake fishery is managed on the basis of the 'conservative' $f_{0.2}$ strategy, it follows that the fishery is likely to be 'closer' to the dynamic maximum economic yield outcome than if it was managed solely on the basis of maximum sustainable yield. Therefore, and second, the gains to be had from shifting the fishery to the dynamic rent maximising position are likely to be different (lower) than those derived from shifting the fishery from the dynamic maximum sustainable yield strategy (as outlined in Chapter 6).

This chapter has two main objectives. First, Section 7.2 establishes the magnitude of the gains in rent to be had from shifting the fishery from the extant ($f_{0.2}$) position to the dynamic rent maximising outcome. Second, given the result that the management targets produced by the dynamic bioeconomic model are robust, and that the potential gains in rent to be had are considerable, Section 7.3 goes on to examine how the fishery is best shifted to, and kept at, the dynamic rent maximising outcome.

7.2 Rent Gains under the Dynamic Maximum Economic Yield Strategy: Dynamic Maximum Sustainable Yield versus $F_{0.2}$

Comparing the dynamic rent maximising outcome with the dynamic maximum sustainable yield outcome provides a useful first guide to policy formulation. However, from a practical stance, it is more reasonable to compare the extant position of the fishery (rather than a hypothetical maximum sustainable yield position) to the dynamic rent maximising outcome as the basis for establishing potential welfare gains. Given that South Africa's fisheries managers have adopted an $f_{0.2}$ strategy which is deliberately more conservative than a maximum sustainable yield strategy,[1] it is argued that the South African hake fishery is no exception. The implications of this are that shifting the west and south coast hake fisheries from extant positions to the dynamic maximum economic yield positions is presumably less disruptive than would otherwise be the case, and that the gains in rent are, accordingly, lower than those hypothesised in Chapter 6.

Table 7.1 provides a summary of the biomass, catch and effort data for four scenarios: the mean of the modelled dynamic maximum economic yield outcome; the mean of the modelled dynamic maximum sustainable yield outcome; the observed average for the period 1984-95, as generated under the $f_{0.2}$ harvesting strategy; and the observed figures for 1995, as generated under the $f_{0.2}$ harvesting strategy. The economic rent values associated with each of the scenarios are also shown.

[1] As noted in Chapter 4, the extant 'biologically conservative' $f_{0.2}$ management strategy involves driving the fishery to the so-called $B_{0.2}$ biomass, equal to approximately 1 000 000 tons on the west coast and 165 000 tons on the south coast.

Table 7.1: Biomass, Catch and Effort Figures under Modelled and Observed Scenarios

	Modelled		Observed	
	Dynamic MEY ($X^{\bullet}_{MEY}$)	Dynamic MSY ($X^{\bullet}_{MSY}$)	Average (1984-95)	1995
West Coast Fishery	Index Points			
Biomass (tons)	878 765	621 782	614 670	778 000
Catch (tons)	125 714	130 893	93 517	96 600
Effort (standard-boat-days)	10 306	13 320	15 074	14 500
Revenue index	125 714.00	130 893.00	93 517.00	96 600.00
Cost index	60 290.10	77 922.00	88 182.90	84 825.00
Rent index	65 423.90	52 971.00	5 334.10	11 775.00

	Modelled		Observed	
	Dynamic MEY ($X^{\bullet}_{MEY}$)	Dynamic MSY ($X^{\bullet}_{MSY}$)	Average (1984-95)	1995
South Coast Fishery	Index Points			
Biomass (tons)	205 169	134 146	150 417	165 000
Catch (tons)	42 709	52 478	49 408	45 800
Effort (standard-boat-hours)	52 319	98 263	79 850	70 500
Revenue index	42 709.00	52 478.00	49 408.00	45 800.00
Cost index	27 205.88	51 096.76	41 522.00	36 660.00
Rent index	15 503.12	1 381.24	7 886.00	9 140.00

The above results allow for a number of conclusions. As already established, both the west and south coast fisheries are in positions other than the dynamic maximum sustainable yield outcome, and this has generally been the case over the entire sample period 1984-95. Accordingly, the implications in terms of potential welfare gains, as well as the reduction in effort required to generate the dynamic rent maximising yield, are somewhat different to those hypothesised in Chapter 6. At the same time, it should be noted that while South Africa's hake fishery managers have set biomass targets (with associated catch and effort targets) based on the $f_{0.2}$, strategy, as at 1995, only the south coast fishery satisfied this target. The west coast fishery biomass – of 778 000 tons – was some 25 percent below the target $B_{0.2}$ biomass of 1 000 000 tons. Nevertheless, the combined west and south coast $B_{0.2}$ biomass of 1 165 000 tons is extremely close to the dynamic rent maximising biomass of approximately 1 100 000 tons (depending upon the selected parameter scenario). However, this is argued to be more a result of coincidence than design, in that the rationale for the $f_{0.2}$ strategy is 'conservative biological management' rather than 'optimal bioeconomic management'. Moreover, upon disaggregation, the dynamic rent maximising outcome and $f_{0.2}$ strategies diverge by considerably more than the modest difference suggested above (see Table 7.1).

Returning to the task at hand, namely establishing the welfare effects of shifting the fishery from the extant (1995) position to the rent maximising position, a number of points should be made. To start with, as set out in Table 7.1, the modelling procedure suggests that shifting the west coast fishery from the 1995 position to the dynamic maximum economic yield outcome would require a 28.92 percent reduction in effort from the 1995 level of 14 500 standard-boat-days to 10 306 standard-boat-days per annum. At the same time, total catch would rise from the 1995 rate of 96 600 tons per annum to 125 714 tons per annum, an increase of 30.14 percent. In turn, the composite effect of falling effort and rising catch would be a more than five-fold increase in the rent index, from the 1995 level of 11 775.00 index points to the dynamic maximum economic rent level of 65 423.90 index points. This increase in rent from the extant (1995) position to the dynamic maximum economic yield position is considerably greater than the 23.51 percent increase that is shown in Chapters 5 and 6 to be achievable by shifting the west coast hake fishery from a 'hypothetical' dynamic maximum sustainable yield position to the dynamic rent maximising position.

Shifting the south coast fishery from the 1995 position to the dynamic maximum economic yield position would require a 25.79 percent reduction in effort from 70 500 standard-boat hours to 52 319 standard-boat-hours per annum. The annual catch would also decline, falling marginally from 45 800 tons to 42 709 tons. However, the net effect of the fall in effort and catch would be a 69.62 percent increase in the level of rent generated by the fishery, from the 1995 level of 9 140.00 index points to the dynamic maximum economic rent level of 15 503.12 index points. The increase in rent achieved by shifting the south coast hake fishery from the extant (1995) position to the dynamic rent maximising position is considerably less than the 1 022.41 percent increase in rent shown in Chapters 5 and 6 to be achievable by shifting the fishery from the 'hypothetical' dynamic maximum sustainable yield position to the dynamic maximum economic yield position.

The above set of findings allows for two main results. First, the gains in rent achieved by shifting the hake fishery from the extant (1995) position to the dynamic maximum economic yield position are considerable. Indeed, in the case of the west coast fishery, for example, the gains in rent to be had by shifting the fishery from the extant position to the dynamic rent maximising outcome are more than five times greater than those to be had from shifting the fishery from a hypothetical dynamic maximum sustainable yield position to the dynamic rent maximising outcome. To be more specific, shifting the west and south coast fisheries from their extant (1995) positions to the dynamic maximum economic yield position produces a four-fold increase in combined rents generated by the west and south hake fisheries from 20 914.00 index points to 80 927.02 index points. Second, irrespective of the relative size of rent increases, the primary result holds: welfare maximisation, by definition and by the above results, demands shifting the hake fisheries to the dynamic rent maximising position. Based on this conclusion, the chief concern of the remainder of this chapter lies with reviewing the various instruments available to fisheries managers as a basis for providing some guide as to how the goal of welfare maximisation may best be achieved and maintained.

7.3 Regulation of the Hake Fisheries

7.3.1 The Goals of Regulation

Fisheries management is constantly confronted with (and sometimes confounded by) the complexities of real-world fisheries systems. Two major sources of such complexities are: (i) the many conflicting goals and objectives faced by fisheries managers; and (ii) the wide variety of socio-economic factors which impact directly on the validity and effectiveness of regulatory instruments (John, 1994). In line with this, this section of this study aims to propose a set of policies, commensurate with the goal of welfare maximisation but, at the same time, capable of incorporating a wider array of policy goals, such as the improvement of equity; and suggest the most appropriate regulatory framework for achieving these goals.

Turning first to an examination of policy options, it must be recognised at the outset that fisheries management potentially encompasses a host of policy goals. For example, Lawson (1984, 157) listed fourteen common goals of fisheries development, including increases to production, employment and fishers' incomes, industry diversification, skills development and the encouragement of both exports and domestic consumption. A report of the Food and Agricultural Organisation (1983, 20) suggested that a similar list of objectives 'could be placed in three groups: maintaining the resources, economic performance, and equity (or social needs)'. In this vein, Clark (1985, 144) provided a more detailed list of possible objectives of fisheries regulation, grouped according to biological and economic (efficiency and equity) goals. Biological goals are said to include the conservation of fish stocks, maintenance of ecosytems, maximisation of sustainable catch, stabilisation of stock levels, prevention of waste and promotion of the use of under-utilised stocks. Economic goals are stated to include efficiency considerations – maximisation of sustainable rents, stabilisation of biomass and catch rates, an increase in fishers' incomes, prevention of waste, improvements in fish quality, improved cost-effectiveness and capacity utilisation, development of under-utilised stocks and increased exports – as well as equity considerations – such as employment creation, reduction of social conflict, protection of recreational fisheries, maintenance of low consumer prices and generation of government revenue.

From this lengthy list of potential goals, certain fishery goals have become well established both in theory and in practice; and, in this regard, the goal of economic efficiency – or rent maximisation – has undoubtedly played a central role in economic studies of fisheries. Indeed, efforts to determine 'economic optima' have often been justified on the basis that they provide benchmarks against which one can determine 'efficiency losses' arising from the pursuit of alternative multi-objective policies (Charles, 1988, 277).[2] This line of reasoning is accepted here, with the policy objective of economic efficiency, that is, dynamic rent maximisation, being set as the principal policy objective. However, in addition to the over-arching objective of economic efficiency, two further criteria are adopted as the basis for 'appropriate' policy formulation, namely equity and stability (Fraser and Jones, 1989). Of course, these criteria are always subject to the strict conditions of biological sustainability.

The rationale for adding the criteria of equity to the over-arching goal of economic efficiency is obvious given the gross inequalities evident through all sectors and all stages (harvesting, processing, marketing and so on) in the South African fishing industry,[3] and given the objectives of the South African government *vis-à-vis* the promotion of equity in the industry, as set out in the Reconstruction and Development Programme (African National Congress, 1994, 40):

> Marine resources must be managed and controlled for the benefit of all South Africans, especially those communities whose livelihood depends on

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[2] In contrast to this, Hannesson (in Charles, 1988, 277) argued that, relative to the single-objective ideal of rent maximisation, a 'best' world is simply not obtainable. Accordingly, Hannesson has suggested that we should not spend too much time deploring this fact, but seek improvements or second-best solutions, given the institutional constraints (such as job-creation).

[3] Examples of the degree of inequality in the industry are given in Chapters 2 and 4.

> resources from the sea .... The democratic government must assist people to have access to these resources.

and (African National Congress, 1994, 104):

> The [fishing] industry is concentrated in the hands of a few major companies which own not only the harvesting rights, but also the processing and marketing concerns .... The primary objective of fisheries policy is the upliftment of impoverished coastal communities through improved access to marine resources and the sustainable management of those resources through appropriate strategies.

Thus, equity considerations necessarily constitute a central consideration in the management of South Africa's commercial marine fisheries.

The adoption of stability as a third goal of fisheries management is due to the substantial immobility of resources involved in the fishing industry. Evidence from other countries reveals that both capital and labour employed in the fishing industry are highly immobile (Crutchfield and Zellner, 1963; Norton and Miller, 1966; Bishop, 1973 & 1975), and there is no reason to expect that this is not the case in the South African hake fishery (Penzhorn, 1992). Indeed, assets invested in the fishing industry are generally regarded as 'sunk' – that is, highly immobile – as capital (especially boats) is typically custom-made. This makes it difficult to shift capital out of the industry should the fishery be scaled-down as part of a regulatory process. As Sumaila (1995, 263) explained:

> ... players undertake investment in capital that is irreversible ... capital embodied in fishing vessels is often non-malleable ... this implies that once a fishing firm or authority invests in a fleet of vessels it either has to keep it until the fleet is depreciated, or else the vessel can only be disposed of at considerable economic loss.

In similar vein, Hersoug (1992, 144) has argued that, should the fishery suddenly contract, it is difficult to sell boats and gear at an acceptable price (although this is probably true of any industry in difficulty). Similarly, any expansion of the fishery might encourage capital augmentation. Again, the situation is difficult to reverse should the industry be forced to contract. Furthermore, Newman *et al* (1979) have suggested that these arguments do not apply to the harvesting sector only, but also extend to capital invested in processing; and evidence of capital immobility has been provided in a number of places (Clark *et al*, 1979; Clark and Kirkwood, 1979; Dudley and Waugh, 1980; Charles, 1983; Charles and Munro, 1985).

With regard to labour in the fishing industry, there is a host of evidence supporting the argument that labour is highly immobile or 'sticky' (Bell, 1972; Platteau, 1984; Terkla *et* al, 1988). Thus, Hersoug (1992, 145) noted that: '... fishing is a "lifestyle" occupation, where the occupants may have problems with changing the rhythm of life ... they [fishers] are trapped in the fisheries'; while Newman *et al* (1979, 31) commented: 'Reducing ... capacity is complicated by the lack of alternative employment for ... fishermen.' Smith and Wick (1983, 32) articulated the point more dramatically:

> Fishing is a complete experience. It stimulates the mind, body and spirit. The feelings and attitudes that make people fish are not just the subject of poets, painters and novelists. Social scientists, too, have shown how love of an activity outweighs economic gain.

This immobility of labour and capital create a number of problems when it comes to policy formulation. To point out two of the more obvious, and more serious, problems:

(i) any policy which allows for short-term expansion or contraction of the fishery is likely to encounter considerable difficulty as the industry readily expands but is extremely reluctant to contract; and

(ii) while theory suggests that, under perfectly flexible input markets, the optimal approach path to the dynamic rent maximising outcome involves a 'bang-bang' approach (no matter how politically difficult),[4] this is no longer the case in the instance of quasi- or non-malleable capital and 'sticky' labour (Clark *et* al, 1979; Clark and Munro, 1982). Indeed, except in the instance of severely over-exploited fisheries, the more efficient harvesting strategy involves 'phased-reductions', that is, gradually scaling-down the fishery to asymptotically approach the dynamic rent maximising outcome (Clark *et* al, 1979, 35).

[4] As discussed in Chapter 3, such an approach requires the complete shut-down of the fishery in the instance that the extant biomass is less than the optimal biomass and an immediate drawing down of the biomass in the instance that the extant biomass is greater than the optimal biomass (Clark *et al*, 1979). The proof of this result is detailed in Appendix 1.

In short, along with the goals of economic efficiency and equity, considerations of stability are argued to be of central importance in the management of South Africa's hake fisheries and, indeed, commercial fisheries more generally. Based on these arguments, the discussion turns to an examination of policy alternatives as the basis for achieving economic efficiency (that is, dynamic rent maximisation) subject to the additional criteria of equity and stability.

Before proceeding to examine policy options in detail, it is worth noting a number of points. First, it is reasonable to expect that the goals adopted as the basis for policy formulation may conflict. Clark (1985, 145), for instance, has argued that the criteria of efficiency and equity are likely to come into conflict; and an example of such a conflict in the hake fishery has been provided by Loayza (1992, 66):

> ... low demand coupled with large economies of scale means that the most efficient industry will consist of a relatively few large firms. The exploitation of hake is a good example. [Because] hake is caught at considerable depths (100 fathoms or more), which requires heavy equipment, the industry requires large-scale operations with onshore freezing and large processing facilities.

In short, efficiency in capital-intensive fisheries may require concentration, which directly violates the principle of equity.[5] Indeed, whether fishery management should properly be concerned primarily with efficiency or with equity is a topic that has been vigorously debated (Scott, 1977; Bishop *et al*, 1981; Clark, 1985); and in practice, there is no question that distributional considerations tend to dominate management decisions. As Clark (1985) showed, the result, unfortunately, is that the overall economic performance of the fishery often receives little attention and consequently is poor by any measure of economic efficiency (Clark, 1985).[6] Where goals come into conflict, trade-offs must be made; and where trade-offs exist, the guiding principle adopted is one of 'reasonableness'. For example, abandoning

[5] Proposals are offered later in this chapter which suggest that the high-barriers to entry due to capital-intensity need not violate equity goals to the extent implied by Loayza (1992).
[6] This has proven to be the case equally in developed and developing countries, and may even be most extreme in countries that depend critically on fish protein supply (Khoo, 1980). Furthermore, it should be noted that Hersoug (1992, 141-2) provided a number of furthe examples of conflicting goals in fisheries development policies in less developed countries.

equity in favour of efficiency which calls for a monopolised industrial structure is rejected as 'unreasonable'. To couch the principle more loosely, modest efficiency losses are accepted as a reasonable sacrifice for 'less modest' equity or stability gains. Of course, this is an issue of judgement.

7.3.2 The Regulatory Framework

Turning to consider the form fisheries management policy should take, one of the first issues to explore is whether the fishery should be regulated at all. In this regard, it was argued in Chapter 3 that competing fishers in an open-access fishery would act myopically in determining their fishing policy, ignoring effects of their own catches on the future fish stock, with the result being rent dissipation (and, in extreme cases, extinction of fish stocks).[7] This is what is referred to as the 'Class I problem of open-access'. As argued elsewhere in this study, there is sufficient evidence to show that over-exploitation of fish stocks under open-access is abundant. The argument has been well summarised by the Access Rights and Resource Implications Task Group (1995, 1):

> Open access fisheries were common before human numbers and technology resulted in over-exploitation of many stocks and heavy competition for the remaining resources. Even today, open access fisheries still occur, but they are almost inevitably marked by over-exploitation, over-capitalisation and gross economic inefficiency. As a result, the open access approach to fisheries management has been totally discredited throughout most of the world. In many nations where such systems remain, there are strong moves and progress towards limited access systems.

However, before accepting regulation as a 'first principle' in fisheries management, two arguments should be considered.

First, 'western' fisheries managers have tended to focus on competition (survival of the fittest) rather than co-operation (such as the potential 'win-win' solution to the well-known 'Prisoners' Dilemma') as the basis for fisheries regulation (Berkes, 1989b, 72). It should be noted, however, that a number of

[7] As was noted in Chapter 3, extinction of fish stocks may not always be economically viable, or even practically feasible (Plourde, 1970 & 1971; Gould, 1972; Clark, 1973a & 1973b; Smith, 1976; Peterson and Fisher, 1977; Clark and Munro, 1978; Hoel, 1978a; Berck, 1979; Cropper *et al*, 1979; Chang and Wang, 1981; Bjørndal *et al*, 1993).

anthropological studies point to successful cases of self-regulation or communal management through, for example, the use of traditional use rights in fishing (TURFs) (Wilson, 1982; Kench, 1984; Berkes, 1989a; Grima and Berkes, 1989; Gibbs and Bromley, 1992; Watson, 1992). Nevertheless, commercialisation invariably places increased pressure on resources, with the effectiveness of communal-based regulatory mechanisms, such as TURFs, being serious compromised.

For example, Bromley and Cernea (1989) noted that common property management often fails because of local institutional failure (due to the existence of, *inter alia*, bribery and corruption). As a result, common property regimes degenerate into open-access which, in turn, precipitates over-exploitation (that is, the Class I open-access problem resurfaces). On this note, the documented evidence of the successful use of communal regulation in industrial fisheries is extremely limited; and the bulk of the available evidence suggests that the success of self-regulation is confined to small-scale or artisanal fisheries (Owers, 1975; Wilson and Olson, 1975; Bromley and Cernea, 1989; Lipton and Strand, 1989; White, 1989; Freeman, 1992; Regier *et al*, 1992; Ruddle, 1992; Miller, 1992). Indeed, given the considerable differences that exist between small-scale or artisanal fisheries and large-scale or 'single-species' fisheries, it is not surprising that regulatory solutions differ considerably.[8]

It must be emphasised, however, that it is not common property regimes *per se* that fail to prevent over-exploitation. Rather, it is infrastructural failure – often due to competitive pressures associated with commercial activity – which results in common-property regimes degenerating into open-access systems. For these reasons, it must be acknowledged that, under the appropriate set of circumstances, communal or self-regulation may constitute

[8] Christy (1986) has provided an excellent review of these differences which include, as noted, single-species versus multi-species targets; differences in vessel capabilities; the level of dependence on home or communal markets; the degree of value-adding that occurs; and the extent to which the fisheries provide alternative employment opportunities for both people and gear.

a viable and effective management system (Bishop, 1973 & 1975; Bromley and Cernea, 1989). As noted by Bromley in (Grima and Berkes, 1989, 37):

> The mischief to arise from the term 'common property' is that many ... do not understand the critical distinction between 'open-access resources' (*res nullius*) and 'common property' (*res communes*). Open-access is free-for-all, while common property represents a well defined set of institutional arrangements concerning who may make use of a resource, who may not make use of a resource, and the rules governing how the accepted users shall conduct themselves.

