Economic Production from the Coral Reef Fisheries of Jamaica and Captured Ecosystem Values

by

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We accept this dissertation as conforming to the required standard

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ABSTRACT

The production of an economic good derived from a renewable natural resource base involves the extraction of ecosystem function values as represented by the contribution made to production by the originating ecosystem. The artisanal fisheries of Jamaica is used as a case study in the examination of the characteristics of economic production processes and the development of a biophysically-based index to account for captured ecosystem values. The following is provided: i) a description of the fisheries of Jamaica and derivation of economic production function models; ii) a description of the socio-economic condition of the fisheries of Montego Bay Marine Park (Montego Bay, Jamaica) which serves to further illustrate the nature of artisanal fisheries in Jamaica, as well as a more traditional economic approach to resource valuation; and, iii) the development of an index which as a proxy measure captures the biophysical values of the contributions of the natural biotic environment (the "embodied ecosystem values") to the fisheries, and an examination of the extent to which those values are proportionately reflected in monetary exchange values. In addition, contributions are made concerning: i) the development of an economic data collection and analysis programme for Jamaica (also more widely applicable to countries of the developing tropics) which will allow for more informed decisions concerning the management of coral reef fisheries; ii) general principles concerning the development of biophysical indices, such as indices of biodiversity, which will ultimately be used to inform government policy and management decisions; iii) the validity of indices derived from ecosystem statistics; and, iv) the potential for the further development of models which explicitly incorporate the contributions of ecosystems to economic production processes.

Cobb-Douglas and translog models of fishing effort are derived from catch and effort data for the years 1996 and 1997 to describe the relationships between catch and firm-level inputs as they vary by fishery within Jamaica. Data on the total catch, crew size, gear soak time, and quantity of gear used yield separate functions of effort for the use of China net, trap, hand line, palanca, speargun, and troll fishing technologies. By further accounting for the month and fishing location (i.e. north coast versus south coast), the seasonal and regional influences on catch rates are explored. Patterns of production include reduced catch rates associated with fishing the north coast shelf and a seasonal peak in catch levels during the late summer and fall. The use of production function models of effort are found to provide informative descriptions of fishery production processes, yet avoid many of the technical difficulties associated with more traditional bioeconomic approaches. The Index of Captured Ecosystem Value (ICEV) is developed from a basis in information theory relevant to an analysis of network flows in ecosystems. Technical coefficients, describing the production relationship between ICEV values and market values of catches associated with individual fishing efforts, revealed that captured ecosystem function associated with fisheries using distinct technologies (i.e. China net, trap, hand line, palanca, and speargun) were valued differently by the market. This "surplus value" appears to be rooted in the observation that certain fisheries target species which are more connected within the coral reef food web than those species typically captured by other fisheries. Consideration of the biophysical contributions of coral reef ecosystems to fisheries production reveals distortions between market and supply-side values, indicating that the role of ecosystems is not being consistently treated.

Examiners:

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Chapter 1: A Context for Examining Ecosystem Contributions to Economic Production

Then leave Complaints: Fools only strive
To make a Great and Honest Hive
T'enjoy the World's Conveniencies,
Be Fam'd in War, yet live in Ease,
Without great Vices, is a vain
Eutopia seated in the Brain.
Fraud, Luxury and Pride must live,
While we the Benefits receive:
Hunger's a dreadful Plague, no doubt,
Yet who digests or thrives without?
Do we not owe the Growth of Wine
To the dry shabby crooked Vine?
Which, while its Shoots neglected stood,
Chok'd other Plants, and ran to Wood;
But blest us with its noble Fruit,
As soon as it was ty'd and cut:
So Vice is beneficial found,
When it's by Justice lopt and bound;
Nay, where the People would be great,
As necessary to the State,
As Hunger is to make 'em eat.
Bare Virtue can't make Nations live
In Splendor; they, that would revive
A Golden Age, must be as free,
For Acorns, as for Honesty.