None the less, in the absence of an appropriate institutional framework, and especially in the face of competitive pressures, it is reasonable to expect commercial fisheries to be driven into over-exploited (or, in rent maximising terms, sub-optimal) states. To return to Hardin (1968, 1244): 'Freedom in the commons brings ruin to all.' Given, then, that regulation of commercial fisheries would appear to be a prerequisite for the prevention of rent dissipation, the discussion below turns to examine alternative regulatory tools.

7.3.2.1 Alternative Regulatory Tools

Accepting regulation of the fishery as a first principle in fisheries management, the question then becomes: 'What form should regulation take?' – that is, what regulatory instruments provide the best tools for achieving the stated goals of economic efficiency, stability and equity. The variety of regulations applied to fisheries is wide-ranging, although regulatory instruments can be divided, at the broadest level, into two distinct groups: non-market or direct controls; and market-based or indirect controls. Historically, fisheries have been regulated by way of non-market controls. However, mainly due to what has become known as the 'Class II open-access problem', success with direct controls has been extremely limited, resulting in the emphasis of regulation shifting away from direct controls, towards the use of market-based instruments with quasi-property rights (Hersoug, 1992, 139).

(a) Direct Controls: Class II and Other Problems

Non-market or direct controls take one of two forms: (i) non-exclusive access, whereby access to the fishery is essentially open to all fishers, but restrictions are placed, for example, on the type of gear fishers may use or when they may fish; or (ii) exclusive access, whereby entry to the fishery is restricted, that is, the fishery becomes a limited or restricted access fishery.

Historically, fisheries have been regulated by way of non-exclusive regulations, that is, the fishery remains 'open-access', while efforts are made to restrict inputs (such as the level of effort directed at the fish stock), or outputs (such as the total allowable catch, that is, total fishing mortality) by way of a host of 'direct' interventions.[9] The main examples of non-exclusive regulations include:

(i) vessel and gear restrictions, which include the imposition of limits on the physical characteristics of vessels (dimensions, tonnage, horsepower, ancillary equipment) or of fishing gear (type, size and number of nets, traps and hooks and lines);
(ii) time and place restrictions, which include restrictions in the form of fishing seasons, times and protected areas;
(iii) catch restrictions, which incorporate limitations on species, size, sex of fish caught and retained; and restrictions on by-catches and discards;[10]
(iv) total allowable catches (TACs), which involve the setting of total catch quotas by species and area and allocation to vessels, plants or other groups; the fishery is closed when the quota has been achieved; and
(v) quality controls, which include, amongst other things, regulations governing the use of ice, freezers and preservatives; the handling and gutting of fish and so on.

The list is not exhaustive: further examples include trip limits, citizenship of fishers and vessel ownership limitations (Gulland, 1977; Clark, 1985; John, 1994).

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[9] See Cushing (1977) for an excellent review of the history of early regulation.
[10] By-catch is the incidental take of species that has value to some other group (Boyce, 1996, 314).

As Clark (1985, 155) argued, there is no doubt that these methods can be effective in achieving conservation of fish stocks and in increasing the size, quality and value of catches. However, while there are examples of the successful application of non-exclusive regulations,[11] these cases are heavily outweighed by the failure of non-exclusive controls to prevent rent dissipation (Johnson and Libecap, 1982; Bennett and Attwood, 1993; John, 1994). The failure of non-exclusive regulations can typically be ascribed to two main problems. First, non-exclusive controls are not underpinned by economic efficiency considerations. Second, non-exclusive controls often result in what is widely known as the 'Class II common property problem' – or, more correctly, the Class II problem of open-access – which refers to the situation where rents are dissipated due to 'crowding' effects, that is, over-investment by fishers in vessels and gear (Munro and Scott, 1985; John 1994). These points are elaborated upon below.[12]

Economic efficiency requires that the optimal harvest be taken at minimum cost (Muraoka, 1990, 149). However, very often, non-exclusive regulations are economically inefficient, as they prevent economies of scale from being fully exploited (Bjørndal and Gordon, 1993, 106). More generally, time restrictions, such as the closure of fisheries, impose substantial costs on fishers, as vessels and crew sit idle or find temporary employment in lesser opportunities. There are also set-up costs – locating to the fishing area, resetting nets or pots and so forth – that must be incurred more than once in fisheries subject to multiple season closures and openings (Karpoff, 1987, 183; Muraoka, 1990). Similarly, capital restrictions, such as vessel length, vessel tonnage, gear type, net length and the number of nets, pots or traps

[11] Studies in South Africa have indicated that marine reserves play a valuable role in preserving biomass, size structure and sex ratios within reserves, as well as in exporting adult fish which are available to the fisheries and which contribute significantly to the overall spawner biomass (Access Rights and Resource Implications Task Group, 1995). See Buxton and Smale (1989), Bennett and Attwood (1991), Buxton (1993a & 1993b) and Attwood and Bennett (1994) for examples of the successful use of marine reserves in fisheries management in South Africa, as well as Tunesi and Diviacco (1993) and Holland (1995) for examples of the successful use of marine reserves in fisheries management in other countries.

[12] As argued in Chapter 3, the term 'common property' is misused, and a more correct term would be the 'Class II open-access problem'.

employable, cannot be defended on efficiency grounds, as these regulations operate by reducing aggregate effort through deliberately imposing inefficiencies on participating fishers (Karpoff, 1987, 183).

Nevertheless, some non-exclusive restrictions can be defended on efficiency grounds. For example, restrictions on the minimum net mesh size help prevent the premature capture of juvenile fish and decrease the incidental catch (by-catch) of secondary species. However, these cases are clear exceptions to the rule: direct regulations are generally not defensible on grounds of economic efficiency (Karpoff, 1987, 183). In short, directly or indirectly, potential rent gains through limited access are dissipated through increased cost. Notwithstanding this, perhaps of greater concern in the issue of the dissipation of rents under non-exclusive access is what is known as the Class II problem of open-access.

The Class II problem of open-access refers to the situation where rents are dissipated due to 'crowding' effects (DuPont, 1991, 156). Here, once appropriate non-exclusive regulations have been put in place, catch levels and quality begin to recover, with commensurate gains in fishers' incomes. Eager to increase their share of the total catch, fishers willingly invest some portion of their higher incomes in vessel or gear improvements (Keen, 1973; Anderson, 1976; McConnell and Norton, 1978; Pearse and Wilen, 1979; Scott, 1979; Squires, 1987). In the absence of regulations precluding such improvement, the obvious result is that all fishers make improvements in order to increase their portion of the catch. At the same time, entry to the fishery will tend to expand as new fishers are attracted by high returns. Unless this is also brought under control, the fishery will converge to a new 'regulated bionomic equilibrium' in which net economic rents are again zero.[13] The nature of the problem has been well described by Munro and Scott (1985, 624):

[13] Indeed, because of the increasing difficulty and expense of enforcing all the regulations, which will probably appear more and more annoying to fishers, the net economic yield of the fishery may well be lower than if no regulations existed. This is much in line with the argument of Smith (1968, 1969, 1971 & 1972).

> If the authorities ... should intervene in the fishery to conserve the resource by imposing [non-exclusive regulations, such as] seasonal or yearly limits on the total harvest, but do nothing to restrict the number of fishermen and vessels competing for the limited harvest, then excess capacity is almost certain to emerge in the fishery ... [this is designated] as the Class II common property [sic] problem. The Class II common property problem results in dissipation of rent because there will be an excessive number of vessels and fishermen competing for the limited harvest. Vessels lie idle or underutilized as too many fishers chase too few fish. Fleet redundancy can lead to rent dissipation through 'crowding'. Also, vessels may impede one another's movement or disrupt one another's gear, thereby leading to economic waste. Part of the rent, we may note, may be dissipated through the processing sector, as the problem of non-salvageable costs arise there too.

In short, the non-exclusive regulatory regime breaks down into a problem of too many fishers, and too much gear, chasing too few fish. Thus, it is argued, non-exclusive regulations fail to prevent the dissipation of economic rents; and there is sufficient evidence to suggest that this is what transpires in practice (Cunningham, 1983; Munro and Scott, 1985; DuPont, 1991).

Accordingly, the problem becomes one of establishing: 'What should managers do when too many fishers are chasing too few fish?'. Historically, the response has been for managers to shift away from non-exclusive access to exclusive or restricted access whilst remaining within the regime of direct controls, whereby some institution establishes administrative pre-conditions that determine who may or may not fish. The vast majority of restricted access programmes involve the adoption of licence limitations on vessels or fishers (Ciriacy-Wantrup and Bishop, 1975; Cunningham, 1983; Townsend, 1990). However, restricted access programmes have generally failed to prevent rent dissipation.

Restricting access to a finite number of fishers, vessels or gear has generally not been successful in reversing rent dissipation experienced under non-exclusive regulations. This is primarily due to what has become known as the 'capital-stuffing' problem (Cunningham, 1983, 74; Clark, 1985, 156). Introduction of a licensing system will result in rent being earned over time from the fishery. This will encourage each fisher to attempt, within the licence constraint, to expand effort (for example, by increasing engine size) provided the expected return exceeds the expected cost. If all fishers have roughly the same expectations, they will be self-defeating in the aggregate as higher than necessary costs will result in partial (if not complete) rent dissipation. As noted by Clark (1985, 154), if the fishery is regulated exclusively by way of limited entry, that is, the exclusion of non-licensed fishers (or 'firms'), the previous result of rent dissipation (the Class II problem) remains unaffected as fishers will over-invest in gear and vessels to secure their catch in a competitive 'scramble' for the total catch (Clark, 1982, 282).

Some writers (Crutchfield, 1979, 746; Campbell, 1990; Campbell and Lindner, 1990) have argued that capital stuffing is 'self-limiting', as inputs can only be substituted at an increasing marginal cost. In similar vein, Anderson (1985a) showed that, under specific circumstances, licence restrictions may indeed not precipitate the capital stuffing problem, and Townsend (1985a) has suggested instances where capital stuffing may improve rents. However, these cases clearly run contrary to the gamut of theory and evidence. Indeed, Cunningham (1983, 74) went so far as to argue: 'no-one disputes that capital-stuffing happens, the debate is whether it happens to a significant degree'; and the response to this would seem to be an emphatic 'yes'. There is sufficient evidence to suggest that considerable over-investment in gear and vessels takes place under licence limitation (Keen, 1973; Meany, 1978; Pearse and Wilen, 1979; Fraser, 1979; Rettig and Ginter, 1980; Clark, 1985, 154; Karpoff, 1987, 180; Rettig, 1989; Townsend, 1985a, 1990 & 1992).[14]

In addition to capital-stuffing, licence limitation has run into a number of further problems. Most notably, the problem of rent dissipation under limited entry programmes has been compounded by problems of fleet redundancy and sub-optimal fleet composition (DuPont, 1990). Moreover, limited entry

[14] The British Columbia salmon fishery provides an excellent case in point. The Canadian authorities introduced a limited entry programme in the British Columbia salmon fishery in the late 1960s which, at the time, was suffering from an acute Class II problem (Anderson, 1977; Munro and Scott, 1985). A strict vessel licensing system was introduced, reducing the number of vessels by 20 percent by 1980. The success, however, was illusory, as it is estimated that by the late 1970s, the amount of capital employed in the fishery may have increased by as much as 50 percent over the previous decade as fishers increased their investments in gear and vessels (just the opposite of what was presumably intended by the authorities) in an attempt to secure their catch by out-competing one another (Munro and Scott, 1985, 660).

may have profound distributional effects (Townsend, 1990, 360-1); the highly unequal allocation of access rights generated under South Africa's limited entry controls provides a suitable example (Chapter 2). In addition, the excess capacity that arises (due to either open-access crowding or capital stuffing) often results in early harvesting, that is, the bulk of harvesting takes place early in the fishing season and little harvesting takes place later in the season. The result is considerable price volatility which compounds management problems (Agnello and Donnelley, 1975; Gowans, 1997b).

Overall, therefore, it seems fair to conclude that limited access is not a *sine qua non* for the prevention of rent dissipation. Indeed, stock and rent depletion due to excess capacity are virtually certain in the instance of limited access restrictions. Invariably, then, limited access restrictions must be coupled with some (if not all) of the earlier non-exclusive regulations in order to prevent harvesting capacity steadily creeping upwards, even though the number of fishers or vessels remains constant or even falls (Anderson, 1976; Clark, 1985, 153; Munro and Scott, 1985, 660; Squires, 1987; Townsend, 1990; Bjørndal and Gordon, 1993). Coupling limited access with non-exclusive regulations, however, precipitates a host of further problems.

First, as already noted, simply restricting access to a finite number of fishers does not necessarily constitute a solution to the problem of rent dissipation. Rather, in order to prevent rent dissipation, limited access needs to be coupled with non-exclusive regulations, such as limited catch, closed seasons, gear restrictions and so on. Indeed, this has been the case in the South African hake fishery (see Chapter 4). However, almost without exception, non-exclusive regulations fail to incorporate efficiency or social welfare considerations (Anderson, 1982). Indeed, some economists have argued that, provided the regulations are sufficiently comprehensive and enforceable and the number of licences sufficiently limited, the incentives leading to economic inefficiency can be defeated by way of direct controls (Turvey, 1964; Tisdell, 1972, 244-5; Clark, 1985, 156). However, this view is rejected here. Non-exclusive controls are deliberately designed to introduce inefficiency into the fishery, as they force fishers to use inputs that do not minimise the opportunity cost of harvesting. In this regard, Morey (1980, 846-7) argued: 'A regulation that institutionalises waste is inconsistent with the socially optimal use of a fish stock. The wasted resources could have been used to produce other items that society valued. Wasting resources results in fewer goods than would have been possible if resources had been used more efficiently.'

Second, regardless of the efficiency (or lack thereof) of direct controls, there is a considerable body of evidence to suggest that direct controls are unable to prevent biological over-exploitation of the fish stock as fishers find loopholes in the regulations (Wilson, 1982). Wilen (1989, 258) has usefully summarised the argument as follows:

> If a central lesson emerges out of the past decade or two of limited entry experiment, it is that such programs have failed to tackle the basic incentives problem, more or less as economists predicted would happen. Conventional limited entry programs, while they may generate rent indirectly, do nothing to encourage efficiency and cost saving. Moreover, such programs do not, in themselves, eliminate excess inputs. Instead, once initiated, such programs take on a (somewhat unpredictable) life of their own – a pattern of action and reaction by fishermen and regulators.

As Scott and Neher (in Munro and Scott, 1985, 661) noted: 'fishermen have generally been more adept at evading the original intent of [direct] restrictions than the regulators have been at devising and imposing them'. An example of the ability of fishers to circumvent direct controls is provided by the case of the Bristol Bay salmon gillnet fishery. In that fishery, direct regulations were introduced to limit vessels to 32 feet in length. However, as noted by Wilen (1989, 256), 'since the introduction of regulations, vessels have become deeper and wider, virtually round in fact, in order to increase hold capacity'.[15] Hence, the problems of capital-stuffing, rent dissipation and over-exploitation of fish stocks re-emerge.

In short, the success of fisheries managers with the use of direct controls as the basis for fisheries regulation is extremely limited, as direct controls run

[15] Similar examples of the ability of fishers to circumvent direct controls are provided in Munro and Scott (1985, 660-1) and Cunningham (1983, 74).

into a plethora of problems which not only undermine the efficiency of regulatory regimes, but also fail to prevent the over-exploitation of fish stocks (Townsend, 1990 & 1992; Emerson, 1997). In other words, direct controls fall down not only on economic grounds, but also on biological grounds. The reason for the failure of direct controls, it is argued, lies in the fact that direct controls treat the symptom – excessive effort – rather than the cause - market failure – of the open-access Class I and Class II problems. In this vein, Munro (1982, 419) argued:

> If quantitative [direct] restrictions on fishing effort prove to be less than satisfactory, presumably one should seek out other means that do what quantitative restrictions fail to do, namely, alter the misleading market signals.

In line with this thinking, the emphasis amongst fisheries managers has, more recently, tended toward the use of regulatory instruments which correct market failure by altering market signals.

These so-called market-based or indirect controls take one of two main forms. The first involves the imposition of what amounts to royalties – taxes on inputs (fishing effort) or, alternatively, on outputs (landings). There is a rich body of theory which demonstrates that, by adjusting market signals (costs and prices), such taxes will lead to an optimal level of exploitation of the resource and will prevent the emergence of redundant labour and capital in the fishery (Smith, 1968, 1969 & 1972; Morey, 1980, 849; Cunningham, 1983, 70). The second method involves establishing a system of individual harvest quotas. The theory and feasibility of these tools are examined in detail below.

(b) Solving for Class I and Class II Problems: The Adoption of Market-Based Regulatory Instruments

Advocates of indirect or market-based controls argue that the problems of open-access are due solely to the absence of well-defined property rights. Accordingly, the problems of over-fishing, crowding and capital stuffing are solved through either the creation of private property rights, or the 'bureaucratic simulation' of the outcome that might be expected from a private property rights system (Wilson, 1982, 418). In line with this reasoning, two forms of market based controls have been emphasised in the literature: taxes and individual (fish or company) quotas (Munro and Scott, 1985, 661).

(i) Taxation as the Optimal Regulatory Instrument

The use of taxation to regulate fisheries has been frequently advocated (Smith, 1968, 1969 & 1972; Morey, 1980, 849; Cunningham, 1983, 70). For instance, as early as the late 1960s, Smith (1968, 429) argued: 'the problem of regulating the competitive fishery can be stated as one of imposing unperceived [external] social costs on the industry'. Employing a static surplus production model, Smith (1968, 429) went on to show that an optimal tax would take the form of a tax on catch plus an annual licence fee on each fishing vessel.[16] Thus, the rationale for a tax or royalty levied on inputs or output is argued to be straightforward: if rent from the fishery can be absorbed by a tax, the tendency toward over-expansion of the fleet and over-exploitation of the fish stock will be curbed (Munro and Scott, 1985, 661). Indeed, in principle, tax rates can be sufficiently fine tuned to ensure that the fishery arrives at the dynamic rent maximising outcome (Turvey, 1964; Strand and Norton, 1980; Rosenman, 1986).

Taxes may be levied either on inputs or outputs; and in theory the end result would be the same: by forcing fishers to incur the full costs of their actions – the external costs of over-fishing and/or over-crowding – the fishery can be driven to the rent maximising optimum. A tax on catch results simply in decreased perceived revenue and a tax on effort in an increase in cost. In both cases, the result is to reduce the open-access level of exploitation from its original position to whatever is considered desirable (Cunningham, 1983, 70). A tax on outputs (per unit of fish) is thus theoretically equivalent to a tax on inputs (such as per unit of effort). This equivalence, however, extends

[16] The validity of this result has been questioned by Fullenbaum *et al* (1971) and defended by Smith (1971).

only to the effort reduction ultimately achieved. In at least two important ways, the input tax and output tax are dissimilar.

Perhaps the most important difference between the input and output tax is in their application. As Hannesson (1987, 91) argued, a tax on effort would require separate optimal taxes on all factors of production constituting fishing effort so as not to distort the choice of cost minimising technology. However, as Clark (1976, 117) pointed out, 'in practice ... taxing the catch appears to be simpler and more effective than taxing effort, particularly [because] the catch can be easily measured whereas effort is notoriously difficult to quantify'.[17] The second major difference between the effectiveness of input and output taxes arises in the presence of uncertainty, where a tax on catch is shown to result in less uncertainty for fishers than a tax on effort (Wilson and Anderson, 1977). This is a result which might well have been expected: in the final analysis, fishers pay their taxes with revenue from catches. A tax related directly to that catch is therefore likely to lead to less uncertainty for the fishers than a tax on effort which depends on an estimate of the expected catch from this effort. From this, Wilson and Anderson (1977, 206) concluded that, by shifting the burden of uncertainty from the fisher to the taxing authority, a tax on actual catch should result in higher receipts than a tax on effort for the same level of catch. Implicit in this argument, however, is the assumption that fishers are risk minimisers. Yet, there is some evidence to suggest that fishers may have a preference for risk (Acheson, 1989; Cicin-Sain *et al*, 1978, 37-38). In such a case, a tax on effort, by increasing risk, might lead to higher revenues. This argument, however, is not well established, and is not likely to hold in the case of a commercial fishery, such as the South African hake fishery, where large-scale operators with high levels of investment are more likely to adopt conservative harvesting strategies. Accordingly, it would seem that, in the case of a fishery

[17] See, for example, Morey (1980) or Squires (1987b) for a review of the problems involved in measuring effort.

characterised by risk averse behaviour, the more appropriate tax is a tax on catch.[18]

While the use of taxes, such as a tax on catch, may seem feasible in regulating the fishery, employing taxation as a regulatory tool proves particularly difficult in practice. Annual (or even seasonal) fluctuations in fish stocks necessitate changing tax rates in response to variations in stocks; and the need to change tax rates may be compounded by, for example, changing growth rates of fish stocks and changing costs or prices across time. All of these factors are identified (in Chapter 6) as being present in the South African hake fishery. And the need to frequently change tax rates makes the administration of taxes extremely difficult. If the tax is set too high, the 'full' optimal harvest will not be taken; if it is set too low, the problem of excess capacity will re-emerge. Moreover, if the harvests of individual vessels consists of a variable mix of species – as is the case in the hake fishery – the difficulties of arriving at the 'correct' tax are compounded (Scott and Munro, 1985, 663; DuPont, 1991, 163).[19]

In addition, it has been argued by some writers (Baumol and Oates, 1971, 51) that the imposition of Pigouvian taxes requires information which regulators rarely possess. This undermines the adoption of taxation as a regulatory regime (Tisdell, 1972, 244). Nevertheless, even if taxes are changed to the 'correct' level, it may take a long time for fishers to respond, given the 'stickiness' of labour and capital (Cunningham, 1983, 71). Indeed, reliance upon taxation alone to restrict effort has the disadvantage that the tax rates required may need to be extremely high, given the low factor mobility in

[18] As an aside, it should be noted that the equivalency of input and output taxes has also been challenged by some writers. For example, Brown (1974) refined Smith's (1968) result by employing a dynamic model to show that the optimal regulatory policy would entail imposing a variable tax on the use of variable input factors (such as vessels and gear) which reflect the marginal social cost of the resource; this result is materially different from that produced by Smith (1968) and is supported by more recent arguments which have extended the analysis to include, for example, considerations of rational expectations (Rosenman, 1986) and uncertainty (Koenig, 1984).

[19] For example, the by-catch (by live-mass) in the offshore trawl fishery for 1991 consisted of, *inter alia*, horse mackerel (25 260 tons), snoek (14 486 tons), ribbon fish (10 598 tons) and monkfish (5 816 tons) (Bergh and Barkai, 1993, 98).

fishing. Furthermore, taxation has proved to be far from popular with fishers. For example, in a study of the Maine lobster fishery, Acheson (1975) found that all fishers were opposed to any kind of taxation scheme. Indeed, the opposition to taxation is perceived to be so great that Wilson and Anderson (1977, 178) concluded that: 'given the fact that no regulatory measures have much hope of being implemented without their support, it is unlikely that taxes will ever become a popular [widely used] regulatory technique'.

In similar vein, Karpoff (1989, 386) has suggested that: 'traditional regulations such as capital constraints and seasonal closures are popular because they redistribute the catch toward the politically dominant fishing groups'. Political difficulties aside, it has been shown elsewhere that, where the state of fish stock is known without error, a combination of specific and *ad valorem* taxes is capable of outperforming any quota in terms of economic efficiency (Koenig, 1984). However, where there is uncertainty as to the state of fish stocks – which is the more likely scenario, especially in, for example, the deep-sea hake fishery – quotas are found to outperform taxation as the regulatory instrument (Koenig, 1984, 125). In addition, the cost of collecting taxes undermines their efficiency *vis-à-vis* other forms or regulation (Tisdell, 1972, 244). These arguments aside, perhaps the most convincing argument against the use of taxation is the fact that there are no obvious examples of the successful use of taxation as a regulatory instrument in commercial fisheries (Brown, 1974; Peterson and Fisher, 1977; Clark, 1982).

In short, while taxation has frequently been argued, in principle, to be capable of solving the Class I and Class II problems of open-access by correcting 'wrong' market signals, the use of taxation runs into a number of problems in practice. As noted, variable biological and economic parameters make establishing optimal tax rates problematic, and the problem is compounded in the case of multi-species fisheries and the presence of information shortages. All of these conditions are likely to pertain in commercial fisheries. Moreover, the use of taxation to 'force' fishers and capital out of the industry is not only politically unpopular, but may also require 'excessively high' tax rates, given that labour and capital involved in the fishing industry are typically 'sticky'. The use of taxation as a management tool is also costly relative to other forms of regulation. Most damning of the feasibility of taxation as a regulatory instrument is the fact that there are no examples of the successful use of taxation in fisheries regulation. Given this result, the discussion turns to explore the feasibility of the second-named market-based regulatory instrument, namely individual transferable quotas (ITQs).

(ii) ITQs as the Optimal Regulatory Instrument

As already noted, the problem of open-access essentially boils down to one of market failure. Whereas Pigouvian-type taxes are an attempt to alter market signals, Cheung (1970, 50 & 53) has taken the argument one step further:

> In the absence of exclusive rights to use of the fishing ground, the right to contract so as to stipulate its use does not exist. This implies the absence of contractual stipulations governing resource use which would exist if the fishing ground were private property, thereby altering the constraint of competition and affecting resource allocation in a number of ways. The alleged 'externalities' in fisheries are thus attributable to the absence of the right to contract .... Should the fishing ground be exclusively owned [owners would act so] as to maximise wealth.

Thus, the problem of social costs, or externalities, dilutes to a problem of the inability of parties to contract (Coase, 1960; Cheung, 1970). Accordingly, the solution is argued to lie not in attempting to correct market signals by way of, for example, Pigouvian taxes, but rather by providing the structure for parties to contract. This essentially entails the creation of private property rights where, in theory, private property rights result in the social costs of production – that is, externalities – being borne by the private producers. In turn, producers are provided with the incentive to manage the resource optimally (Gordon, 1954; Scott, 1955; Smith, 1968; Agnello and Donnelley, 1976).

In the case of fisheries, private property rights for fishers are established through saleable harvest quotas, commonly known as individual transferable quotas or ITQs (Munro and Scott, 1985).[20] By definition, an ITQ is an allocation to an individual (or vessel) of a transferable (that is, tradable) right to harvest a specific amount of the total allocated surplus production of stock (Davies, 1992, 1070). The ITQ is, therefore, an usufruct, a guaranteed right to harvest fish, with an intrinsic value, and so is saleable. The purported advantages of management by ITQs lie in the elimination of important external diseconomies through fishers internalising the external or social costs of harvesting associated with both open-access and limited entry licensing (Copes, 1986, 280). That is, ITQs are argued to be capable of solving both the Class I and Class II problems of open-access.

The guarantee of an individual quota, it is contended, means that fishing operators do not have to race one another to secure their share of the catch as quickly as possible before the total catch quota is filled and the fishery is closed. Furthermore, it is argued, when fishers are assured of their quota, they can spread their effort optimally across an entire season and will have the incentive to use the most economically efficient configuration of labour and capital. As Copes (1986, 280) noted: 'Gone will be the need for competitive escalation of speed and fishing power, requiring large capital inputs and driving costs up unnecessarily.' Other spin-offs are also likely. For instance, operators will find little need to fish in bad weather or under dangerous circumstances in order to keep up their share of the catch. In addition, price volatility may decline as harvest gluts are avoided (or, at least, reduced) (Copes, 1986, 280).

Two crucial aspects of the ITQ system are transferability and divisibility. Transferability means that operators may sell either their entire quota, or parts thereof, to other operators; the evident advantage being that transferability facilitates further gains in economic efficiency (Copes, 1986, 280-1). From a short-term perspective, efficiency would be promoted where an operator was unable to fill a quota (for example, because of illness or vessel or gear

[20] ITQs are variously referred to as stock certificates (Bell and Fullenbaum, 1973), fisherman's quotas (Christy, 1973), quantitative rights (Moloney and Pearse, 1979) and individual fish quotas (Cunningham, 1983).

breakdowns) and was able to (temporarily) transfer part or all of the quota (say, in the form of a lease agreement) to other operators. To give effect to partial transfers, divisibility becomes an important aspect of ITQs. From a longer-term perspective, the evident advantage of transferability is that it facilitates further rationalisation. If there is a surplus capacity of capital and labour in the fishery, rents could be generated in the longer-term by the withdrawal of some fishing units from the fishery. It is reasonable to expect that, in general, it will be the more efficient operators that buy out less efficient operators, as fishers able to capture economies of scale will be able to bid a higher price for quota relative to less efficient fishers, leading to efficiency without the need for costly government regulation (Bjørndal and Gordon, 1993, 106).

In addition, authorities need have no concern about the types of vessel used. Each fisher will have the incentive to use the most efficient harvesting means available, as individual fishers will bid for quota to a maximum value of the difference between the price of fish (that is, the output price) and the cost of production. Thus, quota rights would be consolidated in the hands of the most efficient operators who would be able to fish full-time at the lowest unit cost.[21] In the process, both buyers and sellers of ITQs could share in the benefits of the rents that would be generated (Copes, 1986, 280). Perhaps of greatest importance is Clark's (1980) argument which showed that, if individual quotas are freely transferable and perfectly divisible, the quota system will have the same efficiency terms as taxes, that is the most efficient outcome possible (Munro and Scott, 1985, 663). To would-be entrants to the fishery, the cost of acquiring a quota will be comparable to the cost arising from an efficiently administered tax. To those fishers with quotas, there will also be a cost comparable to a tax in the form of an opportunity cost of holding quotas. Politically, of course, ITQs are far more palatable than taxes (Munro and Scott, 1985, 664).

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[21] Taken to its logical conclusion, in a constant cost industry, quotas will ultimately become concentrated in the hands of the single, most efficient operator. This issue is explored in greater depth below.

In addition to facilitating gains in economic efficiency, ITQs have a number of patent advantages over other regulatory tools. First, the direct control of catches reduces the need for complex restrictions on effort. Second, given that the quota constitutes a transferable asset with an intrinsic or imputable value, it follows that ITQs are capable of providing greater financial security for fishers who can, for example, augment retirement capital or income by selling or leasing their ITQ to existing or new entrants. The sophistication of ITQs as financial instruments is potentially further enhanced by the development of futures and options markets in addition to spot trading. This is a new idea, however, which has not been explored in any great depth. ITQs can also be used to raise government revenue at first-sale, or through the collection of annual fishing fees (essentially a 'modest' tax). These points are returned to in greater detail below.

Perhaps most important, however, is the fact that, unlike other forms of regulation, there are numerous examples – including fisheries in New Zealand (Davies, 1992; Clark, 1994; Hannesson, 1994), Australia (Hannesson, 1994), Iceland (Arnason, 1994), Canada (Copes, 1986), Chile (Zorzano, 1992) and the United States of America (Copes, 1986) – of the successful use of ITQs in practice. Indeed, in view of the theoretical superiority of the ITQ system, operators in the South African hake trawl fishery have argued strongly in favour of the adoption of ITQs. For example, SADSTIA and SECIFA (SADSTIA, 1994, 9) have argued that: 'Fishing rights should be divisible and freely transferable. The sale of quota should not be subject to approval by any political or bureaucratic authority.' In short, it would seem fair to say that, in addition to being theoretically superior to other regulatory tools, ITQs have, in their admittedly relatively short history of use by authorities, demonstrated an ability to solve the Class I and Class II problems of open-access in practice. Moreover, there is increasing acceptance of ITQs amongst authorities and fishers including, as noted, those involved in the South African hake fishery. Based on the above set of arguments, it is suggested that ITQs constitute a feasible and effective alternative to the extant regulatory regime in the South African hake fishery.[22]

While the theoretical advantages of ITQs are widely and generally accepted (Rettig and Ginter, 1978; Pearse, 1979; Anderson, 1991a & 1991b; Neher *et al*, 1989), the implementation of ITQs does present practical problems. As Clark (1985, 173) noted:

> Although ITQs are a relatively new concept in fisheries management, they seem to offer the best practical hope for the economic rationalisation of commercial fisheries. Implementing a successful quota, however, is often a difficult task, as fisheries are complex.

In line with Clark's comments, the aim of the final section of this study is to identify the likely set of problems surrounding the successful implementation of ITQs in the South African hake fishery, and to suggest an acceptable, workable solution to the problem set.

7.4 Implementation of ITQs: Problems in Practice

Broadly speaking, problems with implementation can be grouped into six main areas: (i) first-round distribution; (ii) use of percentage-based or volume-based quotas; (iii) duration of rights under the quota system; (iv) transferability of ITQs and the implications for industrial structure; (v) the optimal approach path and implications of alternative technologies for industrial structure; and (vi) efficient and effective management. Each of these issues is discussed in greater detail below.

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[22] Importantly, this is not to suggest that ITQs – or, more generally, market based instruments – are invariably superior allocative instruments: the market mechanism works better in some circumstances, regulation (meant, in the broadest sense to include communal regulation) works better in others. As noted by Grima and Berkes (1989, 41), in rare cases, even open-access makes sense (for example information that should be freely accessible in the public interest). Of course, it is difficult to think of examples where open-access should be adopted as the basis for the management of fisheries (although, as noted earlier, Smith [1968], amongst others, argued that open-access could constitute a feasible management approach).

7.4.1 First Round Distribution of ITQs

One of the principal problems arising from the adoption of an ITQ system is the question of how to allocate the first issue of ITQs. Copes (1986, 281) noted that two extreme approaches suggest themselves. Either the quotas can be handed out on a 'grandfather' basis or they can be sold by auction (Copes, 1986, 281). Under a 'grandfather' system, rights (quotas) are distributed to the fishers on the basis of their involvement in the fishery over a certain reference period. The typical approach is to supplement this with a minimum landing requirement during this period (Cunningham, 1983, 73).

Distribution by 'grandfathering' has two distinct attractions: the system secures rights for those historically involved and, at the same time, prevents 'non-fishers' competing for rights to 'on-sell' once allocated. The primary disadvantages of 'grandfathering' are two-fold: any historic distortions in access is perpetuated, as would be the case in the South African hake fishery; and, the giving away of rights (especially in perpetuity) produces windfall gains for the recipients at the expense of the rest of society. As Hannesson (1994, 92) noted:

> Those who receive ITQs as a gift would be able to sell them and obtain a windfall gain. But those who want to become fishers after an ITQ system is in place must buy their way in. For them, the value of the ITQs is the cost they can hope to recover when they exit the fishery.

Thus, under the 'grandfathering' system, the gains from the ITQs end up with the first generation of fishers who acquired the ITQs at no cost; and this grossly violates the principle of intertemporal (or intergenerational) equity.[23]

Auctioning the initial allocation of quotas is a more impersonal, and probably less arbitrary, approach than 'grandfathering'.[24] In theory, auctioning ensures that rights go to the most efficient fishers (those willing to pay the highest

[23] It is frequently, and quite correctly, pointed out that this same value also represents an opportunity cost to the recipient as long as the quota is not sold (Cunningham, 1983, 73).
[24] While allocation of rights by way of a lottery is probably the most arbitrary allocation system (all responsibility for determining the allocation of rights is removed from the state), the system has limited (or no) scope for ensuring the efficient use of resources.

price for quotas). However, auctioning rights has its problems. First, so long as markets are efficient, the licensing authority will capture the entire capitalised value (discounted net present value) of the rent which the fishery is expected to yield, and this may not be desirable from an equity perspective in that all (present and future) harvesters would earn zero economic rents. Secondly, quota allocations may be more reflective of the ability of operators to pay than of the efficiency of operators. For example, operators with 'deep capital structures' may be able to out-bid more efficient, but less advantaged fishers;[25] the result being increased concentration (although it is hard to imagine the level of concentration in the South African hake fishery increasing any further from current levels).

In short, there are advantages and disadvantages with either 'grandfathering' or auctioning the first round allocation of ITQs. Here, however, it is proposed that the best of both worlds may be had by capturing the desirable features of the two systems whilst eliminating the worst. Such a system would involve the use of 'grandfathering' to effect the initial allocation and, whilst not employing an auction system, require fishers to pay an initial purchase price to secure these rights (in perpetuity). Such a system, it is argued, would protect the rights of existing fishers and, at the same time, prevent the abuse of the system by 'non-fishers' (witness the abuse of the 'paper' quota system in the South African hake fishery).[26] Moreover, by extracting a price for the quota, the criticism against the 'giving away' of rights violating intertemporal equity considerations would be ameliorated, although not eliminated (indeed, the higher the initial price the lower the intertemporal equity distortion).

[25] Of course, the opposite may hold in industries which enjoy high economies of scale, that is, those with the 'deepest' capital are also the most efficient operators. This argument may hold in the case of the relatively capital-intensive deep-sea fishery for hake.

[26] As noted in Chapters 2 and 4, in an effort to redress past imbalances, so-called paper quotas have been allocated to a number of recipients over the past three years in an effort to encourage new entrants into the hake fishery. However, it has become increasingly apparent that these recipients have often had neither the intention nor the ability to fish. For example, none of the recipients of the 1994, 1995 or 1996 hake 'paper quotas' have attempted to enter the fishery. Rather, they have sold their quotas on to *bona fide* operators. By way of example, all 15 grantees of the 1996 paper quotas 'on-sold' their quotas for about R1 500 per ton, generating an income stream of R8 million (*Cape Business News*, 1997c). The figure for 1997 is expected to be of the order of R18 million. The practice is viewed with increasing scepticism by regulators and operators.

Furthermore, the initial price would act as an incentive to fishers to bid only for quotas which are 'reasonably' profitable; that is, an initial price would provide an incentive to fishers to improve efficiency (or at least eliminate any gross inefficiencies).[27]

In addition to the above, it is proposed that an annual user fee (effectively a tax) should be levied on ITQs. In this regard, given that quota holders enjoy preferential access to a public resource from which they derive economic benefit, it is reasonable to collect an annual charge or rental on, for example, allocated quota (Davies, 1990, 1072; Butterworth in Davies, 1990, 1076).[28] The resultant annual revenue flow would augment the capital pool realised upon first-sale of the ITQs and these, together, could serve as the basis for meeting various socio-economic goals. For instance, these resources could be used to fund research and development in the fishery, to improve management of the fishery and to offset enforcement costs, effectively making the ITQ system self-funding. Moreover, given that the above system fails to address the issue of inequity (which is a particularly serious concern given the high level of concentration in the South African hake fishery), the capital pool and/or revenue flows could be managed by a 'development corporation' to serve as the basis for improving equity by providing start up capital or working capital on preferential terms to new or emerging entrants,[29] as well as allowing for the introduction or enhancement of employee share-ownership schemes. Provision could also be made for spending on employee development, training and retraining, as well as welfare-oriented schemes such as unemployment, sickness or disability and pension schemes.

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[27] It is useful to note that the proposed *Living Marine Resources Bill* (Republic of South Africa, 1997, 26) recommends that the Minister of Environmental Affairs and Tourism be given the discretion to grant quota to applicants who may not be the highest bidder, thus paving the way for the granting of quota on the basis of, for example, economic empowerment criteria. Indeed, the proposal goes on to suggest that a public company be incorporated for the purposes of, *inter alia*, promoting fair and equitable access to fish through actively seeking to promote the activities of small- and medium-sized enterprises. The issue of equity is returned to below.

[28] As noted above, the optimal tax is argued to be a tax on catch in the instance of a commercial fishery with risk averse operators.

[29] This is particularly important in light of the fact that the price on ITQs (either at first sale or upon resale) will serve as a barrier to entry to new and emerging (small-scale) fishers.

Two points should be made regarding the use of such a tax. First, the imposition of an annual tax will potentially reduce the price at which ITQs are first (and subsequently) sold as ITQs would carry a lower imputed value. However, from an intertemporal perspective, it is clearly easier to allocate spending from revenue flows than from capital stores. Second, as noted earlier, taxes are equivalent to ITQs in their efficiency effects. The implication being that the adoption of a 'partial' tax (that is, in conjunction with ITQs rather than as opposed to ITQs) has no effect on efficiency levels. This outcome is explained by the fact that the remainder of 'external costs' would be represented by lower, but still market-determined, quota prices. Moreover, the 'partial' tax leaves regulators free from the problem of setting or adjusting the tax on an annual basis in order to reflect changes in the economic or biological status of the fishery, as these changes will be reflected in the ITQ price.

For the purposes of illustration, an effort is made here to provide a practical example of the mechanics of the proposed scheme. The figures set out in Table 7.1 provide for dynamic rent maximising catches of 125 714 tons per annum in the west coast fishery and 42 709 tons per annum in the south coast fishery. Employing ratios provided by Stuttaford (1995, 29-31) for the conversion of catch figures to landed mass, a total catch of 168 423 tons per annum converts to a landed mass of 105 435 tons per annum, with an annual value of R260 million at first sale (1993 prices). Based on an operating profit margin of 20 percent, this converts to an annual rent from the hake fishery of the order of R52.3 million on landed value. Using a discount rate of between 3 and 6 percent, as proposed in Chapter 6, the capitalised value of the hake fishery is calculated to lie somewhere between R925.5 million and R1 798.7 million (1993 prices). For the sake of simplicity, an average of these net present values is assumed as the 'true' capitalised value of the fishery, that is, R1 313.5 million (equivalent to approximately R7 800 per ton in 1993 prices). It is this capitalised value that would enable managers to arrive at an initial quota price, as well as to set a 'low but effective' tax capable of encouraging efficiencies and providing a significant revenue base whilst, at the same time remaining sufficiently low to avoid serious objection or encourage cheating or other avoidance behaviour.

For the sake of argument, a figure of 20 percent of capitalised value is selected as the issue price and 2 percent of capitalised value as the tax rate.[30] In monetary terms, these figures convert to R1 560 for the right to harvest one ton of hake in perpetuity, with an annual charge of R156 per ton (1993 prices). Based on this scenario, the capital pool generated by the fishery would amount to R262.7 million, augmented by an annual revenue flow of R26.3 million (all in 1993 prices).