Bernard Mandeville,
excerpt from *Fable of the Bees* (1705)

Introduction

The satirical words of Bernard Mandeville which denied the "good" or divine righteousness believed to be inherent within human-kind may have shocked early 18th century England trying to grapple with the philosophies of the Enlightenment, yet his words still carry meaning relevant today. The immediate economic advantages of human vice are indeed evident, be it war, crime, or other indulgences which may be deemed less than virtuous. Yet, one may apply the same principles to the exploitive relationship humans have with nature. An overextension of the waste assimilation capacities of the environment, the overextraction and alteration of natural resources, the elimination of species, and a lack of consideration for maintaining the integrity of natural ecosystems are documented behaviours of the economy. The short-term economic goals of individuals and firms, problems associated with the efficient provision of public goods, difficulties associated with the management of commons resources, and other market externalities have all contributed to pervasive environmental deterioration.

Central to the above concerns is the question of valuation and accounting - that is, accounting for the impact or potential impact of human activities on the environment, placing appropriate values on ecosystem contributions to economic production processes and non-use benefits, and internalising these values into decision-making processes. Until we can begin to efficiently and effectively consider the role of natural ecosystems in contributing to economic welfare, they will continue to be undervalued, leading to significant economic distortions and undesirable outcomes.

The Economics of Natural Capital and Ecosystem Services

It is generally recognised that a *laissez-faire* market economy is unable to efficiently allocate scarce resources when externalities, or values not accounted for in the market, are involved (e.g. Pearce and Turner 1990; Seneca and Taussig 1984). It has become a central pursuit of environmental and natural resource economics to determine the appropriate value of environmental goods and services and incorporate these values into economic accounting. The field has developed many useful and beneficial tools to assist in the estimation of external monetary benefits and costs associated with the natural environment (e.g. hedonic pricing; travel cost; contingent valuation to assess the willingness-to-pay) in an attempt to internalise such values for the determination of economically efficient outcomes. Of all the methods available, contingent valuation has seemingly received the most attention in recent years (particularly due to its ability to reveal non-use values), yet it is subject to significant caveats and limitations, and has been the subject of much debate (e.g. Arrow *et al.* 1993; Carson *et al.* 1996; Cummings and Harrison 1995).

In addition to significant methodological difficulties associated with the measurement of external values, traditional economic approaches may suffer from an inability to adequately consider "shocks" associated with the supply or functioning of biotic resources. This is due to inadequate awareness through political, cultural, or institutional structures affecting the choices available and altered perceptions of the consequences of actions. There are further problems of uncertainty and risk surrounding the associated ecological-economic processes. A lack of information exists concerning ecosystem processes and structures, the changes to ecosystem processes and structures that may be the result of human actions, and the effects of such changes on the human socio-economic condition (e.g. Bingham *et al.* 1995). Even where such information does exist, it is often unavailable to the decision-makers and the larger community. Indeed, a significant difficulty with contingent valuation methodology is its reliance on expressed willingness-to-pay regarding the provision of ecological services, the operation of which the respondents may have very little knowledge. Irreversibilities may also make the correction of any undesirable effects of past use decisions unlikely or impossible. This results in an inability to comprehensively internalise the values associated with certain natural system constraints, as well as ecosystem structural and functional relationships, within the economic system given currently accepted valuation methodologies.

The functioning of the human economy and ecosystems involve competing, complimentary, and interacting production processes that operate in a co-evolutionary environment (Holling *et al.* 1995). The values perceived by humans and the preferences expressed in the market system, or through other monetary valuation means, may not take into account what is necessary or relevant for ecosystem integrity for the maintenance of natural ecosystems and their associated services, nor does it necessarily take into account the co-evolutionary and complex nature of the environment-economy system. By recognising these drawbacks, the implicit assumption is made that we are no longer concerned only with individual human preferences operating within a framework of limited information. New tools are required to adequately value the environment.

Valuing the Environment

The application of capital theory to issues concerning the measurement of a society's ability to achieve sustainable development has received considerable attention. The central premise is similar to that associated with the operation of a business which must maintain its stock of produced capital to continue to earn income through its use - society as a whole must also maintain the stock of all capital on which it relies. If the stock of capital is depleted over time, this means that society is effectively living off the depletion of capital rather than living off the income generated through its sustainable use (Daly 1994; Daly and Cobb 1994). This principle is traced to the concept of Hicksian income: "the maximum amount that a community can consume over some time period and still be as well off at the end of the period as at the beginning" (Hicks 1946, as cited in Daly 1994). This does not necessarily require maintaining or enhancing the total physical stock of all forms of capital, as long as the potential for humans to draw a flow of goods and services from that capital never decreases in value (various operational definitions for maintaining capital are evident - e.g. see Spash and Clayton 1997).