Here, it is worthwhile noting that the global operational costs of the management (which include administration, control and research costs) of South Africa's sea fisheries amounted to approximately R60 million in 1996 (Department of Environmental Affairs, 1997, 10). Of this total, approximately 20 percent is consumed by administration, 17 percent by the operation of marine vessels, 29 percent by marine control and 34 percent by the activities of the Sea Fisheries Research Institute (Department of Environmental Affairs and Tourism, 1997, 10). In 1996, approximately R9.5 million (or 16 percent) of total costs was sourced from fisheries-related fees and levies (channelled through the Sea Fishery Fund) (Department of Environmental Affairs and Tourism, 1997, 10). Inflating the 1993 figures by the fishing industry producer price index to 1996 prices produces values of R353.7 million for the capital pool, augmented by annual revenue flows of R35.3 million. To put the matter differently, on their own, annual revenues generated by the hake fishery would be sufficient to cover almost 60 percent of the global operating costs of managing South Africa's sea fisheries.

[30] No particular significance is attached to the values of 20 percent and 2 percent selected for the issue price and tax rate respectively; although these figures are not out of line with rates adopted by authorities in fisheries of a similar nature in other countries (World Bank, 1992; Loayza, 1994a).

Before proceeding, an important point should be noted. The ability of fishers to pass on tax charges to consumers is relatively high, given the concentrated structure of the hake fishery. Thus, while the fishery operates at the 'optimal' outcome, the distribution of rents is distorted with rents accruing to fishers being disproportionately high. If this outcome is to be avoided, it is necessary that regulators promote a more competitive industrial structure. This point is merely noted here and is returned to below where the issue of concentration and competitiveness is examined in closer detail.

7.4.2 PERCENTAGE-BASED OR VOLUME-BASED QUOTAS

A second problem of implementation is whether to allocate quotas on a volume or percentage basis. Under a volume-based system, quotas represent a given tonnage of catch, and remain fixed irrespective of the size of the total allowable catch. Under a percentage-based system, quotas represent a given percentage of the total allowable catch, and thus vary with the total allowable catch. The method of controlling the total allowable catch will differ under the two systems, and is best examined by taking the case of a desired decrease in the total allowable catch. Under a percentage scheme, the lower catch is simply announced and each percentage share is consequently worth less fish; whereas with a system based on quantities, the regulatory authority would be required to enter the market to buy up the excess rights (Cunningham, 1983, 75).

The principal advantages of the percentage-based scheme appear to be that it may be easier to implement and it requires a lower level of funding.[31] However, such a scheme suffers a serious drawback in that decreases in total catch may prove difficult to achieve because they require reducing each fishers' catch by the same amount (Moloney and Pearse, 1979). With a volume-based scheme, total allowable catch is altered by the regulatory authority selling or buying ITQs. This approach seems more feasible, both

[31] Although monitoring costs under a percentage-based scheme may be relatively high. This issue is discussed below.

politically and administratively. Indeed, the fact that a volume-based ITQ system allows for open market operations, while a percentage-based scheme (by definition) cannot, is perhaps the strongest argument in its favour (Cunningham, 1983, 75).[32] The use of market intervention means that fishers (as a group) only reduce effort if the authorities make it worth their while to do so. If control is exerted simply by altering the total allowable catch, then the process becomes compulsory and resistance to it may be expected. This has the potential to raise administrative costs considerably through a need for closer enforcement. Moreover, under the volume-based ITQ system, except in extreme instances (such as a moratorium on fishing), there is certainty as to the catch each fisher can take, as it is given by the ITQs held by the fisher. Under the percentage-based system, an element of uncertainty is introduced as annual fluctuations in the total allowable catch – which are not 'knowable' with perfect foresight – result in fluctuations in the catch for all fishers.

While the net benefits of a volume-based system appear to outweigh those of a percentage-based system, two caveats ought to be noted. First, the successful implementation of a volume-based system requires a greater level of funding than a percentage-based system. However, too much should not be made of this problem. As noted above, the initial round of sales, coupled with the annual levy (as hypothesised earlier, equal to R353.7 million and R35.3 million in 1996 prices respectively) should provide more than sufficient funding. Furthermore, the only time when the amount of funding is likely to be excessive is in the early stages of the management of a severely over-exploited fishery (Cunningham, 1983, 75); and this is not applicable to the South African hake fishery at this stage. Second, if volume-based ITQs are to be effective, then it is vital that a well-traded (that is, efficient) market for quota exists. As evidence of this, the failure of a recent experiment with the use of volume-based ITQs in New Zealand's fisheries is primarily attributed to the inability of the New Zealand government to buy and sell quota on the open market due to a relative absence of willing buyers or sellers of quota

[32] Reductions in catch brought about under a volume-based system by way of 'announcement' may also impinge upon individual rights associated with the purchase of quotas (especially if rights are purchased in perpetuity which, as argued below, is a highly desirable outcome).

(that is, market 'thinness') (Access Rights and Resource Implications Task Group, 1995). To this end, it may be necessary for the regulatory authority to appoint 'market-makers' (that is, principals) responsible for generating market liquidity. To some extent, the problem of 'market thinness' would be further ameliorated through the creation of derivative markets (that is, futures and options markets, as suggested in Section 7.3.2.1).

Before moving on, it should be noted that any quota scheme which allows for multiple processing forms will need to cater for the 'conversion problem'. In this regard, volume-based (or, for that matter, percentage-based) quotas represent a certain total catch. However, in the presence of multiple processing forms, such as at-sea processing and land-based processing, fish are landed both 'unprocessed' (or 'green') as well as processed. Accordingly, there is a need to convert, for example, landed weight to green weight. This may complicate (and increase) the cost of setting quotas, thereby undermining the desirability of quota-based systems. However, the 'conversion problem' is considered to be sufficiently easily resolved so as to leave the integrity and desirability of regulation by ITQs intact (Anderson, 1991b).

7.4.3 Duration of Rights

How long should quota rights be transferred for? Here, two arguments present themselves. First, the longer the period over which rights are allocated, the greater the degree of security enjoyed by fishers and, accordingly, the stronger the likelihood that fishers will protect the resource and strive for efficiency in harvesting. As Davies (1992, 1076) argued: 'Security of tenure stimulates investment in upgrading capital equipment and in developing measures which increase product quality and promote marketing initiatives that maximise economic returns on quotas.' Short-term rights, on the other hand, are likely to undermine the propensity of fishers to invest in (for example) harvesting or processing equipment (Penzhorn, 1992). Short-term quotas also provide an incentive to cheat, that is, over-fish, as operators' time horizons are reduced. As noted by the Access Rights and Resource Implications Task Group (1995, 39):

> Short-term rights carry with them many of the dangers of open-access. If a fisher has no guarantee that he or she will retain the right into the next allocation period, behaviour will become very similar to an open-access system. Fishers will be driven by a desire to maximise benefits, perhaps resorting to fishing illegally beyond their allocation, while they retain access, without regard to long-term sustainability.

Furthermore, short-term rights will result in price distortions in the ITQ market which, in turn, will produce a sub-optimal outcome. Indeed, the granting of rights which are other than perpetual will produce distortions in the ITQ market which will precipitate over-fishing.

Nevertheless, there is a possibility that the granting of a perpetual usufruct may result in some fishers not exploiting rights or abusing these rights. Two solutions to this problem are offered here. First, fears of under-harvesting are reduced by the adoption of a 'low but effective' tax on ITQs, as suggested above. Second, potential abuse of rights can be significantly reduced by counterbalancing the rights of operators with duties. As an example, SADSTIA (1994) has suggested that access to ITQs should hinge upon, *inter alia*, compliance with a Code of Conduct regulating employment practices and social responsibilities; ownership of adequate trawling facility; full utilisation of commercial catch; support for research and development; use of local facilities and labour; provision of secure non-seasonal formal employment; and strict adherence to fishing regulations.[33]

7.4.4 Transferability and Concentration

Transferability is a fundamental requirement for effective operation of the ITQ system. However, transferability brings with it the potential to generate further

[33] It has also been suggested that the abuse of long-term rights could be prevented by the adoption of a system of gradual attrition of rights. Under such a system, used in at least one fishery in Chile, long-term rights are granted to fishers but a fixed percentage reverts to the state each year to be re-auctioned in smaller blocks (Access Rights and Resource Implications Task Group, 1995). While such a system would have obvious advantages (such as the promotion of equity over the longer term), a principal drawback is that even a low level of attrition is likely to encouraging more 'aggressive' (open-access) exploitation of fish stocks as rights are eroded.

concentration in South Africa's already highly concentrated hake fishery. If quotas are freely tradable, then there is every reason to believe that more efficient operators will buy up the quotas of less efficient fishers. Indeed, this is argued to be one of the reasons why transferable quotas precipitate economic efficiency. However, transferability implies that, in the longer run, the ITQ system facilitates industrial concentration rather than competition, and evidence from other countries supports this result. As an example, *The Economist* (1997) noted with respect to the Icelandic cod fishery:

> As far as fish are concerned, quotas have been a roaring success. Stocks of cod, the most valuable fish, are starting to recover. But the quota system has led to concentration, as big companies have bought up the fishing rights of smaller ones. From 1990 to 1994, the number of quota holders dropped by 26%. The result has been to generate fierce opposition to quota trading.

As Grima and Berkes (1989) argued, the general problem with private property and specifically with ITQs as regulatory mechanisms over open-access (or common property resources for that matter) is that access limitation and the allocation of exclusive rights is a zero-sum game in terms of rights-to-use. The creation of exclusive rights for one group necessarily means the exclusion of some other group (witness the disenfranchisement of fishing communities along the South African coastline). As some anthropologists have pointed out, there are not only 'tragedies of the commons', but also 'tragedies of the commoners' when inequities and losses occur with privatisation of resources (McCay in Grima and Berkes, 1989,52).

In addition, transferability may also foster economic inefficiency, as it is not necessarily the most efficient operators, but rather the most powerful operators that secure higher quotas. As argued earlier, a firm's financial resources, rather than its economic efficiency, can play a role in the industry restructuring process. Larger firms, that can more readily afford the costs associated with quota purchase or lease, may not necessarily be the most efficient firms. In contrast, smaller firms, without large financial resources, may be unable to afford quotas or secure the necessary loans. This has apparently taken place, at least to some extent, in the ITQ fisheries for sole and plaice in New Zealand and The Netherlands (Squires *et al*, 1994, 201).

Moreover, as Anderson (1991a) has argued, the economic efficiency effects of creating property rights depends upon the workings of the market for both the rights and the final product, and concentration of ITQs in the fishery creates the potential for market failure in the quota market, the result being rent dissipation. This has two adverse implications. First, under a concentrated industrial structure, the rents generated by the fishery accrue to a handful of operators. Under extant market structures, it can be safely assumed that I&J and Sea Harvest would receive the bulk of rents. Second, the 'inefficiency' effects brought about by the acquisition of ITQs by larger less efficient operators imply that the level of rents earned by the fishery may still be high, but are, none the less, sub-optimal.

Therefore, it becomes necessary not only to promote competitiveness but, at the same time, to ensure that any transfers that take place, do so on the basis of efficiency rather than financial muscle. The proposed Code of Conduct could go some way toward ensuring that transfers take place on the back of efficiency. Furthermore, the capital fund proposed above should provide a basis for enabling new or emerging operators to secure or improve their respective positions in the fishery, thereby facilitating increased competitiveness and, by implication, improved efficiency. Increased competition will also reduce the ability of the industry to pass on costs – such as quota prices and taxation charges – to consumers. It should be noted, however, that there is a danger that dissecting the industry into too many parts may also induce inefficiency by undermining economies of scale or scope. For example, the Association of Small Hake Quota Industries considers a quota of 2 000 tons per annum to be the threshold of viability for new trawling ventures (*Cape Business News*, 1997d). Ideally, then, the industry structure lies somewhere between perfect competition and oligopoly. This is an admittedly broad spectrum, and further work needs to be done towards the end of establishing the most appropriate industry structure.

7.4.5 The Optimal Approach Path and Industry Structure

7.4.5.1 The 'Bang-Bang' versus Asymptotic Approach Path and the Implications for Industrial Structure

The figures set out in Table 7.1 show that both the west and south coast fisheries are in sub-optimal positions, and that the dynamic bioeconomic biomass is larger than the extant biomass by about 15 percent in the west coast fishery and 25 percent in the south coast fishery. It follows that achieving dynamic efficiency involves reducing effort in both fisheries to allow fish stocks to recover. The traditional fisheries theory, as set out in Chapter 3, shows that the optimal approach path, in these instances, is the so-called 'bang-bang' approach, which involves driving the fishery to the dynamic rent maximising outcome by either forcing fishers out of the fishery or imposing a moratorium on effort. However, as set out earlier in this chapter, this argument only holds if labour is mobile and capital is malleable. Evidence drawn from the literature and from the comments and observations of participants in the South African hake fishery suggest that the exact opposite is true: labour is immobile and capital is 'sunk' or non-malleable. Under these circumstances, the optimal approach to the dynamic rent maximising outcome becomes a 'phased' approach path. However, the literature offers no guide as to what a 'phased' approach involves. Presumably, the approach path should be sufficiently gradual, and sufficiently long, to allow for 'smooth' structural adjustments in the harvesting and associated sectors (processing, maintenance and so on) to occur.

This is an area that requires deeper investigation and, in the absence of reliable data on industry cost structures, comment is limited. For the sake of argument, however, such an approach path might involve allowing for an increase in the west coast hake biomass of, say, 1.3 percent per annum on the back of an annual 1.3 percent increase in catch and 1.4 percent fall in effort. Over a ten-year period, these changes would produce the 12.95 percent increase in biomass and 30.14 percent increase in catch on the back of a 28.92 percent decrease in effort required to shift the west coast hake fishery from its extant (1995) position to the dynamic rent maximising position. Similarly, an annual increase in south coast biomass of the order of 1.4 percent on the back of a 1.2 percent drop in catch and 1.4 percent fall in effort would bring about the necessary changes in biomass, catch and effort to shift the south coast hake fishery from the extant position to the dynamic rent maximising position over a ten-year period (see Table 7.1).

An approach path of this type, it is argued, will not only allow labour and capital sufficient time to adjust, thereby reducing the extent of redundancies (and so inefficiency), but also reduce the likelihood of smaller players being marginalised in the adjustment process. The need for such an approach is recognised by the Department of Environmental Affairs and Tourism in its White Paper, *Marine Fisheries Policy for South Africa*, where it is proposed that changes to management and control structures in marine fisheries be introduced in phases, rather than *en mass*, in an effort to reduce uncertainty and promote stability in the industry (Department of Environmental Affairs and Tourism, 1997, 13 & 17).

Before proceeding, it is useful to provide some comment on the likely impact on the industrial structure of the fishery brought about by shifting the hake fishery from its extant status to the dynamic rent maximising outcome. It must be emphasised that the figures provided are necessarily crude and are only suggestive of what might transpire. It is first necessary to make a number oı restrictive assumptions. Specifically, it is assumed that the capital:labour ratio is fixed, the input:output ratio is constant and no adjustments are made in harvesting or processing technology. The last assumption is particularly restrictive and is revisited below. Turning to the figures shown in Table 7.1, it is noted that shifting the fishery from the extant position to the dynamic rent maximising biomass would require substantial reductions in effort – approximately 29 percent in the west coast fishery and 26 percent in the south coast fishery – which, in turn, translates into a reduction in harvesting capacity. At the same time, the shift to the optimal dynamic outcome is shown to produce changes in total catch – an increase of 30 percent in the west coast fishery, but a fall of approximately 7 percent in the south coast fishery – which implies adjustments in the processing sector.

Based on these figures, it is possible to arrive at some crude predictions of the likely impact of shifting from the extant position to the dynamic bioeconomic optimum on the industrial structure of the hake fishery. To start with, the hake fishery currently employs 9 000 people in a ratio of roughly 1:2 between the harvesting and processing sectors. In line with the estimates of the required reduction in effort, it is reasonable to assume that employment in the harvesting sector would fall by approximately 25 percent. This would translate into a loss of 250 jobs in the harvesting sector. However, the loss of jobs in the harvesting sector would be more than offset by an increase of 1 100 jobs in the processing sector. The increase in employment is due, primarily, to the substantially higher catch in the west coast fishery, which translates into a net gain of 1 250 jobs in that fishery, marginally offset by a loss of 120 jobs resulting from the 7 percent fall in catch in the south coast fishery.

Whilst crude, the figures do allow for two important conclusions. First, shifting the fishery to the dynamic rent maximising outcome will probably have little impact on total employment levels: the figures set out above suggest a loss of 850 jobs in the harvesting sector, but a gain of 1 100 jobs in the processing sector, producing a net gain of 250 jobs in the fishery. This amounts to a modest 3 percent increase in employment numbers in the hake fishery. Second, the employment structure is likely to shift further in favour of the processing sector. It should also be noted that shifting the fishery from the extant position to the dynamic rent maximising position is likely to have similar implications for the capital structure of the fishery, with the reduction in effort implying retirement of capital in the harvesting sector. However, capital (and labour) remaining in the harvest sector will be used more efficiently: as was noted in Chapter 5, catch per unit effort in the west and south coast fisheries are modelled to increase by 40 percent and 55 percent respectively. The higher catch, in turn, implies a need for increased capital investment in the processing sector.

It is worth noting here that the ratio of the dynamic rent maximising catch between the west and south coast hake fishery, as set out in Table 7.1, is of the order of 3:1. This is significantly different to the historic ratio of 2:1 noted by Japp (1996), and is more than one-and-a-half times the ratio of catch between the west and south coast fisheries recorded over the period 1984-95, as shown in Table 7.1. To put the matter differently, the structure of the hake fishery is not only likely to shift within the west and south fisheries, but also between fisheries, with the west coast fishery becoming increasingly dominant as managers shift the fisheries from their extant positions to the dynamic rent maximising positions.[34]

A further point should be made here. As argued above, managing the fishery on the basis of a dynamic maximum sustainable yield strategy is likely to impact on the industrial structure of the fishery by altering both input (capital and labour) and output (catch) levels. It is not inconceivable, indeed it is highly likely, that the changed industrial structure has implications for cost and price levels. In turn, this implies that modelling of the hake fishery requires on-going refinements, and is not a one-off exercise. Such an outcome would not be unique. An example should serve to illustrate the point. The adoption of ITQs in the Icelandic fisheries in the early 1980s saw capital investment in the commercial fisheries between 1985-90 increase by approximately 3C percent over the 1978-85 average. At the same time, the introduction of ITQs is argued to have been responsible for actual fishing effort being almost 60 percent less than the level that could have been expected under the previous (common property) management framework, while catch values increased by

[34] This study makes no effort to establish the breakdown of the total dynamic rent maximising catch between the deep-sea, midwater and inshore trawl fisheries. However, based on historic ratios, it is possible to arrive at 'crude estimates' of the ratio of the dynamic rent maximising catch between these fisheries. Specifically, Japp (1996) showed the ratio between the deep-sea, midwater and inshore fisheries to be 91.6:0.2:6.6. Based on a total dynamic rent maximising catch of 168 423 tons per annum, the ratio converts to annual catch figures of 154 275 tons (the catch in 1995 was 130 438 tons) in the deep-sea fishery; 11 115 tons (9 398 tons) in the inshore fishery; and 337 tons (285 tons) in the midwater trawl fishery.

a factor of five between 1984 and 1990, implying considerable gains in efficiency through reduced costs, as well as improved revenues through higher prices received for the catch (Arnason, 1994). That said, in the absence of reliable data, establishing the likely trend in parameters *a priori* is likely to prove an exceptionally difficult task.

The above analysis also makes the assumption that fishers continue to practice extant harvesting methods, that is, trawling, to take the bulk of the quota. However, in its purest form, the ITQ system places no restriction on the use of alternative technologies (for instance, long-lining as opposed to trawling) subject, of course, to the satisfaction of various biological criteria. Accordingly, before leaving the topic of industrial structure, it is useful to consider, albeit briefly, a further possibility: the potential for changes in harvesting technology, rather than simply considering changes inside of the extant harvesting technology.

7.4.5.2 Alternative Technologies and the Implications for Industrial Structure

The use of long-lines to harvest hake is still in its infancy in South Africa, with less than 3 percent of the total allowable quota allocated to long-liners in an experimental fishery (Stuttaford, 1995, 91). Early information from the pilot study suggests that shifting the hake fishery away from trawling toward long-lining has potentially dramatic implications for the industrial structure of the fishery. To start with, while barriers to entry in the hake fishery are typically high, they are considerably lower for long-lining than trawling, with deep-sea trawlers costing 15-20 times as much as the R1 million needed to purchase a purpose-built long-line vessel (*Cape Business News*, 1997d). Moreover, although cost per ton of long-lined hake is two-and-a-half times the cost for trawled hake, the quality of long-lined hake is superior to trawled hake, such that prices are two-and-a-half times as great (Japp, 1995).

In terms of the model developed in this study, the implications of the cost and price effects are such that while the optimal dynamic catch remains unchanged, the absolute size of rents earned by operators would increase by a factor of 2.5. In addition to having lower capital requirements in the harvesting sector, long-lining would also have implications for employment: long-lining is relatively more labour-intensive than trawling. The possibility also exists that operators would pass on some portion of the higher rents to workers in terms of wages, profit-sharing, benefits or conditions of employment.

In short, long-lining has the potential to generate significant benefits for participants in the hake fishery, particularly small-scale fishers. As Diaw (1992) has noted, the potential for small-scale fishers to facilitate socio-economic development is considerable,[35] and Haakonsen (1992) has identified a number of areas in which small-scale fishers have a meaningful advantage over commercial fishers. However, there is also evidence to suggest that small-scale fishers encounter a substantially greater number of difficulties than commercial fishers. The list of potential problems is considerable, and includes lack of collateral (Haakonsen, 1992, 49); an absence of markets, such as insurance markets (Platteau, 1992); an absence of backward and forward linkages (Hersoug, 1992); conflicting policy goals and project objectives (Hersoug, 1992); managers' ignorance of the socio-economic characteristics of small-scale fisheries (Jul-Larsen, 1992; Johnsen, 1992; Lindqvist and Mölsä, 1992); and problems with the adoption or adaptation of technology (Bækgaard and Overballe, 1992; Skjønsberg, 1992). Moreover, there are indications that long-liners tend to catch larger, more fecund fish than trawlers. While this has obvious implications for the biomass and yields, these are not yet well understood by scientists.[36] None the less,

[35] Small-scale fisheries are typically characterised by relatively low capital intensity and by the ease with which fishing communities appropriate fishing technology (Platteau, 1989, 568; Holland, 1995, 1). However, in providing this description, it should be cautioned that while some distinction between commercial and small-scale fishers must be made, the division is often 'fuzzy', and can be ideologically biased and misleading. Accordingly, extreme caution should be exercised in formulating policy which makes a distinction between commercial and small-scale operators. See Loayza (1992) for a more detailed discussion of the classification of fisheries.

[36] Per. comm. (Leslie, Sea Fisheries Research Institute).

the potential that long-lining has to facilitate socio-economic upliftment remains considerable, and although the hake long-line experiment is in its infancy, it is immediately apparent that the issue of alternative technologies and their socio-economic consequences deserves fuller consideration.

7.4.6 Management Effectiveness and Efficiency

Successful fisheries management hinges upon its effectiveness and efficiency. Effectiveness, because weak enforcement may result in, for example, poaching by foreign or domestic fleets which undermines management outcomes, resulting in partial or complete rent dissipation. Efficiency (that is, least cost), because the costs of controlling fisheries exploitation have the potential to reduce (or, in the extreme, completely dissipate) rents (Sutinen and Andersen, 1985). As Wilson (1982, 425) has noted, costs do not only include set-up costs, which may be extensive, but also the on-going costs of enforcement and administration of rights and obligations.

Considering the potential for effective management in an ITQ system Townsend (1992, 186) has noted: 'ITQs have their own Achilles' heel: enforcement.' Quotas (individual or otherwise) are most easily enforced when all landings must flow through some easily monitored 'choke point'. This choke point may be on the vessel, at the dock, at the processing plant or at the export control point (for exported species). Enforcement is difficult if such a restricted flow does not exist. For example, small boat fisheries selling directly to consumers represents the most difficult enforcement situation.[37] With regard to cost efficiency, the feasibility of ITQs is undermined if harvests are highly variable and unpredictable, as the administrative costs and difficulties of running such a system become considerable (Munro and Scott, 1985, 664; Copes, 1986, 283).

[37] For example, problems encountered in ITQ enforcement in the Atlantic purse seine fishery for herring in Canada and the mackerel fishery of south-west England are well documented in Crouter (1985) and (Derham, 1985) respectively.

The South African hake fishery escapes both of the above difficulties. As noted in Chapter 2, the bulk of hake landings occur at two main ports. Furthermore, the capital pool and revenue streams generated by the proposed ITQ system would be able to finance a relatively sophisticated administration and monitoring network. Finally, unlike pelagic stocks, for example, the hake biomass is relatively slow growing and exhibits low variability (see Chapters 2 and 4). However, two problems that are likely to arise under ITQs, and which have the potential to undermine the effectiveness of management, are the problems of 'high-grading' and 'by-catch'; and the presence of multiple processing forms.

As New Zealand's recent experience shows, the adoption of ITQs presents the additional problems for regulators of 'high-grading' and, in multi-species fisheries, 'by-catch depletion' and 'discarding' (DuPont, 1991, 163; Turner, 1997). The problem of high-grading occurs where fishers are faced with a limited quota. In this instance, operators may attempt to maximise the value of their quota by sorting their catch, that is, throwing back lower-valued (smaller or damaged) target-species fish. Similarly, in the instance of multi-species fisheries, fishers will unavoidably catch some fish of species for which they do not hold quotas. This 'incidental' catch or by-catch gives rise to two problems (Clark, 1982, 285): depletion, whereby non-target species become depleted primarily because they are a by-catch of the target species; and 'discarding', whereby, as with high-grading, 'trash' fish are discarded at sea in order to accommodate more valuable species in the vessel's hold (high-grading).

As was noted in Chapter 4, there is sufficient evidence to conclude that high-grading has taken place on a considerable scale in the hake fishery, leading to stock depletion. Moreover, the hake fishery is characterised by a high incidence of by-catch. For example, in 1991, the offshore trawl fishery by-catch was made up of, *inter alia*, horse mackerel (25 260 tons), snoek (14 486 tons), ribbon fish (10 598 tons) and monkfish (5 816 tons) (Bergh and

Barkai, 1993, 98), collectively equal to about one-half the hake catch by live-weight.

Given that both the high-grading and by-catch problems have the potential to occur on a significant scale, it becomes necessary to question whether regulation by way of ITQs remains optimal. The answer, as Clark (1982, 286) has noted, is not clear. No obvious solution – other than 'direct' monitoring – to the high-grading problem presents itself, and a tax or subsidy on some species may be necessary to prevent discarding.[38] However, while it is not clear whether ITQs remain 'optimal' in the presence of the high-grading and by-catch problems, there is no reason to expect either problem to dissipate under some other regulatory regime, such as vessel licensing, closed seasons or, for that matter, taxation. These appear to be problems of regulation *per se* rather than any given form of regulation. Based on these arguments, it is accepted here that, in the absence of evidence or arguments to the contrary, ITQs remain 'optimal', but further work on the high-grading and by-catch problems is urgently required.

Finally, in considering the cost efficiency of enforcement, it is useful to turn to the New Zealand experience which provides a number of important insights. In October 1986, an ITQ management system was introduced into the New Zealand fisheries for all significant commercial species (Clark *et al*, 1989, 117). Prior to the introduction of the quota system, New Zealand enforced its fisheries policies through a standard game warden approach, apprehending law-breakers and providing a presence to discourage illegal behaviour. However, effective management by this method is costly and was seen by the government of New Zealand as contrary to the spirit of the new management approach, which emphasised deregulation, reduced regulatory intervention by

[38] In the case of multi-species fisheries where there is by-catch, regulators could allocate quotas for each species according to some formula based on, say, historic catch relationships. But, chances that operators' catches conform even modestly with the proportions of the various species quotas are almost nil (Copes, 1986, 285). This second problem could, perhaps, be overcome by a variety of mechanisms, such as allowing fishers to smooth out fluctuations over the longer term, to sell or purchase 'surplus' or 'deficit' quotas or to average their catch across groups or consortia. See Boyce (1996) for an economic analysis of the by-catch problem.

government, economic efficiency and cost effectiveness. The focus of enforcement was therefore changed: the new role of the enforcement authorities became not so much policing fishers as monitoring, following product flow, and seeking to establish a paper trail from the fishing vessel to retail disposal. Enforcement activity now takes place more on land than at sea, and is carried out by people who are more auditors than game wardens. The result has been an enforcement system which is not only cost efficient, but also effective in preventing illegal activity (Clark *et al*, 1989, 137). Similar efficiencies could be enjoyed in the regulation of the South African hake fishery, and in the regulation of fisheries generally.

It ought to be noted that while the policy recommendations made in this chapter are in broad agreement with the regulatory framework proposed under the *Marine Living Resources Bill* (Republic of South Africa, 1997),[39] there are a number of significant differences *vis-à-vis* the specific form to be taken by the regulatory framework. The most significant of these differences are outlined below.

i) It is recommended in the Bill that access rights be granted for a limited period (fifty years), with rights being eroded over that period. This is in contrast to the recommendation made in Section 7.4.3 of this study that rights be allocated in perpetuity. In line with the arguments presented in that section, the allocation of 'stinted' rights is likely to undermine economic efficiency and precipitate intergenerational equity problems.
ii) In contrast to the argument presented in Section 7.4.4 of this study – that ITQs be perfectly divisibility and freely tradable – it is recommended in the Bill that quotas be divisible and tradable subject to the discretion of the Minister of Environmental Affairs and Tourism. This recommendation is likely to limit tradability of ITQs and, in so doing, undermine the efficiency of the regulatory system. Furthermore, no guidelines are offered as to what criteria the Minister might adopt in the approval of transfers.
iii) It is proposed in the Bill that moneys received in respect of, *inter alia*, fines, penalties, donations and parliamentary appropriations should accrue to the

[39] Hereafter 'the Bill'.

Sea Fishery Fund,[40] but that moneys paid in respect of right of access and annual fees should accrue to central government (the National Revenue Fund). Against the background of the funding system proposed in Section 7.4.1, in which right of access and annual fees are argued to provide the funding basis for the proposed regulatory framework, it is argued that the set of funding proposals made in the Bill threaten to strip the management system of the potential to sustain itself and, more importantly, of the ability to pursue socio-economic objectives, such as the promotion of the small business sector or the funding of research activities.

iv) Whereas the arguments set out in Section 7.4.2 are in favour of a volume-based ITQ system, the Bill proposes the issuing of percentage-based quota. It is argued in Section 7.4.2 that the adoption of a percentage-based system is likely to encounter problems where authorities attempt to achieve reductions in annual quota. The percentage-based system is also likely to result in higher levels of uncertainty (which is argued to promote the 'incentive for fishers to cheat').

v) The Bill proposes that enforcement should continue to take the form of 'policing' through a 'hands-on' approach, rather than through 'monitoring' in a deregulated, 'hands-off' environment. However, in line with the arguments presented in Section 7.4.6, it is suggested that the adoption of a 'hands-on' approach to policing is likely to be ineffective and to undermine the efficiency of the ITQ system.

In light of the arguments set out in this chapter, and the available evidence relating to experiences in other countries, it is submitted that these issues, in particular, deserve fuller consideration.

[40] The Sea Fishery Research Fund was created by the Sea Fishery Act No. 12 of 1988 as the basis for financing the management – defined to include administration, regulation, research and so on – of the sea fisheries.

7.5 SUMMATION OF POLICY RECOMMENDATIONS

In summation, while the gains in economic rent achieved by shifting the South African hake fisheries from their extant (approximating $f_{0.2}$) positions to the dynamic maximum economic yield position are different to those hypothesised in Chapters 5 and 6, the gains remain considerable. Based on this result, there can be little doubt that adopting a dynamic rent maximising target as the basis for managing the fisheries has the potential to yield considerable welfare gains. The problem then becomes one of establishing the optimal regulatory approach. To this end, at the broadest level, three approaches present themselves: self-regulation, non-regulation or regulation by some authority. In the case of a competitive, commercial fishery, it has been argued that the two first-mentioned approaches are inappropriate. Accordingly, the problem becomes one of establishing the optimal regulatory framework. Historically, fisheries managers have opted for direct or non-market interventions. The success of this approach has been limited. Experience shows that while direct intervention has been able to prevent the Class I problem of over-fishing, it has, in many instances, led to a second rent depletion problem of 'capital stuffing', the so-called Class II problem of open-access. Accordingly, more recently, the attentions of regulators have switched to consider the feasibility of market-based instruments.

Two market-based instruments are suggested in the literature: Pigouvian-type taxes, and the creation of private property rights in the form of ITQs. While, in theory, taxes and ITQs are equivalent, in practice, taxes run into a host of problems which reduce the feasibility of taxation as a management tool. ITQs, on the other hand, not only embrace the notion, and satisfy the criteria, of economic efficiency, but have also proved to be feasible and effective in practice. However, ITQs are capable of taking on a number of forms. The arguments presented above suggest a form of ITQ which, it is believed, is likely to present managers of the South African hake fishery with an efficient

and effective regulatory package, incorporating the sale of 'perpetual right, volume-based, transferable and perfectly divisible' ITQs which:
(i) are allocated on the basis of 'grandfathering' principles;
(ii) allow for the adoption of a 'low but effective' annual tax;
(iii) embrace the principles of 'phased' adjustment to ensure minimisation of inefficiency through forced retirement of labour or capital;
(iv) facilitate efficient and effective enforcement by 'monitoring' rather than 'policing' fishing activity; and
(v) allow for maximum gains in efficiency through the transferability of quotas across technologies.

In summary, it is concluded here that bioeconomic management, through the use of appropriate market-based regulatory tools, has the capacity to restore and preserve the integrity of the hake fishery. More importantly, it is concluded that an appropriately designed market-based ITQ system has the capacity to ensure that the welfare and socio-economic contributions made by the hake fishery to the South African economy are maximised on a sustainable basis.

CHAPTER 8: CONCLUSION

The central concern of economics, as a discipline, lies with the optimal utilisation and allocation of scarce resources as the basis for maximising social welfare. Within the context of resource economics, economists have historically tended to focus on the use and management of non-renewable resources. The reasons for this are readily apparent. By definition, non-renewable resources are relatively more scarce than renewable resources, which serves to escalate the threat of resource exhaustion and jeopardise sustainable economic growth and development. Furthermore, historically, non-renewable resources (such as iron, copper and fossil fuels), rather than renewable resources, have served as the platform for economic growth and development. By contrast, so long as the harvesting of renewable resources did not exceed the natural regeneration rate, the continued availability of renewable resources was seen as relatively unproblematic.

More recently, however, the concern of economists has shifted to incorporate the optimal utilisation and management of renewable resources. This shift in emphasis is due to a variety of factors although, arguably, the most important of these is the recognition that sustainable economic growth and development hinges critically upon the optimal use and management of both renewable and non-renewable resources. In this regard, starting with the seminal work of Gordon (1954), considerable advances have been made in the theory of optimal use and management of renewable resources. However, these developments have not exerted a widespread impact upon policy formulation. The unfortunate result has been the over-exploitation of a wide range of renewable resources on a global scale. South Africa has not escaped this experience; indeed, there exists a substantial body of evidence highlighting the unsustainable use and mismanagement of renewable resources in South Africa.[1]

[1] Evidence of the over-exploitation of renewable resources is presented in Section 1.2 of Chapter 1.

A central argument of this study is that the over-exploitation and mismanagement of renewable resources is ascribable (in South Africa as well as in other countries) to three main factors. To start with, contrary to the prescriptions of Gordon (1954) and others, in many instances renewable resources have gone unprotected. This has resulted in the 'mining' of resources, that is, exploitation of renewable resources at rates that are not sustainable. Second, where authorities have attempted to manage renewable resources, they have tended to adopt biological principles, such as maximum sustainable yield, as opposed to the bioeconomic principle of rent maximisation prescribed by Gordon (1954) as the basis for managing the use of renewable resources. That is to say, authorities have adopted management objectives that fail to maximise rents generated by the exploitation of renewable resources. Third, authorities have generally relied upon direct controls as the basis for the regulation of renewable resource use, but with limited success. Thus, in spite of regulations, authorities have largely failed to prevent over-exploitation of renewable resources.[2]

Against this background, the aims of this study are two-fold. Based on the principles of resource economics developed over the last four decades, this study seeks to develop and apply a bioeconomic model of optimal resource use in the South African context. In addition, given the results of the modelling process, and by drawing on the substantial body of literature and evidence that has emerged on the subject of the management of renewable resources, this study seeks to provide some guide as to the appropriate form resource management should take if the goal of dynamic rent maximisation is to be achieved in the utilisation of renewable resources.

This study is undertaken with reference to South Africa's renewable marine resources. There are a number of reasons for focusing on renewable marine resources. As noted in Chapter 2, South Africa's renewable marine resources are of considerable socio-economic importance. To be more

[2] These arguments, and the associated evidence, are detailed in Section 1.3 of Chapter 1.

specific, the industry is a small but significant contributor to gross domestic product, employment, investment, wealth creation and foreign exchange earnings. Moreover, the industry has spill-over effects into other activities such as boat-building and maintenance. The fish stock also serves as the basis for the commercially important recreational fishery. In addition, the industry is an important supplier of nutrition for both humans and animals. Furthermore, the importance of the fishing industry is likely to grow as the demand for fish and fish products increases as a result of domestic and international growth in population and *per capita* incomes, as well as shifts in consumption patterns towards fish, and shifts in production patterns toward value-added products.[3]

A further reason for focusing on South Africa's renewable marine resources exists. There is considerable evidence to suggest that, in spite of a plethora of controls and regulations, South Africa's renewable marine resources have been severely over-exploited.[4] Historically, South African fisheries have been subject to biological over-exploitation, that is to say, stocks have been over-fished such that the yield (growth) of the biomass is below the biologically determined maximum sustainable yield. In the face of such over-exploitation, numerous efforts have been made to reduce the rate at which stocks are harvested. However, in many cases, these efforts have failed to reverse the effects of over-exploitation, with a large proportion of stocks remaining in a severely depleted state. A number of factors have contributed to this failure. Regulations have promoted higher levels of concentration of ownership within and between the harvesting and processing sectors. In turn, higher levels of concentration have provided incentives for illegal activity, which has exacerbated problems of over-exploitation. Stock management has been further undermined by an inadequate policing system. Arguably the most important explanation for over-exploitation lies in the fact that the management of South Africa's commercial fisheries has been based primarily upon biological principles. A central argument of this study is that biological

[3] The socio-economic importance of the South African fishing industry is described in greater detail in Sections 2.2 and 2.3 of Chapter 2.
[4] Each of these problems is more fully explored in Section 2.4 of Chapter 2.

principles are a necessary, but insufficient, basis for the management of fisheries (and, indeed, renewable resources in general).[5]

The substantial origins of renewable resource economics can be traced back to Gordon (1954), who employed a static surplus production model (based on the well-known Schaefer function of population growth under harvesting) to show that, unless regulated, open-access renewable resources, such as fish stocks, will be driven to a point at which the economic rent (revenue less economic costs) of renewable resource extraction is zero (the bionomic outcome). This is the so-called Class I problem of open-access renewable resources. Hence, it is necessary for authorities to regulate the use of renewable resources to prevent the over-exploitation of these resources. Historically, authorities have responded by adopting biological goals, such as maximum sustainable yield (or some derivative thereof), as the basis foı sustainable exploitation of renewable resources. It is in this regard that Gordon (1954) provided a further important principle *vis-à-vis* the 'optimal' exploitation of renewable resource. Specifically, Gordon went on to show that biological principles are an incomplete basis for the regulation of renewable resource use. Rather, the optimal exploitation of renewable resources is given by maximisation of economic rents.

While these principles of renewable resource use developed by Gordon have served as the centrepiece for all subsequent developments in the theory of renewable resource economics, in retrospect, Gordon's work was deficient ir a number of ways. Since the mid-1950s, however, a vast literature has emerged to cater for these various deficiencies. Arguably the most successful extension of Gordon's work has involved shifting the model into a dynamic setting whereby the dynamic rent maximising biomass (X^*_{MEY}) is given by:

$$G'(X^*_{MEY}) - \frac{c'(X^*_{MEY})G(X^*_{MEY})}{p - c(X^*_{MEY})} = \delta, \qquad \text{(Equation 3.18)[6]}$$

[5] A detailed examination of this argument is offered in Chapter 3.

with the dynamic rent maximising biomass, X^*_{MEY}, lying somewhere between the bionomic equilibrium ($\overline{X}$) and the static rent maximising biomass (X_{MEY}).

It is argued in Chapter 3 that this dynamic surplus production model is well-suited to the central task of this study, that is, developing and applying a bioeconomic model of South Africa's commercial fisheries. The model is easily adapted to reflect a range of biological and economic characteristics.[7] Furthermore, a number of restrictive assumptions of the model, including linearity and autonomy in costs and prices, are easily relaxed. The model can also be extended to allow for 'heterogeneous' fisheries (such as South Africa's hake or rock lobster fisheries) which consist of biologically distinct stocks. Nevertheless, the model does suffer from a number of limitations, the most serious of which is a failure to incorporate the problems of uncertainty and species interaction.[8] However, the issue of uncertainty is extremely complex and there are large gaps in the literature. Furthermore, efforts to explore the issue of uncertainty have generally not translated into real changes in modelling. Moreover, while several authors have extended single-species analysis to include multi-species modelling – including competing species; predator-prey systems; and harvesting interactions in multi-species systems – multi-species modelling is an extremely complex task. Indeed, it is argued by some writers that, more often than not, the undertaking proves to be impossible. For these reasons, the biological model employed in this study does not attempt to incorporate uncertainty or multi-species interaction.[9]

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[6] A detailed discussion of the salient features of the dynamic bioeconomic model is undertaken in Chapter 3.
[7] As noted in Chapter 3, for example, while the model set out in Equation 3.18 is based on the Schaefer growth function, it is easily adapted to other population growth functions. The model is also easily extended to allow for alternate formulations of the growth function in terms of which, for example, extinction of fish stocks is feasible.
[8] Fuller considerations of the issues of uncertainty and multi-species harvesting are offered in Section 3.4 of Chapter 3.
[9] The issue of multi-harvesting technologies is, however, explored in Chapter 7.