Natural capital has been defined as the stock that yields the flow of all renewable and non-renewable resources - the source of income and non-marketed benefits generated through the use of those resources (e.g. Costanza and Daly 1992; Daly 1994; Folke *et al.* 1994). The concept of natural capital represents an extension of the usual concept of capital, central to neo-classical economics, in which capital is taken to include only human-produced durable means of production. More all-encompassing definitions for capital have been put forward, including "anything which yields a flow of productive services over time and is subject to control in production processes" (Kneese and Herfindahl 1974, as cited in Victor 1991), or broadly "a stock that yields a useful flow of goods or services into the future" (Daly 1994). Such a definition could include labour, land, and natural resources as capital. Further, capital used in production need not be restricted to physical capital, but may also include intangible forms such as information (Costanza *et al.* 1997). In all, it has become more common to distinguish between four forms of capital: i) produced capital; ii) natural capital (including physical stocks, energy, and information); iii) human capital (the set of human abilities and know-how); and, iv) social capital ("the set of norms, networks and organisations through which people gain access to power and resources, and through which decisions and policy formulation occur" and that affects and is affected by economic outcomes, including aspects of horizontal associations, civil and political society, social integration, and legal and governance aspects; World Bank 1997, p.78).

Valuing the environment is ultimately linked to estimating the value of the flow of ecosystem services (taken to include goods, services, or other benefits derived from natural capital). Questions arise as to whether or not marginal values (prices), producers' and consumers' surplus (considering total value), or producers' surplus (rent) should be considered in the valuation (Ayres 1998; El Sarafy 1998), yet the decision on the appropriate methodology must be specific to the intended use of the information. de Groot (1994) defined ecosystem functions as they relate to economic values as "the capacity of natural processes and components to provide goods and services that satisfy human needs (directly and/ or indirectly)." It must be borne in mind that economic value itself is not achieved until a useful ecosystem function is applied to satisfying such needs. Heuting (1980) emphasised the potential as well as current use of the environment, noting the need to sacrifice the use of current production in order to ensure availability in the future. Services from such functions include materials, energy, and information used to contribute to human welfare (e.g. Costanza *et al.* 1997).

The bounds of what ecosystem function should be subject to valuation or what indeed is meant by "values" is by no means agreed upon within the academic community (e.g. Bingham *et al.* 1995). However, more strictly within economics, some guidelines have emerged. It is usual within the field of natural resource economics to divide economic values associated with ecosystem services into two primary categories: 1) use values (including direct use values in the case of marketed goods and services, indirect use values in the case of unmarketed environmental or ecosystem services, and option values); and, 2) non-use values (including existence and bequest values).

There have been many attempts to place a value on ecosystem services. The specifics of the methodologies and nature of the controversies will not be expanded upon here (e.g. see Daily 1997; various contributions in a special issue of *Ecological Economics* 25(1), 1998). The list of potential services is long and varied. The following have been addressed in the literature: generating and maintaining soils; maintaining hydrological cycles; regulating gas; nutrient cycling and storage; assimilation and elimination of pollution and wastes; regulating disturbances; biological control; pollination of crops; food production; maintaining species and genetic resources; maintaining biogeochemical cycling; and, regulating weather and climate (Bingham *et al.* 1995; Costanza *et al.* 1997; Folke *et al.* 1994; Myers 1996). Perhaps the most notable recent valuation is that of Costanza *et al.* (1997) who reviewed the literature and estimated a current marginal value of ecosystem services of at least US$33 trillion annually on a global basis, most being outside the market system (significantly greater than the estimated global GNP of US$25 trillion for the base year 1994). This estimate and the methodology used to arrive at the estimate, however, do not lack criticism. The exercise did serve to draw considerable attention to the issue of ecosystem service valuation, although care must be taken in not providing a misleading perception of the science behind the estimates, and the extent of their direct utility, through such popular accounts (Toman 1998).

The Concept of Economic Rent - Classical Inspirations and Neoclassical Treatments

A classical economic conceptualisation of rent provided inspiration for the exploration of ecosystem contributions to production as approached by this dissertation. The classical economists, most notably including Malthus, Ricardo and Marx, had developed their models of rent with obvious roots in the earlier physiocratic theories, which considered nature as the source of wealth and rent as "unrecompensated" work done by nature (Beer 1939; Cleveland 1987). Succinctly, rent as defined by Ricardo (1821/1960, p.33) is "...that portion of the produce of the earth which is paid to the landlord for the use of the original and indestructible powers of the soil." In general terms, land rent is conceived as a payment for the contribution of nature to productive value. Granted, ecological resources such as fisheries are more accurately described as potentially renewable as opposed to "indestructible" resources, yet classical economists, including Ricardo, often considered fisheries and agriculture as analogues in the exposition of natural resource economic principles applied to land envisioned to have a characteristic set of natural properties.

More formally, Ricardian rent in the extensive case is the return to the landlord or the owner of the natural resource equal to the difference in productivity between the land in question and the land of lowest quality brought into production with a given amount of capital and labour to meet a given demand for a particular commodity (the land of lowest productivity for the given commodity earning no rent; Ricardo 1821/1960). Specifically,

> "If all land had the same properties, if it were unlimited in quantity, and uniform in quality, no charge could be made for its use, unless where it possessed peculiar advantages of situation. It is only, then, because land is not unlimited in quantity and uniform in quality, and because, in the progress of population, land of an inferior quality, or less advantageous situation, is called into cultivation, that rent is ever paid for the use of it." (Ricardo 1821/1960, p.35)

Thus, rent can similarly be defined as payment due to the inelastic supply of land of a specified quality. This is not to say that the highest quality of land is necessarily brought into production first, as demand and location conditions may not be sufficient to so affect resource use decisions (Marshall 1890/1961; Marx 1894/1991). Ricardian rent is not earned for goods and services provided by nature which are not scarce and exclusive, but arises due to the scarcity of land of particular quality, exclusion of use through ownership, and variations in natural productivities between lands.

> "If air, water, the elasticity of stream, and the pressure of the atmosphere were of various qualities: if they could be appropriated, and each quality existed only in moderate abundance, they, as well as the land, would afford a rent, as the successive qualities were brought into use." (Ricardo 1821/1960, p.39).

It is important to note that Ricardo considered rent as separate from returns to capital and labour employed in the production process; specifically, profits and wages are determined differently from rent, which itself can only be attributed to the natural attributes of the land. Ricardo also considered the intensive rent case - that is, the application of different production technologies (i.e. an intensification of capital) on different parcels of land. However, intensification of the application of capital would not necessarily contribute to a change in the rent earned, for as defined

> "...rent is always the difference between the produce obtained by the employment of two equal quantities of capital and labour." (Ricardo 1821/1960, p.36)

This is to say that a change in productivity due to a change in the production technologies employed will result in a change in the returns to capital and thus labour, and not to a change in the rent unless it subsequently changes the differential yields between parcels of land not attributable to differences in the use of technologies. Changes in the use of production technologies across lands of varying qualities for the production of a given commodity greatly complicates the calculation of Ricardian rent. In a more general manner, estimation is further complicated by, among others, the difficulty in determining the appropriate return to capital and labour, the existence of implicit rents, the segregation of markets, market disequilibrium, and uncertainty.

Other classical economists similarly conceptualised land rent. Of note, Marx (1894/1991) also attempted to develop a theory of rent, borrowing heavily from the earlier works of Malthus and Ricardo. Marx distinguished between extensive rent due to differences in the natural qualities of the land (although also recognising the additional possibility of a von Thünen-type location rent),

> "...it arises from the greater relative returns from certain particular capitals invested in a sphere of production, as compared with those capital investments that are excluded from these exceptional, favourable conditions of productivity which have been created by nature." (Marx 1894/1991, p.785)

as well as intensive rent due to differences in the application of capital in the absence of land quality differences. However, Marx was unable to fully reconcile the concept of land rent within a labour theory of value. As with any capitalist production, work can be subdivided into necessary and surplus labour:

> "Just as a part of agricultural labour is objectified in products that either serve simply for luxury or form industrial raw materials but in no way go into foodstuffs, at least not foodstuffs for the masses, so on the other hand a part of industrial labour is objectified in products that serve as necessary means of consumption for agricultural and non-agricultural workers alike." (Marx 1894/1991, p.771)

But Marx further considered the application of capital, which adds value to the product of the land, to become incorporated over time as a component of rent captured by the landlord. This represents a redistribution of surplus value - surplus value being created from surplus labour and the application of capital as with any industrial process. For Marx, land rent is derived from surplus value alone. This ultimately lead to an inconsistent treatment of payments for capital improvements, a difficulty in defining socially necessary labour values as applied to land use, and thus an insufficient theory of rent (Bryan 1990).