Turning to consider the application of the model, it was noted in Chapter 2 that South Africa's fishing industry is extremely diverse. For this reason, it is necessary to construct a number of models to represent the various sectors of the industry. This is a considerable task, which is well beyond the scope of the current study. Instead, this study focuses on the country's most important commercial fishery, the hake fishery, which represents approximately one-third of the commercial catch by value and one-quarter by mass. In order to apply the dynamic bioeconomic model to the South African hake fishery, however, it is first necessary to examine the major biological and economic features of hakes and the hake fishery to inform the modelling process. This task, undertaken in Chapter 4, brings forward a number of important points.

Considering biological features first, the South African hake stock is made up of two species, namely *M. capensis* and *M. paradoxus* (shallow-water hake and deep-water hake respectively), which are primarily distributed along the west and south coasts. While differences between rates of growth in species and sex have been noted, these differences are not taken into account explicitly in modelling, due to the inability of fishers to target fish by species or sex. However, differences in growth rates of hake species are taken into account indirectly. In this regard, the modelling process recognised the hake stock as being distinguished along geographical lines, with the fishery divided into two biologically distinct stocks: the west coast and south coast hake stocks.[10] This allows for differences in growth rates between species to be taken into consideration in the modelling of the fisheries. Specifically, estimates of the instantaneous growth rate of hakes in the south coast fishery are typically modelled as being twice as great as the growth rate in the west coast fishery. This is due the relative abundance of the faster growing *M. capensis* (as well as favourable environmental factors in the south coast fishery). It should also be noted that, mainly due to differences in upwelling conditions, the carrying capacity of the west coast fishery (approximately

[10] The south coast fishery is further divided into an inshore fishery and deep-sea fishery although, for the purposes of modelling, no distinction is made in the south coast fishery between these fisheries.

1 665 000 tons) is generally modelled as being five to six times as great as that of the south coast fishery (approximately 275 000 tons).

As far as the economic features of the hake fishery are concerned, a number of points should be noted. First, biological features of the hake stock play a primary role in determining a number of central economic characteristics of the South African hake fishery. For example, the joint effect of differences in carrying capacities of the two fisheries and growth rates of the hake stocks is that the hake catch is typically taken in the ratio 2:1 in favour of the west coast fishery. More important, however, is the result that the bulk of the annual hake catch is taken in the deep-sea bottom-trawl fisheries, with a small portion of the catch (approximately one-twentieth of the total catch) taken in the inshore bottom-trawl fisheries. Alternative harvesting technologies, which incorporate the mid-water trawl, handline and longline fisheries, account for an insignificant portion of the total catch.[11] For this reason, this study focuses on the bottom-trawl fishery.

Second, while open-access conditions led to rapid growth in the hake fishery over the first half of the century, by the early 1970s the resource had become severely over-exploited.[12] This resulted in the introduction of a number of direct controls – which included the ejection of foreign fleets from the fishery; the adoption of various vessel and gear restrictions; and the issuing of a 'global' quota (and later company quotas) based on the 'biologically conservative' $f_{0.2}$ harvest strategy – aimed at preventing or reversing the effects of over-exploitation. While these controls did facilitate a recovery, albeit modest, in hake stocks over the course of the 1980s, regulations have also had adverse effects on the industry.[13] For example, the set of direct controls adopted served to promote excessive concentration of ownership and control in the fishery. More significant is the argument that, regardless of

[11] See Section 4.3 of Chapter 4 for a more detailed discussion of the breakdown of catch by sector.
[12] Detailed evidence of over-exploitation of hake stocks is presented in Section 4.3 of Chapter 4.
[13] The nature and extent of the recovery in hake stocks is detailed in Sections 4.3 and 4.4 of Chapter 4.

the degree of success enjoyed by fisheries managers in reversing the over-exploited state of the hake resource, the management strategy has relied almost entirely on biological principles. This is in direct contrast to the central theoretical arguments of renewable resource economics, as set out in Chapter 3. In short, South Africa's fisheries managers have yet to incorporate economic principles into the management of fisheries. In an effort to provide a first step in overcoming this shortcoming in renewable resource management in South Africa, Chapters 5 and 6 of this study are devoted to applying the dynamic bioeconomic model developed in Chapter 3 to South Africa's hake resource.

Applying the dynamic bioeconomic model to South Africa's hake fishery requires reducing the dynamic bioeconomic model to a form in which it is easily estimated. As noted, the dynamic rent maximising biomass (X^*_{MEY}) is determined by the fundamental equation (Equation 3.18) set out above. Furthermore, in the event that the surplus production function [$G(X_t)$] is the Schaefer logistic function, Equation 3.18 reduces to a quadratic equation, from which the following closed formula for X^*_{MEY} is readily obtained:

$$X^*_{MEY} = \frac{1}{4}\left\{\bar{X}_t + K(1-\frac{\delta}{r}) + \sqrt{\left[\left(\bar{X}_t + K\left(1-\frac{\delta}{r}\right)\right)^2 + \frac{8K\bar{X}_t\delta}{r}\right]}\right\}. \quad \text{(Equation 5.1)[14]}$$

It should be emphasised that while the model developed in Chapters 3 and 5 (as set out in Equation 5.1) is based on the Schaefer population growth function – as per Gordon (1954) and others (Clark, 1976 & 1985) – the model is easily adapted to different specifications of the growth function, such as those discussed by Pella and Tomlinson (1969), Fox (1970) and Shepherd (1987). Nevertheless, in line with arguments presented elsewhere in this

[14] A detailed discussion of the main features of the model and its parameters is undertaken in Chapter 5.

study, it is assumed that the Schaefer function constitutes the 'best' choice of function in the case of the South African hake resource.[15]

The next step in the modelling process requires establishing values for the various biological and economic parameters of the model, namely, the instantaneous growth rate of the fish stock (r), the size of the pristine biomass (K), the catchability coefficient (q), the price of fish (p), the cost of harvesting (c) and the discount rate (δ), as set out in Equation 5.1. For convenience, the parameters are divided into two subsets: biological parameters (r, K and q) and economic parameters (δ, c and p).

Estimates of biological parameters values are based on those generated by scientists at the Sea Fisheries Research Institute (Cape Town) through the application of a Schaefer-type surplus production model to commercial data, and 'fine tuned' with the aid of supplementary analyses. However, a number of caveats should be noted with regard to the given set of estimates. First, although estimates of parameter values are available from the inception of the fisheries, this study is based on observations drawn from 1984 onwards. This is because changes in the structure of the fishery and corruption of the data make the use of earlier data highly questionable. Second, contrary to the assumptions of the dynamic bioeconomic model, biological parameter estimates are not constant across time. While there is evidence to suggest that variability in parameter values may be due to model misspecification, a more likely explanation lies in the inaccuracy of catch and effort data due to changes in industry structure and harvesting techniques that have not been factored into the figures. Thus, remodelling of the data is required to eliminate bias in parameter estimates. However, data remodelling runs into a number of problems leading some writers (Bergh and Barkai, 1996) to argue that, until these problems are resolved, managers should continue to employ extant estimates. This study adopts a slightly modified stance. Given that the estimates of the biological parameters vary across time; that substantial data problems exist; and that the modelling process is not likely to

[15] See Section 5.3 of Chapter 5.

be able to model effects such as random environmental shocks, it is argued that range values, which allow for a considerable margin of error (rather than point values), based on extant parameter estimates provide for a more sensible modelling procedure. For this purpose, a 95 percent confidence interval is constructed to represent the 'true' values for each of the biological parameter estimates.[16]

Turning to consider estimates of economic parameter values, a number of points should be noted. To start with, given the assumptions of autonomy and linearity in prices and costs, and given South Africa's inflationary environment, this study employs price and cost figures based on operating margins – reflecting the real price:cost ratio – as opposed to nominal price and cost data. The task of establishing operating profit margins is, however, complicated by a number of factors, particularly the absence of reliable cost data at the fishery level. None the less, as demonstrated in Section 5.3 of Chapter 5, by coupling information made available at the company level with other industry information, it is possible to arrive at a reasonably reliable set of estimates of operating margins in the trawl fishery for hake. In the present study, a margin of 20 percent is assumed in the first instance. With regard to estimates of the discount rate, as opposed to employing conventional discounting, this study employs the method of quasi-discounting to convert future costs and benefits into net present values, a procedure which is based on the argument that project evaluation involves the imputation of utility directly to each act. There is nothing novel about this approach. Indeed, the adoption of project-specific or resource-specific discount rates is widely accepted. It is argued in Chapter 5 that extrapolating reasonable values for rates of growth of consumption and taking a moderate elasticity of marginal utility of consumption produces a quasi-discount rate for a reasonably slow growing renewable resource, such as hake, of 3-6 percent per annum. Based on this result, a mean discount rate of 4.5 percent per annum is assumed in this study.

[16] The major features of the range estimates are discussed in Section 5.3.1 and summarised in Table 5.2 of Chapter 5.

Having established a set of values for the various model parameters, the concern of this study then shifts to an estimation of the optimal dynamic outcome (as per Equation 5.1) in the west and south coast hake fisheries. For comparative purposes, results are also generated for the dynamic maximum sustainable yield biomass, X^{*}_{MSY} (as per Equation 5.3); as well as the static maximum economic yield biomass, X_{MEY} (as per Equation 5.2). In addition, the model calculates the total catch (Equation 3.9) and effort (Equation 3.7) levels associated with each biomass outcome. The central features of these results are presented below in Table 8.1.

Table 8.1: Biomass, Catch and Effort Levels under the Dynamic Bioeconomic Model

Maximum Sustainable Yield				Maximum Economic Yield		
Biomass (Tons)	Catch (Tons)	Effort (Days)	West Coast Hake Fishery	Biomass (Tons)	Catch (Tons)	Effort (Days)
621 782	130 893	15 074	Sample Mean	878 765	125 714	10 305
113 485	11 561	365	Sample Standard Deviation	187 397	9 719	450
18.25	8.83	2.42	Coefficient of Variation (%)	21.33	7.73	4.37
Biomass (Tons)	Catch (Tons)	Effort (Hours)	South Coast Hake Fishery	Biomass (Tons)	Catch (Tons)	Effort (Hours)
134 146	52 478	98 263	Sample Mean	205 169	42 708	52 319
9 079	1 371	704	Sample Standard Deviation	15 897	1 321	1 023
6.77	2.61	0.72	Coefficient of Variation (%)	7.75	3.09	1.96

Source: See Tables 5.4 and 5.5 of Chapter 5.

The results of the bioeconomic modelling process have implications of varying (but significant) proportions for the west and south coast fisheries. For the west coast fishery, the dynamic bioeconomic model calculates that, on average, the rent maximising biomass is approximately 40 percent greater than the maximum sustainable yield biomass. In turn, the rent maximising biomass produces a sustainable catch which is, on average, 3.96 percent less than the maximum sustainable yield. However, the average level of effort required to take the dynamic rent maximising catch is approximately 30 percent less than that required to take the maximum sustainable yield. This outcome, in turn, precipitates significant gains in economic efficiency (catch per unit effort is modelled to increase by an average of more than 40 percent). That is to say, allowing the west coast biomass to 'rebuild' from the yield maximising level to the rent maximising level, produces a higher catch on the back of a substantial fall in the required level of effort. In turn, the combination of higher catch and lower effort generates a significant improvement in economic efficiency.

For the range of biological parameters adopted, the dynamic bioeconomic model calculates that the rent maximising biomass in the south coast fishery is, on average, approximately 50 percent greater than the maximum sustainable yield biomass. In turn, the rent maximising biomass produces a sustainable catch which is, on average, four-fifths the size of the maximum sustainable yield. However, the level of effort required to take the rent maximising catch is, on average, only one-half of that required to take the maximum sustainable yield. This outcome, in turn, precipitates an average improvement in catch per unit effort in the south coast fishery of the order of 55 percent, representing substantial improvements in fishing efficiency. Thus, allowing the south coast biomass to 'rebuild' from the yield maximising level to the rent maximising level, results in a fall in catch accompanied by an even greater fall in the required level of effort, the net result of which is a significant improvement in economic efficiency.

The results of this modelling process have a number of implications for social welfare. First, based on the theory presented in Chapter 3, it is accepted, *a priori*, that shifting the fishery from the dynamic maximum sustainable yield to dynamic maximum economic yield position involves an improvement in social welfare. Thus, the concern becomes one of quantifying the extent of gains in economic rents. This is achieved by re-employing the price-per-ton:cost-per-standard-boat-day (or standard-boat-hour) ratios as derived in Section 5.3 of Chapter 5 to convert catch and effort data into revenue and cost data. The results of this procedure are summarised in Table 8.2.[17]

Table 8.2: Economic Rent under the Dynamic Maximum Sustainable Yield and Dynamic Maximum Economic Yield Strategies in the West and South Coast Fisheries

	Dynamic MSY (1)	Dynamic MEY (2)	Increase of (2) over (1) (Percent)
	Index Points		
West Coast Fishery Index	42 712.11	65 424.09	57.34
South Coast Fishery Index	1 381.24	15 503.52	4 362.30
Combined Average	44 093.35	80 927.61	83.54

Source: See Table 5.6 of Chapter 5.

The results of this modelling process, summarised in Table 8.2, show that shifting the west and south coast fisheries from a dynamic maximum sustainable yield outcome to a dynamic maximum economic yield outcome would generate gains in economic rents averaging 57.34 percent and 4 362.30 percent in the west and south coast fisheries respectively. However, a better picture of the extent of potential gains in social welfare is given by summing the figures for both fisheries which shows that shifting the fisheries from the dynamic maximum sustainable yield outcome to the dynamic maximum economic yield outcome would approximately double the rent generated by the hake resource. As an aside, it should be noted that these increases in rents are achieved on the back of (often considerable) adjustments to the level of biomass, catch and effort in the fishery which implies alterations to the structure of the fishery (as per Table 8.1). This issue is complex, and fuller comment is reserved for Chapter 7 where the

[17] It should be emphasised that in the absence of accurate price and cost data, these results should not be read as monetary values, but rather as relative values. This has no influence on the significance of the results.

implications of adopting bioeconomic, as opposed to biological, management principles are considered in closer detail.

Before considering the policy implications of bioeconomic management, it is first necessary to examine the sensitivity (or robustness) of the dynamic bioeconomic model to fluctuations in parameter values. There are at least two reasons for this. Given the problems involved in arriving at accurate parameter estimates, it follows that it is vital – at least from a management perspective – that the model outputs be tested for sensitivity to fluctuations in values of the various parameters.[18] However, a more fundamental reason exists for examining the relationship between parameter values and the model outputs. The dynamic Gordon model assumes that the values of biological and economic parameters are constant across time. There is evidence to suggest that these assumptions are flawed, with the implication being that even if it were possible to arrive at reliable estimates of parameter values, these values are expected to change across time.[19] Against this background, Chapter 6 is devoted to the task of testing the sensitivity of the model to fluctuations in parameter values.

Extending the dynamic bioeconomic model to allow for non-autonomy in parameter values requires restating Equation 5.1 to provide a clear guide to the relationship between the dynamic rent maximising biomass (X^*_{MEY}) and each of the biological and economic variables. This is done by expanding Equation 5.1 to explicitly include c, p and q as determining variables of the dynamic rent maximising biomass, such that:

[18] Because range values, as opposed to point values, are employed as the basis for modelling the hake fisheries, the impact of variable biological parameter values has already been considered (in Chapter 5). However, more than simply testing the sensitivity of model estimates to fluctuations in biological parameter values, the task of Chapter 6 shifts to one of extending the model into a variable parameter setting and, in so doing, providing a more detailed consideration of the impact of variable biological (as well as economic) parameter values on the dynamic bioeconomic outcome.

[19] This evidence is reviewed in Section 6.2 of Chapter 6.

$$X^{*}_{MEY} = \frac{1}{4}\left\{ \frac{c}{pq} + K(1-\frac{\delta}{r}) + \sqrt{\left[\left(\frac{c}{pq} + K\left(1-\frac{\delta}{r}\right)\right)^2 + \frac{8K\frac{c}{pq}\delta}{r}\right]} \right\}. \quad \text{(Equation 6.1)}$$

In addition, given the formulae for catch (Equation 3.9) and effort (Equation 3.7) associated with any given biomass, it is possible to establish the nature of the relationship between the dynamic rent maximising catch and effort of each of the model parameters. Furthermore, by employing the methodology set out in Chapter 5, it is possible to establish the impact of variable parameter values on welfare levels.

The analysis with regard to the impact of variable parameters on the bioeconomic model, undertaken in Chapter 6, produces a number of significant results.[20] Considering the impact of variable biological parameter values, it is readily apparent from the model presented above (Equation 6.1) that relationships exist between each of the biological parameters and the dynamic rent maximising biomass and, therefore, with the dynamic rent maximising catch and effort levels.[21] However, the estimates produced under the variable parameter model reveal that the dynamic rent maximising (as well as dynamic yield maximising) biomass, catch and effort levels for both the west and south coast fisheries are highly insensitive to fluctuations in biological parameters values.

Specifically, estimates of dynamic rent maximising biomass, catch and effort in the west and south coast fishery are calculated to fluctuate by less than one-half of the level of variation allowed for in estimates of biological

[20] The analysis undertaken with respect to the relationship between the biological parameters and the dynamic rent maximising outcome is based on the assumption of biological parameter 'scenarios' (as per Chapter 5), rather than examining the responsiveness of the outcome to variations on a parameter-by-parameter basis. The reason for this is that, based on the evidence presented in Section 6.2.1, and as per the parameter scenarios presented in Chapter 5, the variations in biological parameters display clear relationships across time, with the value of r being negatively correlated with K and positively correlated with q.

[21] The relationships are detailed in Section 6.4.1 of Chapter 6.

parameter values.[22] Moreover, results generated by the dynamic bioeconomic model reveal that variable biological parameter values have a muted impact upon welfare levels.[23] More significant, however, is the result that, irrespective of the values assumed for the biological parameters, the increase in rents achieved by shifting the fisheries from the dynamic maximum sustainable yield outcome to the dynamic maximum economic yield outcome is considerable. In the case of the west coast fishery, the average increase in rents is 53.17 percent. In the south coast fishery, the impact upon rents is more dramatic, with rents increasing by an average 1 022.43 percent. These results are summarised in Table 8.3.

Table 8.3: The Impact of Variable Biological Parameter Values on Welfare Levels under Dynamic Maximum Economic Yield and Dynamic Maximum Sustainable Yield in the West and South Coast Hake Fisheries

West Coast Rent (Index Points)			South Coast Rent (Index Points)	
Maximum Sustainable Yield	Maximum Economic Yield	Parameter Scenario	Maximum Sustainable Yield	Maximum Economic Yield
17 891.14	11 789.04	Range	2 116.88	1 538.48
42 712.11	65 424.09	Mean	1381.24	15 503.52
9 620.70	7 148.62	Standard Deviation	1 145.99	794.53
1.86	1.65	Range/Standard Deviation	1.85	1.94
22.52	10.93	Coefficient of Variation	82.97	5.12

Source: See Tables 6.1, 6.3 and 6.4 of Chapter 6.

Given the insensitivity of the model estimates to variations in paramete values, the remainder of this study is based on the assumption that biological parameter values are given by the mean value of parameters, as set out in Table 5.2 of Chapter 5. The reason for this is readily apparent: assuming constant values for biological parameters considerably reduces the number of

[22] These results are detailed in Table 6.3 of Chapter 6.
[23] Detailed evidence of this is presented in Table 6.4 of Chapter 6.

possible outcomes (in this case by a factor of five) without violating the integrity of the model or its results.

The analysis of the impact of variable economic parameter values on the dynamic rent maximising outcome provides for two main results.[24] First, it is readily apparent from the model presented above (Equation 6.1) that relationships exist between each of the economic parameters and the dynamic rent maximising biomass and, therefore, with the dynamic rent maximising catch and effort levels.[25] However, assuming a range of values for the various economic parameters based on the extent of historical fluctuations, it is possible to show that the estimates of dynamic rent maximising (as well as dynamic yield maximising) biomass, catch and effort levels produced under the variable parameter model for both the west and south coast fisheries are highly insensitive to fluctuations in economic parameters values.[26] Specifically, the evidence presented in Chapter 6 shows that estimates of dynamic rent maximising biomass, catch and effort fluctuate by less than one-third the level of variation allowed for in economic parameter values. Second, while variations in economic parameter values impact upon estimates of the size of economic rents generated under various scenarios, the extent of the impact is limited.[27] More significant, however, is the result that the estimates of rent levels under the dynamic maximum economic yield strategy are, without exception, considerably higher than those generated under the dynamic maximum sustainable yield strategy. Rents generated by the dynamic maximum economic yield estimates are, on average, 44.58 percent and 862.99 percent greater than those generated under the dynamic maximum economic yield scenario for the west and south coast fisheries respectively. These results are summarised in Table 8.4.

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[24] In contrast to the sensitivity analysis conducted with regard to biological parameter values, sensitivity of the dynamic bioeconomic model to economic parameters is conducted on a parameter-by-parameter basis. The reason for this is that, unlike the case of biological parameters, there are no obvious reasons for expecting relationships to exist between the various economic parameters. This is supported by anecdotal evidence with respect to the relationship (or rather absence thereof) between economic parameter values across time.