It is not the intent to provide a comprehensive summary of classical rent theory here (e.g. see Kurz and Salvadori 1995; Sraffa 1960), but to highlight the most significant contributions as they provided inspiration for the treatment of ecosystem contributions to the production process. The distinct discussion of contributions to economic value by the natural qualities of land, as done by classical economists such as Ricardo and Marx, is useful for our purposes. Although admittedly a simplification and abstraction, the essential feature of the classical economic concept, which should be gleaned for use in the present study, is more conceptual than necessarily measurable - *rent is a return for the productive value provided by nature.*

Neo-classical treatments of rent differ markedly from the classical economic approach. The more recent neo-classical analyses have modified the concept - in essence, equating rent with producers' surplus specific to any institutional arrangement which does not allow for the given surplus to be competed away because an inelastic supply is maintained, at least in the short-run (rent maintained in the short-run but not the long-run is usually referred to as quasi-rent; producers' surplus can be defined as the difference between the actual price at which something is sold and the minimum price at which it would be sold - more simply, it is the difference between the price paid and the opportunity cost of providing the good or service). The development of the neo-classical microeconomic marginal approach has not found it useful or practical to maintain the classical emphasis on the productivity differences of land as leading to a separate and distinct concept of rent. Production processes are instead preferably modelled as involving only two primary factors of production - capital and labour. Many criticisms by neo-classical economists have been aimed at Ricardian rent theory (e.g. see the reviews in Bird 1975; Kurz and Salvadori 1995).

Marshall, whose ideas marked the beginning of neo-classical analysis, noted that the properties of land consist of both the "...original and inherent properties, which the land derives from nature, and the artificial properties which it owes to human action..." (1961, p.147), but maintained that both were effectively inseparable. The productive characteristics of the soil, in interaction with market supply and demand conditions, will determine the producers' surplus to be earned. It is the problem associated with the inseparability of the components of producers' surplus, the lack of a singular or operational means to determine the contribution by the "indestructible powers" of nature to production, and a belief that such an exercise is unwarranted and undesirable within a neo-classical paradigm which has lead to the transformation of the classical conceptualisation of rent into the broader realm of producers' surplus.

The theory of the firm as provided by neo-classical economics has indeed provided for very fruitful and useful explorations in natural resource economics (e.g. Dasgupta and Heal 1979; Scott 1985; Smith and Krutilla 1982). However, as natural resource rent is largely relegated to analysis within the general neo-classical body of theory, one even questions the need for "rent" as a separate and useful concept (Bird 1975). This has promoted a continued inability to treat the provision of environmental goods and services as produced by nature as separate and apart from supply attributed to value-added by human actions through the application of either capital or labour alone.

The Project - Scope and Objectives

The dissertation will focus on the extraction of ecosystem function value as represented by the flow from natural biota to the human economy (Figure 1.1). This is best conceptualised as operating within a closed system which is governed by a set of natural laws which determine the limits and terms of interaction between the abiotic set of resources and physical conditions, the biotic set of ecosystems, and the human economy. Both the human economy and the activities of the natural biota are viewed as competing, complimentary and interacting production processes that operate in a co-evolutionary environment.

The project will take as given the following two premises: 1) the production of goods and services by the human economy relies on the drawing of services provided by natural capital and thus on the productivity of natural biotic systems; and, 2) the drawing of services provided by natural capital effectively captures or "embodies" function value of the originating ecosystem. It is through this involvement of natural biota in economic production processes that contributions by the originating ecosystem are made to the value of the final economic goods and services (i.e. through a "value-added" process of which the natural biota is a part). In effect, this dissertation will be concerned with exploring a form of contributory value (Ulanowicz 1991).

The total monetary value of the contribution of natural resources to the final value of goods and services whose production utilises those resources is ultimately represented economically by the resource rent earned. Such rent is paid to the owners or users of the resource, be they government, private sector, or non-government organisation. Rents earned by scarce resources as traditionally defined within neo-classical economics may effectively become negligible or zero (or even negative) due to, in part, the value of the resource being external to product supply decisions.

The classical conceptualisation of land rent is particularly attractive as it provided inspiration for this project because it holds possibilities of being harmonised with a theory of value based on ecosystem contributions. However, as strictly interpreted, the classical definition necessarily dictates that one must determine the absolute level of the scarce natural resources which are made available by nature. Although inspired by a classical definition of rent, the project will not revert to a classical analysis in which there are known or realised natural resource limits. This project will not approach the problem at hand from the perspective of neo-classical economic rent, the interpretation of which is complicated by institutional and historical circumstances. Note also that zero, or even negative, resource rents (as more broadly defined in neo-classical terms) theorised for ineffectively managed natural resources (e.g. Gordon 1954; Schaefer 1957) do not necessarily mean that the "embodied ecosystem values" are reflected in the monetary exchange values of such goods - the underlying biophysical values may simply have a zero or negative supply price. The dissertation will explore a biophysically-based analysis of the contribution of natural ecosystems to economic production processes.

Specifically, this project will focus on the examination of the direct extraction of ecosystem function value in the coral reef artisanal fisheries of Jamaica (Figure 1.2). The dissertation is divided into the following sections: 1) a description of the fisheries of Jamaica and derivation of economic production function models for the artisanal fisheries (Chapter 2); 2) a description of the socio-economic condition of the fisheries of Montego Bay Marine Park (Montego Bay, Jamaica) which serves to further illustrate the nature of artisanal fisheries in Jamaica, as well as a more traditional economic approach to resource valuation (Chapter 3); and, 3) the development of an index which, as a proxy measure, captures the biophysical values of the contributions of the natural biotic environment (the "embodied ecosystem values") to the fisheries, and an examination of the extent to which those values are proportionately reflected in monetary exchange values (Chapter 4). In addition, contributions will be made in the dissertation concerning: i) the development of an economic data collection and analysis programme for Jamaica (also more widely applicable to countries of the developing tropics) which will allow for more informed decisions concerning the management of coral reef fisheries; ii) general principles concerning the development of biophysical indices, such as indices of biodiversity, which will ultimately be used to inform government policy and management decisions; iii) the validity of indices derived from ecosystem statistics; and, iv) the potential for the further development of models which explicitly incorporate the contributions of ecosystems to economic production processes.

The ecosystem function value inherent in the extraction, production and sale of fisheries products will be considered to only be represented by the products themselves as reflected in a developed index. It must be emphasised that this is not a bioeconomic exercise in the traditional sense - the question is not what is the point of maximum sustainable economic yield of the fisheries, but how the biophysical structure and function value represented by harvested fishes is reflected in marketed product. Indeed, it is the development of a general, meaningful and applicable model for the consideration of ecosystem value in recognition of the indeterminancies inherent in the co-evolutionary nature of economic and ecological systems, rather than the specific documentation of ecosystem state and response to perturbation, which is the ultimate goal.