[25] The relationships are detailed in Section 6.5.1 of Chapter 6.

[26] Detailed evidence of this is presented in Tables 6.5-6.10 of Chapter 6.

[27] Detailed evidence is presented in Table 6.11 of Chapter 6.

The results of the sensitivity analysis undertaken in Chapter 6 are significant Specifically, in spite of biological and economic parameter fluctuations, the management implications of the dynamic bioeconomic model remain unaltered. Thus, while variations in economic and biological parameter values impact on biomass, catch and effort levels, as well as economic rent levels, these output values are relatively insensitive to changes in input values. More significantly, the rents generated under the dynamic maximum economic yield strategy are, without exception, substantially greater than those generated under the dynamic maximum sustainable yield strategy. Thus, the dynamic bioeconomic model developed and applied in Chapters 3 – 6 is capable of providing fisheries managers with a high degree of confidence in the targets that are set as the basis for achieving gains in economic rents. In short, the result provides fisheries managers with an unambiguous optimal harvest strategy in the management of the South African hake fishery: dynamic maximum economic yield.

Table 8.4: A Comparison of the Mean Values of Economic Rents Generated under the Dynamic Maximum Economic Yield and Dynamic Maximum Sustainable Yield Strategies under the Assumption of Variable Economic Parameter Values

Variable Parameter Scenario	Dynamic Maximum Economic Yield ($X^{\bullet}_{MEY}$) (1)	Dynamic Maximum Sustainable Yield ($X^{\bullet}_{MSY}$) (2)	Welfare Gain (1) versus (2) (Percent)
West Coast Hake Fishery			
	Index Points		
Price	70 787.43	48 724.30	45.28
Cost	70 773.43	48 724.30	45.26
Quasi-Discount Rate	70 490.56	49 223.90	43.20
South Coast Hake Fishery			
	Index Points		
Price	15 978.59	1 651.24	867.67
Cost	15 965.63	1 651.24	866.89
Quasi-Discount Rate	15 692.77	1 644.22	854.42

See Table 6.11 of Chapter 6.

Before considering the management implications of the above result, it is noted in Chapter 7 that, given that the west and south coast hake fisheries are managed on the basis of an $f_{0.2}$ principle, it follows that both fisheries are in positions other than the dynamic maximum sustainable yield outcome. Indeed, this has been the case over the entire sample period (1984-95). Therefore, gains in rent to be achieved by shifting the fishery to the dynamic rent maximising position are different to those modelled earlier on the basis of the assumption that the fishery is initially in a yield maximising position. None the less, gains to be had from shifting the fisheries from their extant positions to the dynamic rent maximising outcome remain considerable. Table 8.5 provides a summary of the biomass, catch and effort data for two scenarios: the mean of the modelled dynamic maximum economic yield outcome; and an assumed extant position given by the observed figures for 1995, as generated under the $f_{0.2}$ harvesting strategy. The economic rent values associated with each of the scenarios are also shown.

Table 8.5: Biomass, Catch and Effort Figures under Modelled and Observed Scenarios

	Modelled	Observed
	Dynamic MEY (X^{*}_{MEY})	1995
West Coast Fishery		
Biomass (tons)	878 765	778 000
Catch (tons)	125 714	96 600
Effort (standard-boat-days)	10 306	14 500
Rent (index points)	65 423.90	11 775.00
South Coast Fishery		
Biomass (tons)	205 169	165 000
Catch (tons)	42 709	45 800
Effort (standard-boat-hours)	52 319	70 500
Rent (index points)	15 503.12	9 140.00

Source: See Table 7.1 of Chapter 7.

The results of the modelling procedure, summarised in Table 8.5, suggest that shifting the west coast fishery to the dynamic maximum economic yield position would produce a 30.14 percent gain in catch on the back of a 28.92 percent reduction in effort from the extant level. This, in turn, would produce a more than five-fold increase in the rent index from the 1995 level.[28] Shifting the south coast fishery from the 1995 position to the dynamic maximum economic yield position would produce a marginal fall in catch against a 25.79 percent reduction in effort from extant levels.[29] The composite effect would be a rise in economic rent of approximately 70 percent above the extant level.

The above set of findings allows for two main results. First, the gains in rent achieved by shifting the hake fishery from the extant (1995) position to the dynamic maximum sustainable yield position are considerable. Second, irrespective of the relative size of rent increases, the primary result holds: welfare maximisation, by definition and by the above results, demands shifting the hake fisheries to the dynamic rent maximising position. Based on this conclusion, the final portion of this study is devoted to reviewing the various instruments available to fisheries managers as a basis for providing some guide as to how the goal of welfare maximisation may be best achieved and maintained.

Fisheries management potentially encompasses a host of policy goals. This study adopts three main goals in the formulation of policy.[30] In line with the theory of renewable resource economics, the policy objective of economic efficiency – that is, dynamic rent maximisation – is set as the principal policy objective. To this, two further goals are added: equity and stability. The adoption of the equity criterion is obvious given the gross inequalities that pervade the South African fishing industry. The addition of the stability criterion is due to the substantial immobility of fishing industry resources, which renders rapid industrial contraction (or expansion) highly disruptive to

[28] This is considerably greater than the 23.51 percent increase that is shown in Chapters 5 and 6 to be achievable by shifting the west coast hake fishery from a 'hypothetical' dynamic maximum sustainable yield position to the dynamic rent maximising position.
[29] The increase in rent achieved by shifting the south coast hake fishery from the extant (1995) position to the dynamic rent maximising position is considerably less than the 1 022.41 percent increase in rent shown in Chapters 5 and 6 to be achievable by shifting the fishery from the 'hypothetical' dynamic maximum sustainable yield position to the dynamic maximum economic yield position.
[30] Section 7.3 of Chapter 7 provides a review of potential policy goals.

factor and product markets. As noted in Section 7.3 of Chapter 7, however, it is not unreasonable to expect policy goals to come into conflict. Where conflict arises, it is suggested that trade-offs must be made under the guiding principle of 'reasonableness'. Of course, this is a matter of judgement.

Given the policy objectives of efficiency, equity and stability, the problem then becomes one of establishing the form fisheries management policy should take. One of the first issues to explore is whether the fishery should be regulated at all. As shown elsewhere in this study, there is sufficient evidence to show that, in the absence of regulations, over-exploitation of fish stocks – the so-called Class I problem of open-access – is abundant.[31] Thus, regulation of the fishery, as the basis for securing the dynamic rent maximising outcome, is accepted as a first principle in fisheries management. The problem then becomes one of establishing the set of regulatory tools that best provide for achieving the stated policy goals.

At the broadest level, regulatory instruments can be divided into two distinct groups: non-market or direct controls; and market-based or indirect controls. Historically, fisheries have been regulated by way of non-market controls. However, while there are examples of the successful application of non-market regulations, these cases are heavily outweighed by the failure of non-market controls to prevent rent dissipation.[32] The failure of non-market regulations can typically be ascribed to two main problems. First, non-market controls are void of economic efficiency considerations. Second, non-exclusive controls often result in what is widely known as the 'Class II common property problem' – or, more correctly, the Class II problem of open-access – which refers to the situation where fish stocks are preserved but rents are dissipated due to the effects of 'crowding' and 'capital stuffing' (that is, over-investment by fishers in vessels and gear). Furthermore, there is a considerable body of evidence to suggest that direct controls are unable to

[31] See, in particular, the evidence presented in Chapters 1, 2 and 7.
[32] See Section 7.3 of Chapter 7 for a review of the evidence.

prevent biological over-exploitation of the fish stock as fishers find loopholes in regulations.[33]

Consequently, the emphasis amongst fisheries managers has, more recently, tended toward the use of market-based controls which typically take one of two forms. The first involves the imposition of taxes on inputs or outputs. There is a rich body of theory which demonstrates that, by adjusting market signals (costs and prices), such taxes will lead to an optimal level of exploitation of the resource. However, the use of taxation as a regulatory tool runs into a number of problems in practice, including difficulties in establishing optimal tax rates, political unpopularity and the relative cost ineffectiveness of taxation as a regulatory tool. Most damning of the feasibility of taxation as a regulatory instrument, however, is the fact that there are no examples of the successful use of taxation in fisheries management.[34]

Thus, regulatory authorities have tended toward the use of the second main form of market-based control, namely, individual transferable quotas (ITQs), as the basis for fisheries management. ITQs are based on the argument that the Class I and Class II problems of renewable resource use stem from the inability of parties to contract in the absence of private property rights. Accordingly, the solution is argued to lie not in attempting to correct market signals, but rather in the creation of private property rights. Private property rights for fishers are established through saleable harvest quotas, commonly known as ITQs. The purported advantages of management by ITQs lie in the elimination of important external diseconomies through fishers internalising the external costs of harvesting associated with both open-access and limited entry licensing. That is, ITQs are argued to be capable of solving the Class I and Class II problems of open-access.[35]

In addition to facilitating gains in economic efficiency, ITQs have a number of patent advantages over other regulatory tools. The direct control of catches

[33] Some of the available evidence is reviewed in Section 7.3 of Chapter 7.
[34] See Section 7.3.2 of Chapter 7.
[35] These arguments are detailed in Section 7.3 of Chapter 7.

reduces the need for complex restrictions on effort. Furthermore, given that the quota constitutes a transferable asset with an intrinsic value, it follows that ITQs are capable of providing fishers with greater financial security. Perhaps most important, however, is that, unlike other forms of (direct and indirect) regulation, there are numerous examples of the successful use of ITQs in practice. Indeed, in view of the theoretical superiority of the ITQ system, operators in the South African hake trawl fishery have argued strongly in favour of the adoption of ITQs. However, while the theoretical advantages of ITQs are widely and generally accepted, the implementation of ITQs presents problems in practice.

Broadly speaking, problems with the implementation of ITQs can be grouped into six main areas: (i) first-round distribution of ITQs; (ii) the use of percentage-based or volume-based quotas; (iii) duration of rights under the quota system; (iv) transferability of ITQs and implications for the industrial structure of the fishery; (v) the optimal approach path and implications of alternative technologies for the industrial structure of the fishery; and (vi) ensuring an efficient and effective management approach.[36]

It is argued in Chapter 7 that through the adoption of an appropriately structured ITQ system, it is possible for regulatory authorities to overcome these problems whilst, simultaneously, providing a management framework capable of achieving the goals of efficiency, equity and stability set up as the basis for fisheries management. Specifically, the arguments presented in Chapter 7 suggest that ITQs adopted as the basis for regulating the South African hake fishery should be freely transferable, perfectly divisible, volume-based ITQs, allocated in perpetuity on the basis of 'grandfathering' principles coupled with an initial purchase price and a 'low but effective' annual tax. Furthermore, the proposed ITQ should embrace the principles of 'phased' adjustment to ensure minimisation of inefficiency through forced retirement of labour or capital; facilitate efficient and effective enforcement by 'monitoring'

[36] These problems are reviewed in greater detail in Section 7.4 of Chapter 7.

rather than 'policing' fishing activity; and allow for maximum gains in efficiency through the transferability of quotas across technologies.

In addition, it is proposed that the capital pool and revenue flows generated by this arrangement could serve as the basis for meeting various policy goals, including the funding of research and management, and the promotion of equity through the creation of a development corporation. Furthermore, the adoption of a volume-based system may require the regulatory authorities to appoint 'market-makers' responsible for generating market liquidity, that is, promoting market efficiency. It is also suggested that a Code of Conduct be adopted as the basis for regulating industry behaviour in the face of potential threats of the abuse of fishing rights under a perpetual rights system. The proposed Code of Conduct should also serve as the basis for facilitating improved competitiveness and efficiency through, for example, promotion of the small business sector, and the setting of guidelines for the transfer of ITQs between fishers.

In short, it is concluded that bioeconomic management, through the use of appropriate market-based regulatory tools, has the capacity to restore and preserve the integrity of the hake fishery. More importantly, it is concluded that an appropriately designed market-based ITQ system, based on the policy goals of equity, stability and efficiency, has the capacity to ensure that the welfare and socio-economic contributions made by the hake fishery to the South African economy are maximised on a sustainable basis.

While this study constitutes a first attempt to introduce economic principles into the modelling and management of biologically renewable resources in South Africa, it is limited in a number of ways. As already noted, the model developed and applied in Chapters 3-6 does not take uncertainty into account (although it can be said that the adoption of confidence intervals and range values based on 'random' variations in parameter values does, in a limited manner, cater for uncertainty). Moreover, the model does not attempt to incorporate the issue of multi-species interaction. Further work is required to address these shortcomings. In addition, the policy implications presented in Chapter 7 suggest a number of further areas for research. The most significant of these – each of which constitutes the basis for an extensive investigation in its own right – are discussed briefly below.

To start with, while a 'phased' approach path is argued to be optimal in the presence of factor market rigidities, there is a need for a more detailed examination of the nature of the optimal approach path. Specific details of the nature of the approach path aside, it is unlikely that the structure of the industry will remain unaffected. Levels of employment of capital and labour within sectors, and the proportions in which resources are spread across sectors, are likely to change as the fishery is driven from the extant position to the dynamic rent maximising outcome.[37] The relative significance of the various geographic components of the fishery – that is, the west versus south and deep-sea versus inshore – are also likely to alter as the fishery is driven from its extant to dynamic rent maximising position.[38] Further work is required in this regard.

Furthermore, the analysis undertaken in this study is based on the assumption that fishers continue to practice extant harvesting methods. However, in its purest form, the ITQ system places no restriction on the use of alternative technologies (subject, of course, to the satisfaction of various biological criteria). In the context of the South African hake fishery, the most obvious change would involve shifting hake-directed effort from trawling to longlining. The implications of such a switch are potentially dramatic given that barriers to entry in the longline fishery are relatively low; operating profit margins are relatively high; and longlining is substantially more labour-intensive than trawling for hake. In a similar vein, this study does not consider the issue of optimal fleet composition or the optimal harvest strategy (that is,

[37] See Section 7.4.5.1 for a preliminary analysis of the likely implications of bioeconomic management for factor usage.
[38] See Section 7.4.5.1 for a preliminary analysis of the likely implications of bioeconomic management for the sectoral composition of the hake fishery.

pulse versus continuous fishing). Each of these issues demands further investigation.

In addition, successful fisheries management hinges upon its effectiveness and efficiency. In this regard, it is suggested in Chapter 7 that, because of the presence of 'choke points' and the absence of 'high stock variability', the hake fishery escapes a number of problems typically encountered in regulation. However, two problems that are likely to arise under ITQs, and which have the potential to undermine the effectiveness of management, are 'high-grading' and 'by-catch' as well as the presence of multiple processing forms. 'High-grading' and 'by-catch' have the potential to undermine the effectiveness of controls through biological over-exploitation, whereas a 'sub-optimal' mix of multi-processing forms has the potential to undermine controls by introducing economic inefficiencies. While ITQs remain optimal in the presence of these problems, further work is required to address these complications if rents generated by the fishery are to be preserved. Related to this, further work needs to be done on the design and implementation of a deregulated, 'monitoring-based' enforcement programme capable of ensuring efficient and effective regulation of the South African hake fishery.

It is recognised that this study is by no means definitive or complete. Further extensions and refinements to the bioeconomic modelling and management of the hake fishery are required. Moreover, similar exercises need to be carried out with respect to other renewable resources. None the less, it is believed that this study makes a contribution to the application of the theory of resource economics in South Africa. As such, it provides a starting point in recognising bioeconomic principles as the basis for achieving social welfare maximisation through the optimal use and effective and efficient management of renewable resources in South Africa within the context of sustainable economic growth and development.

APPENDIX 1: THE DYNAMIC BIOECONOMIC MODEL

The solution to the dynamic Gordon model is given here. The problem is:

$$Maximise \int_0^{\infty} (pqX_t - c)E_t e^{-\delta t} dt\,, \qquad \text{(Equation A1)}$$

subject to the conditions:

$$\frac{dX_t}{dt} = G(X_t) - qE_tX_t\,; \qquad \text{(Equation A2)}$$

$0 \le E_t \le E_{MAX}$; and

$X_t \ge 0$.

Also, the initial biomass X_0 is assumed known.

First, by solving for E_t in Equation A2 and substituting into the objective integral (Equation A1), Equation A1 then becomes:

$$\int_0^{\infty} e^{-\delta t}\left[p - c(X_t)\right]\left[G(X_t) - \frac{dX_t}{dt}\right]dt \qquad \text{(Equation A3)}$$

where $c(X_t) = \dfrac{c}{qX_t}$. Then, define the function $Z(X_t)$ as follows:

$$Z(X_t) = \int_{\overline{X}_t}^{X_t}\left[p - c(u)\right]du \qquad \text{(Equation A4)}$$

where $\overline{X}_t = \dfrac{c}{pq}$. Note that for $Z_t = Z(X_t)$ we have:

$$\frac{dZ_t}{dt} = [p - c(X_t)]\frac{dX_t}{dt}. \qquad \text{(Equation A5)}$$

Integration by parts therefore yields:

$$\int_0^{\infty} e^{-\delta t}[p - c(X_t)]\frac{dX_t}{dt}dt = \int_0^{\infty} e^{-\delta t}\frac{dZ_t}{dt}dt = e^{-\delta t}Z_t\Big|_0^{\infty} + \delta\int_0^{\infty} e^{-\delta t}Z_t dt$$

$$= Z(X_0) + \delta \int_0^\infty e^{-\delta t} Z_t dt .$$

Because the term $Z(X_0)$ can be ignored, the integral now becomes:

$$\int_0^\infty e^{-\delta t} V(X_t) dt \qquad \text{(Equation A6)}$$

with:

$$V(X_t) = [p - c(X_t)][G(X_t) - \delta X_t]. \qquad \text{(Equation A7)}$$

But obviously Equation A6 is maximised if $X_t = X^*_{MEY}$, where X^*_{MEY} maximises the expression (X_t) (Equation A7), and the biomass X_t should be adjusted from X_0 to X^*_{MEY} (by fishing at $E_t = 0$ or E_{MAX} as required) as rapidly as possible; this is the so-called 'bang-bang' approach.

A straightforward calculation shows that the necessary condition $V'(X^*_{MEY}) = 0$ can be expressed in the form:

$$G'(X^*_{MEY}) - \frac{c'(X^*_{MEY})G(X^*_{MEY})}{p - c(X^*_{MEY})} = \delta \qquad \text{(Equation A8)}$$

which is the desired result.

(It should be noted here that the above argument, to the effect that X_t should be adjusted by fishing to X^*_{MEY}, tacitly assumes that the function $V(X_t)$ is monodromic, that is, that $V(X_t)$ has a unique local maximum on $\bar{X}_t \leq X_t \leq K$. It also assumes that E_{MAX} is sufficiently large that X_t can be adjusted to and kept at X^*_{MEY}. Strange – and quite interesting – things happen if either of these assumptions is violated.)[1]

Finally, X^*_{MEY} is a decreasing function of δ for the dynamic Gordon model presented here, in which:

[1] See Spence and Starrett (1975) and Majumdar and Mitra (1983).

$$G(X_t) = rK\left(1 - \frac{X_t}{K}\right) \text{ and } c(X_t) = \frac{c}{qX_t}.$$

In this case the left side of Equation A8 becomes:

$$r\left(1 - \frac{2X_t}{K}\right) + \frac{cr(1 - X_t / K)}{pqX_t - c}.$$

This is obviously a decreasing function of X_t for $X_t > \overline{X}_t = c / pq$, and its value approaches $+\infty$ as $X_t \to \overline{X}_t$ from above. It therefore follows from Equation A8 that X^*_{MEY} is a decreasing function of δ and also that $X^*_{MEY} \to \overline{X}_t$ as $\delta \to +\infty$.