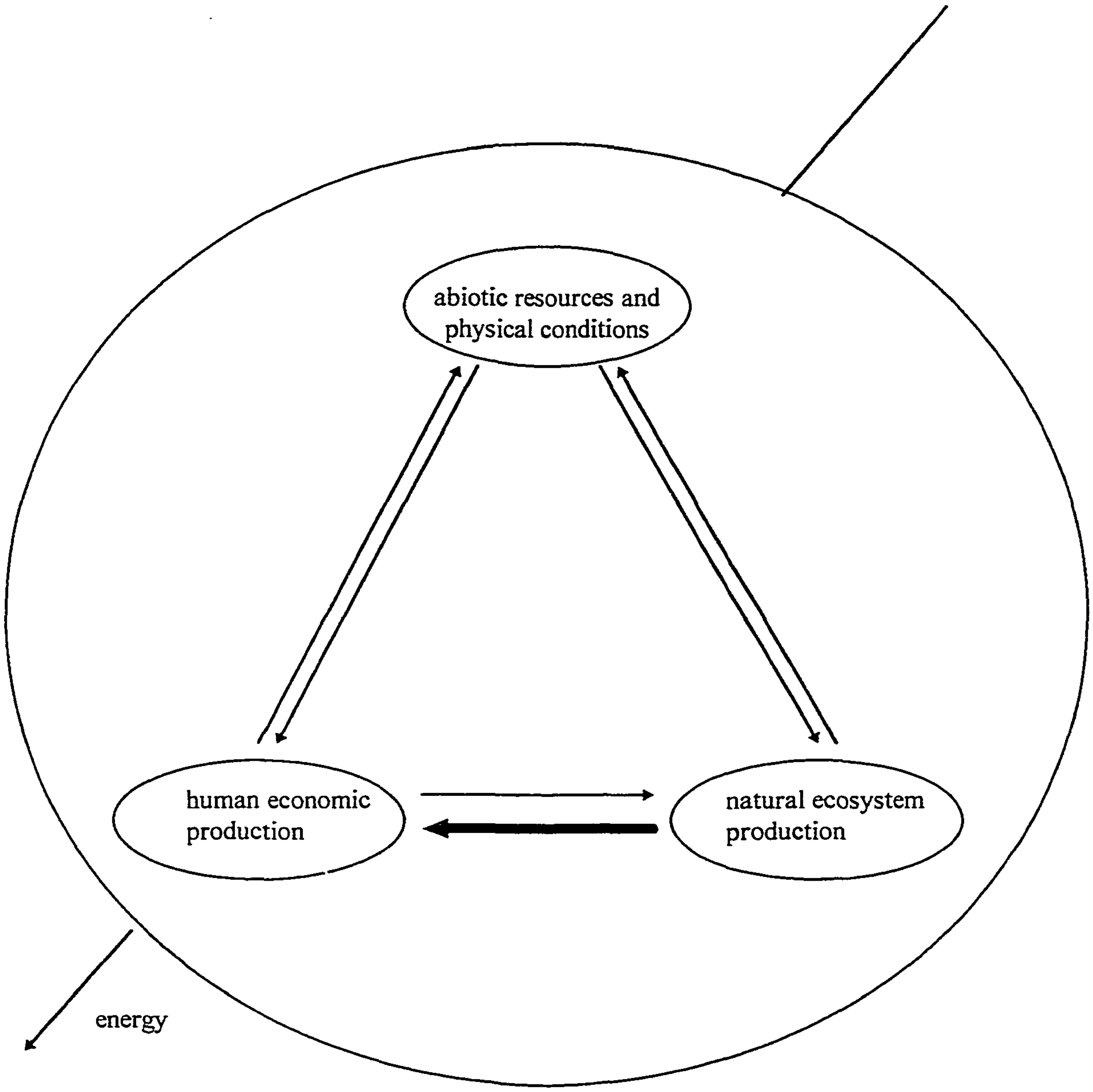

Figure 1.1. The interactions and interdependencies of abiotic resources and physical conditions, human economic production, and natural ecosystem production as envisioned within a closed system. Energy is captured by all three components of the system. Emphasised arrow shows the focus of this study.