APPENDIX 2: DYNAMIC BIOECONOMIC MODEL ESTIMATES FOR THE WEST AND SOUTH COAST HAKE FISHERIES

DYNAMIC BIOECONOMIC MODEL OF THE WEST COAST HAKE FISHERY

MODEL INPUTS: PARAMETER ESTIMATES BASED ON LOWER CONFIDENCE INTERVAL VALUES

Intrinsic growth rate	r	0.2577
Carrying capacity	K	1 803 148
Catchability coefficient	q	1.04E-05
Cost	c	5.85
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	901 574	1 182 824	116 168	104 863	12 389	8 524
3.00	796 618	1 130 109	114 593	108 704	13 832	9 249
4.50	744 140	1 105 496	112 626	110 225	14 553	9 587
6.00	691 662	1 082 035	109 870	111 514	15 274	9 910

MODEL INPUTS: PARAMETER ESTIMATES BASED ON MINIMUMS OF OBSERVED VALUES

Intrinsic growth rate	r	0.3092
Carrying capacity	K	1 678 000
Catchability coefficient	q	1.19E-05
Cost	c	5.85
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	839 000	1 084 798	129 709	118 577	12 992	9 186
3.00	757 596	1 041 807	128 488	122 130	14 252	9 851
4.50	716 895	1 021 458	126 962	123 575	14 882	10 166
6.00	676 193	1 001 877	124 825	124 821	15 513	10 470

MODEL INPUTS: PARAMETER ESTIMATES BASED ON MEANS OF OBSERVED VALUES

Intrinsic growth rate	r	0.3905
Carrying capacity	K	1 424 167
Catchability coefficient	q	1.44E-05
Cost	c	5.85
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS (OBSERVED MEAN CONFIDENCE INTERVAL)

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	712 084	915 209	139 034	127 721	13 559	9 691
3.00	657 378	885 598	138 214	130 779	14 601	10 255
4.50	630 025	871 406	137 188	132 074	15 122	10 525
6.00	602 672	857 625	135 752	133 226	15 642	10 788

MODEL INPUTS: PARAMETER ESTIMATES BASED ON MAXIMUMS OF OBSERVED VALUES

Intrinsic growth rate	r	0.4806
Carrying capacity	K	1 192 000
Catchability coefficient	q	1.71E-05
Cost	c	5.85
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	596 000	767 053	143 219	131 422	14 053	10 020
3.00	558 797	746 890	142 661	134 039	14 930	10 495
4.50	540 195	737 146	141 963	135 186	15 368	10 725
6.00	521 593	727 629	140 987	136 233	15 807	10 949

MODEL INPUTS: PARAMETER ESTIMATES BASED ON UPPER CONFIDENCE INTERVAL VALUES

Intrinsic growth rate	r	0.5233
Carrying capacity	K	1 045 186
Catchability coefficient	q	1.84E-05
Cost	c	5.85
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	522 593	681 560	136 736	124 084	14 220	9 894
3.00	492 634	665 905	136 287	126 453	15 035	10 320
4.50	477 654	658 324	135 725	127 512	15 443	10 527
6.00	462 674	650 909	134 939	128 493	15 851	10 729

MODEL INPUTS: PARAMETER ESTIMATES BASED ON LOWER CONFIDENCE INTERVAL VALUES

Intrinsic growth rate	r	0.6545
Carrying capacity	K	310 824
Catchability coefficient	q	3.60E-06
Cost	c	0.52
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	155 412	227 634	50 859	39 875	90 903	48 659
3.00	148 288	225 082	50 752	40 638	95 069	50 152
4.50	144 727	223 844	50 618	40 998	97 153	50 876
6.00	141 165	222 631	50 431	41 344	99 236	51 586

MODEL INPUTS: PARAMETER ESTIMATES BASED ON MINIMUMS OF OBSERVED VALUES

Intrinsic growth rate	r	0.6850
Carrying capacity	K	302 500
Catchability coefficient	q	3.70E-06
Cost	c	0.52
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	151 250	221 520	51 803	40 621	92 568	49 561
3.00	144 626	219 144	51 704	41 365	96 622	51 015
4.50	141 314	217 990	51 580	41 717	98 649	51 722
6.00	138 002	216 858	51 406	42 056	100 676	52 414

MODEL INPUTS: PARAMETER ESTIMATES BASED ON MEANS OF OBSERVED VALUES

Intrinsic growth rate	r	0.7405
Carrying capacity	K	285 775
Catchability coefficient	q	4.00E-06
Cost	c	0.52
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	142 888	207 888	52 904	41 956	92 563	50 456
3.00	137 099	205 756	52 817	42 662	96 313	51 836
4.50	134 204	204 718	52 709	42 998	98 188	52 509
6.00	131 310	203 699	52 557	43 322	100 063	53 169

MODEL INPUTS: PARAMETER ESTIMATES BASED ON MAXIMUMS OF OBSERVED VALUES

Intrinsic growth rate	r	0.8003
Carrying capacity	K	269 600
Catchability coefficient	q	4.30E-06
Cost	c	0.52
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	134 800	195 265	53 940	43 087	93 058	51 317
3.00	129 747	193 372	53 864	43 756	96 547	52 623
4.50	127 220	192 448	53 770	44 075	98 291	53 261
6.00	124 694	191 540	53 637	44 384	100 035	53 888

MODEL INPUTS: PARAMETER ESTIMATES BASED ON UPPER CONFIDENCE INTERVAL VALUES

Intrinsic growth rate	r	0.8265
Carrying capacity	K	260 726
Catchability coefficient	q	4.40E-06
Cost	c	0.52
Price	p	1.00
Discount rate	δ	As per scenario

MODEL OUTPUTS: OPTIMAL BIOMASS, CATCH AND EFFORT

	Biomass		Catch		Effort	
δ	MSY	MEY	MSY	MEY	MSY	MEY
0.00	130 363	189 454	53 873	42 804	93 920	51 348
3.00	125 631	187 701	53 802	43 451	97 330	52 611
4.50	123 265	186 846	53 713	43 759	99 034	53 227
6.00	120 899	186 003	53 589	44 059	100 739	53 834

APPENDIX 3: WELFARE IMPLICATIONS UNDER THE DYNAMIC BIOECONOMIC MODEL

WELFARE IMPLICATIONS UNDER THE DYNAMIC BIOECONOMIC MODEL

WEST COAST HAKE FISHERY

Lower Confidence Limit

r	0.5233
K	1 045 186
p	120.00
c	100.00
q	1.84E-05
δ	4.50

	MSY	MEY
Biomass	477 654	658 324
Catch	135 724	127 512
Effort	15 443	10 527
Revenue	135 724	127 512
Cost	90 342	61 583
Rent	45 382	65 929
Net welfare change (percent)		45.27

SOUTH COAST HAKE FISHERY

Lower Confidence Limit

r	0.8265
K	260 726
p	120.00
c	100.00
q	4.40E-06
δ	4.50

	MSY	MEY
Biomass	123 265	186 846
Catch	53 713	43 759
Effort	99 034	53 227
Revenue	53 713	43 759
Cost	51 498	27 678
Rent	2 215	16 081
Net welfare change (percent)		625.90

TOTAL WELFARE IMPLICATIONS

		MSY	MEY
Rent:	West Coast Fishery	45 382	65 929
	South Coast Fishery	2 215	16 081
Total Rent		47 598	82 010
Net welfare gain (percent)			72.30

WELFARE IMPLICATIONS UNDER THE DYNAMIC BIOECONOMIC MODEL

WEST COAST HAKE FISHERY

Minimum Observed Values

r	0.4806
K	1 192 000
p	120.00
c	100.00
q	1.71E-05
δ	4.50

	MSY	MEY
Biomass	540 195	737 146
Catch	141 963	135 186
Effort	15 368	10 725
Revenue	141 963	135 186
Cost	89 903	62 741
Rent	52 060	72 445
Net welfare change (percent)		39.16

SOUTH COAST HAKE FISHERY

Minimum Observed Values

r	0.8003
K	269 600
p	120.00
c	100.00
q	4.30E-06
δ	4.50

	MSY	MEY
Biomass	127 220	192 448
Catch	53 770	44 075
Effort	98 291	53 261
Revenue	53 770	44 075
Cost	51 111	27 696
Rent	2 659	16 379
Net welfare change (percent)		516.07

TOTAL WELFARE IMPLICATIONS

		MSY	MEY
Rent:	West Coast Fishery	52 060	72 445
	South Coast Fishery	2 659	16 379
Total Rent		54 719	88 824
Net welfare gain (percent)			62.33

WEST COAST HAKE FISHERY

Mean Observed Values

r	0.3905
K	1 424 167
p	120.00
c	100.00
q	1.44E-05
δ	4.50

	MSY	MEY
Biomass	630 025	871 406
Catch	137 188	132 074
Effort	15 122	10 525
Revenue	137 188	132 074
Cost	88 464	61 571
Rent	48 724	70 503
Net welfare change (percent)		44.70

SOUTH COAST HAKE FISHERY

Mean Observed Values

r	0.7405
K	285 775
p	120.00
c	100.00
q	4.00E-06
δ	4.50

	MSY	MEY
Biomass	134 204	204 718
Catch	52 709	42 998
Effort	98 188	52 509
Revenue	52 709	42 998
Cost	51 058	27 305
Rent	1 651	15 693
Net welfare change (percent)		850.40

TOTAL WELFARE IMPLICATIONS

		MSY	MEY
Rent:	West Coast Fishery	48 724	70 503
	South Coast Fishery	1 651	15 693
Total Rent		50 376	86 196
Net welfare gain (percent)			71.11

WELFARE IMPLICATIONS UNDER THE DYNAMIC BIOECONOMIC MODEL

WEST COAST HAKE FISHERY

Maximum Observed Values

r	0.3092
K	1 678 000
p	120.00
c	100.00
q	1.19E-05
δ	4.50

	MSY	MEY
Biomass	716 895	1 021 458
Catch	126 962	123 575
Effort	14 882	10 166
Revenue	126 962	123 575
Cost	87 060	59 471
Rent	39 902	64 104
Net welfare change (percent)		60.65

SOUTH COAST HAKE FISHERY

Maximum Observed Values

r	0.6850
K	302 500
p	120.00
c	100.00
q	3.70E-06
δ	4.50

	MSY	MEY
Biomass	141 314	217 990
Catch	51 580	41 717
Effort	98 649	51 722
Revenue	51 580	41 717
Cost	51 297	26 895
Rent	283	14 822
Net welfare change (percent)		5 146.20

TOTAL WELFARE IMPLICATIONS

		MSY	MEY
Rent:	West Coast Fishery	39 902	64 104
	South Coast Fishery	283	14 822
Total Rent		40 185	78 925
Net welfare gain (percent)			96.41

WELFARE IMPLICATIONS UNDER THE DYNAMIC BIOECONOMIC MODEL

WEST COAST HAKE FISHERY

Upper Confidence Limit

r	0.2577
K	1 803 148
p	1.00
c	5.85
q	1.04E-05
δ	4.50

	MSY	MEY
Biomass	744 140	1 105 493
Catch	112 626	110 225
Effort	14 553	9 587
Revenue	112 626	110 225
Cost	85 135	56 085
Rent	27 491	54 140
Net welfare change (percent)		96.93

SOUTH COAST HAKE FISHERY

Upper Confidence Limit

r	0.6545
K	310 824
p	1.00
c	0.52
q	3.60E-06
δ	4.50

	MSY	MEY
Biomass	144 727	223 844
Catch	50 618	40 998
Effort	97 153	50 876
Revenue	50 618	40 998
Cost	50 520	26 456
Rent	98	14 542
Net welfare change (percent)		14 672.94

TOTAL WELFARE IMPLICATIONS

		MSY	MEY
Rent:	West Coast Fishery	27 491	54 140
	South Coast Fishery	98	14 542
Total Rent		27 590	68 682
Net welfare gain (percent)			148.94

APPENDIX 4: THE IMPACT OF VARIABLE ECONOMIC PARAMETERS ON BIOMASS, CATCH, EFFORT AND RENTS IN THE HAKE FISHERIES

WEST COAST HAKE FISHERY: VARIABLE PRICE SCENARIO

	Dynamic Maximum Economic Yield					Dynamic Maximum Sustainable Yield					
Price	Catch	Effort	Revenue	Cost	Rent	Catch	Effort	Revenue	Cost	Rent	Gain (%)
0.75	123 805	9 071	92 854	53 065	39 788	137 188	15 122	102 891	88 464	14 427	175.79
0.80	126 157	9 433	100 926	55 183	45 743	137 188	15 122	109 750	88 464	21 287	114.89
0.85	128 076	9 752	108 865	57 049	51 815	137 188	15 122	116 610	88 464	28 146	84.09
0.90	129 657	10 038	116 691	58 722	57 969	137 188	15 122	123 469	88 464	35 006	65.60
0.95	130 972	10 294	124 423	60 220	64 204	137 188	15 122	130 329	88 464	41 865	53.36
1.00	132 074	10 525	132 074	61 571	70 503	137 188	15 122	137 188	88 464	48 724	44.70
1.05	133 004	10 735	139 654	62 800	76 854	137 188	15 122	144 047	88 464	55 584	38.27
1.10	133 794	10 927	147 173	63 923	83 250	137 188	15 122	150 907	88 464	62 443	33.32
1.15	134 469	11 102	154 639	64 947	89 693	137 188	15 122	157 766	88 464	69 303	29.42
1.20	135 048	11 263	162 058	65 889	96 169	137 188	15 122	164 626	88 464	76 162	26.27
1.25	135 547	11 412	169 434	66 760	102 674	137 188	15 122	171 485	88 464	83 021	23.67

0.50	Range	62885.15	68594.00	152.11
1.00	Mean	70787.43	48724.30	62.67
0.17	Standard Error	6300.33	6859.40	14.08
16.58	Coefficient of Variation (%)	8.90	14.08	22.47

WEST COAST HAKE FISHERY: VARIABLE COST SCENARIO

	Dynamic Maximum Economic Yield					Dynamic Maximum Sustainable Yield					
Cost	Catch	Effort	Revenue	Cost	Rent	Catch	Effort	Revenue	Cost	Rent	Gain (%)
4.39	136 235	11 635	136 235	51 049	85 186	137 188	15 122	137 188	66 348	70 840	20.25
4.68	135 547	11 412	135 547	53 408	82 139	137 188	15 122	137 188	70 771	66 417	23.67
4.97	134 786	11 189	134 786	55 637	79 149	137 188	15 122	137 188	75 194	61 994	27.67
5.27	133 953	10 967	133 953	57 741	76 212	137 188	15 122	137 188	79 617	57 571	32.38
5.56	133 049	10 746	133 049	59 721	73 328	137 188	15 122	137 188	84 041	53 147	37.97
5.85	132 074	10 525	132 074	61 571	70 503	137 188	15 122	137 188	88 464	48 724	44.70
6.14	131 029	10 306	131 029	63 305	67 724	137 188	15 122	137 188	92 887	44 301	52.87
6.44	129 914	10 086	129 914	64 903	65 011	137 188	15 122	137 188	97 310	39 878	63.02
6.73	128 730	9 868	128 730	66 387	62 343	137 188	15 122	137 188	101 733	35 455	75.84
7.02	127 478	9 650	127 478	67 743	59 735	137 188	15 122	137 188	106 156	31 032	92.50
7.31	126 157	9 433	126 157	68 979	57 178	137 188	15 122	137 188	110 580	26 608	114.89

2.93	Range	28008.25	44231.85	94.64
5.85	Mean	70773.43	48724.30	53.25
0.97	Standard Error	2801.76	4423.19	9.20
16.58	Coefficient of Variation (%)	3.96	9.08	17.27

WEST COAST HAKE FISHERY: VARIABLE QUASI-DISCOUNT RATE SCENARIO

		Dynamic Maximum Economic Yield					Dynamic Maximum Sustainable Yield					
Discount Pατε		Catch	Effort	Revenue	Cost	Rent	Catch	Effort	Revenue	Cost	Rent	Gain (%)
3.38		131 117	10 323	131 117	60 390	70 727	137 996	14 731	137 996	86 176	51 820	36.49
3.60		131 315	10 364	131 315	60 629	70 686	137 853	14 809	137 853	86 633	51 220	38.00
3.83		131 510	10 405	131 510	60 869	70 641	137 700	14 887	137 700	87 089	50 611	39.58
4.05		131 701	10 445	131 701	61 103	70 598	137 539	14 965	137 539	87 545	49 994	41.21
4.28		131 889	10 485	131 889	61 337	70 552	137 368	15 043	137 368	88 002	49 366	42.91
4.50		132 074	10 525	132 074	61 571	70 503	137 188	15 122	137 188	88 464	48 724	44.70
4.73		132 256	10 565	132 256	61 805	70 451	136 999	15 200	136 999	88 920	48 079	46.53
4.95		132 434	10 605	132 434	62 039	70 395	136 800	15 278	136 800	89 376	47 424	48.44
5.18		132 610	10 644	132 610	62 267	70 343	136 597	15 354	136 597	89 821	46 776	50.38
5.40		132 782	10 684	132 782	62 501	70 281	137 188	15 122	137 188	88 464	48 724	44.24
5.63		132 951	10 723	132 951	62 730	70 221	137 188	15 122	137 188	88 464	48 724	44.12
2.25	Range					506.00					3095.35	7.63
4.50	Mean					70490.56					49223.90	43.33
0.17	Standard Error					50.57					0.00	1.29
16.58	Coefficient of Variation (%)					0.07					0.00	2.98

SOUTH COAST HAKE FISHERY: VARIABLE PRICE SCENARIO

	Dynamic Maximum Economic Yield					Dynamic Maximum Sustainable Yield					
Price	Catch	Effort	Revenue	Cost	Rent	Catch	Effort	Revenue	Cost	Rent	Gain (%)
0.75	34 359	37 759	25 769	19 635	6 135	52 709	98 188	39 532	51 058	- 11 526	na
0.80	36 762	41 433	29 410	21 545	7 864	52 709	98 188	42 167	51 058	- 8 891	na
0.85	38 748	44 682	32 936	23 235	9 701	52 709	98 188	44 803	51 058	- 6 255	na
0.90	40 407	47 575	36 366	24 739	11 627	52 709	98 188	47 438	51 058	- 3 620	na
0.95	41 807	50 170	39 717	26 088	13 628	52 709	98 188	50 074	51 058	- 984	na
1.00	42 998	52 509	42 998	27 305	15 693	52 709	98 188	52 709	51 058	1 651	850.40
1.05	44 019	54 628	46 220	28 407	17 813	52 709	98 188	55 344	51 058	4 287	315.55
1.10	44 900	56 559	49 390	29 411	19 979	52 709	98 188	57 980	51 058	6 922	188.63
1.15	45 666	58 324	52 516	30 328	22 187	52 709	98 188	60 615	51 058	9 558	132.14
1.20	46 335	59 945	55 602	31 171	24 431	52 709	98 188	63 251	51 058	12 193	100.37
1.25	46 922	61 438	58 653	31 948	26 705	52 709	98 188	65 886	51 058	14 828	80.09

0.50	Range	20570.17	26354.50	na
1.00	Mean	15978.59	1651.24	na
0.17	Standard Error	2069.47	2635.45	na
16.58	Coefficient of Variation (%)	12.95	159.60	na

SOUTH COAST HAKE FISHERY: VARIABLE COST SCENARIO

		Dynamic Maximum Economic Yield					Dynamic Maximum Sustainable Yield					
Cost		Catch	Effort	Revenue	Cost	Rent	Catch	Effort	Revenue	Cost	Rent	Gain (%)
0.39		47 754	63 682	47 754	24 836	22 918	52 709	98 188	52 709	38 293	14 416	58.98
0.42		46 922	61 438	46 922	25 558	21 364	52 709	98 188	52 709	40 846	11 863	80.09
0.44		46 031	59 199	46 031	26 166	19 865	52 709	98 188	52 709	43 399	9 310	113.38
0.47		45 079	56 965	45 079	26 660	18 419	52 709	98 188	52 709	45 952	6 757	172.60
0.49		44 068	54 735	44 068	27 039	17 029	52 709	98 188	52 709	48 505	4 204	305.05
0.52		42 998	52 509	42 998	27 305	15 693	52 709	98 188	52 709	51 058	1 651	850.40
0.55		41 868	50 286	41 868	27 456	14 412	52 709	98 188	52 709	53 611	- 902	na
0.57		40 680	48 068	40 680	27 495	13 185	52 709	98 188	52 709	56 164	- 3 455	na
0.60		39 432	45 853	39 432	27 420	12 012	52 709	98 188	52 709	58 716	- 6 007	na
0.62		38 126	43 641	38 126	27 232	10 894	52 709	98 188	52 709	61 269	- 8 560	na
0.65		36 762	41 433	36 762	26 931	9 831	52 709	98 188	52 709	63 822	- 11 113	na
0.26	Range					13087.47					25528.88	na
0.52	Mean					15965.63					1651.24	na
0.97	Standard Error					1310.95					2552.89	na
16.58	Coefficient of Variation (%)					8.21					154.60	na

SOUTH COAST HAKE FISHERY: VARIABLE QUASI-DISCOUNT RATE SCENARIO

Discount Rate		Dynamic Maximum Economic Yield					Dynamic Maximum Sustainable Yield					Gain (%)
		Catch	Effort	Revenue	Cost	Rent	Catch	Effort	Revenue	Cost	Rent	
3.38		42 747	52 005	42 747	27 043	15 704	52 794	96 788	52 794	50 330	2 464	537.29
3.60		42 798	52 107	42 798	27 096	15 702	52 779	97 063	52 779	50 473	2 306	580.86
3.83		42 848	52 207	42 848	27 148	15 700	52 763	97 350	52 763	50 622	2 141	633.32
4.05		42 898	52 308	42 898	27 200	15 698	52 746	97 625	52 746	50 765	1 981	692.42
4.28		42 948	52 408	42 948	27 252	15 696	52 727	97 913	52 727	50 915	1 812	766.10
4.50		42 998	52 509	42 998	27 305	15 693	52 709	98 188	52 709	51 058	1 651	850.40
4.73		43 047	52 608	43 047	27 356	15 691	52 688	98 475	52 688	51 207	1 481	959.48
4.95		43 096	52 708	43 096	27 408	15 688	52 668	98 750	52 668	51 350	1 318	1090.28
5.18		43 145	52 807	43 145	27 460	15 685	52 645	99 038	52 645	51 500	1 145	1269.61
5.40		43 194	52 906	43 194	27 511	15 683	52 623	99 313	52 623	51 643	980	1499.90
5.63		43 242	53 005	43 242	27 563	15 679	52 598	99 600	52 598	51 792	806	1845.33
2.25	Range					25.00					1658.24	1308.04
4.50	Mean					15692.77					1644.22	975.00
0.17	Standard Error					2.49					165.82	1.29
16.58	Coefficient of Variation (%)					0.02					10.09	0.13

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