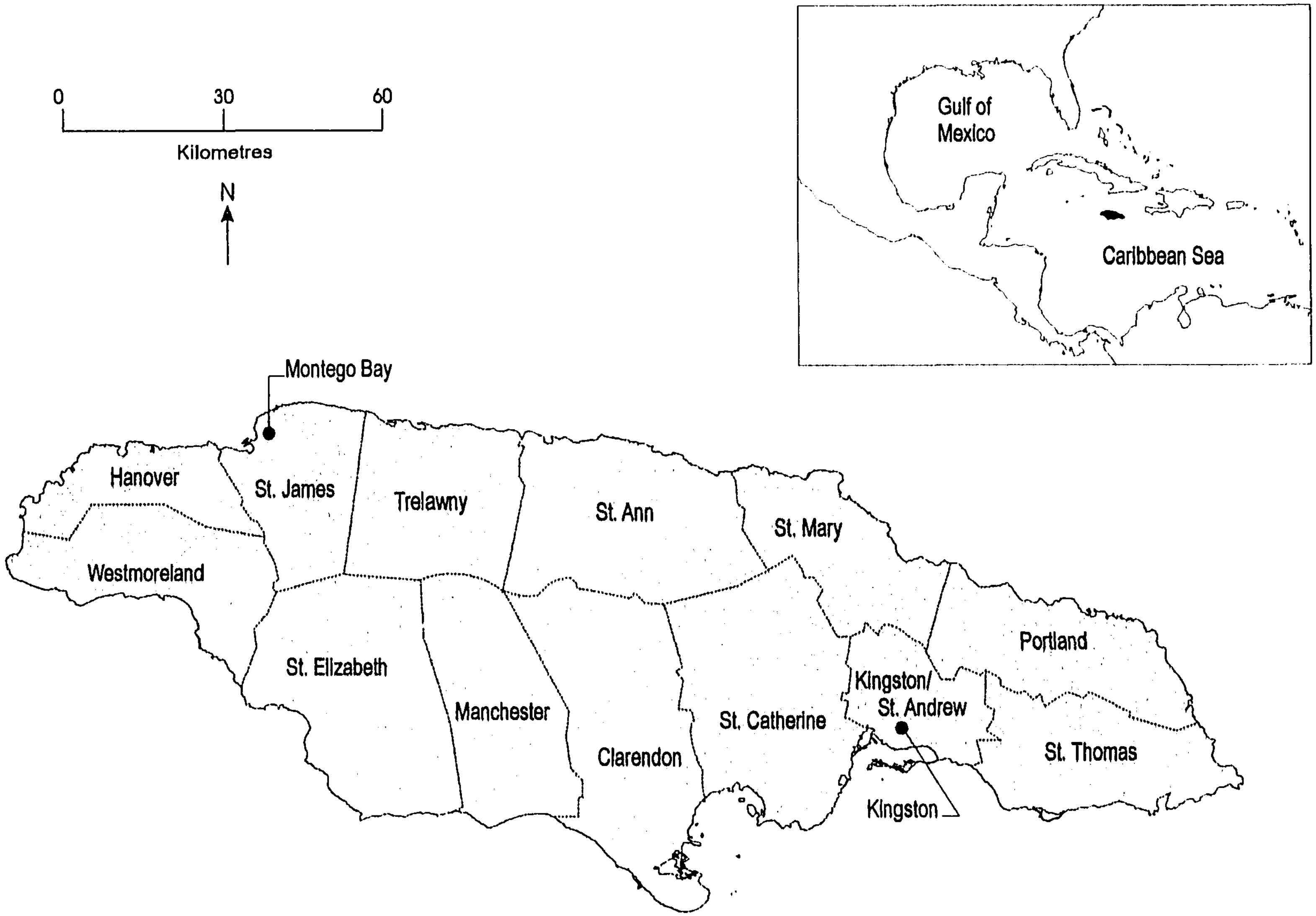

Figure 1.2. Jamaica, showing parishes and the locations of the urban centres of Kingston and Montego Bay.

Chapter 2: Production in the Coral Reef Fisheries of Jamaica

Introduction

Jamaican fisheries can be broadly classified into two sectors - the inshore fishery and the offshore fishery (based on the different logistics involved; Aiken 1993; Espeut 1992; Espeut and Grant 1990; Haughton 1988). Alternatively, it can be viewed as operating within four distinct geographical regions - the north shelf, the south shelf, the small proximal banks, and the offshore banks (i.e. the Pedro and Morant Banks which are Jamaican territory, as well as approximately eight banks in international waters further to the south and west; Espeut 1992; Espeut and Grant 1990; Figure 2.1). The south shelf reaches a width of approximately 24 km, while the north shelf reaches a width no greater than 1.6 km (Aiken 1993; Haughton 1988). The Jamaican island shelf and the proximal banks have a total area of approximately 4,170 km^2 (Aiken 1993; Haughton 1988; the quoted total area figure refers to nine proximal banks, not a larger complement of twelve as identified by Espeut 1992 and Espeut and Grant 1990, and as shown on Figure 2.1). The Pedro and Morant Banks have average depths of approximately 20 to 30 m, the Morant Bank has an estimated area of 100 km^2, while the much larger Pedro Bank has an estimated total area of 8,036 km^2 (Aiken 1993). Both Pedro Bank and Morant Bank contain three cays, with two on each bank being occupied by fishers on a permanent basis (Espeut 1992; Espeut and Grant 1990).

The inshore fisheries are typified by the use of small canoe-type vessels (cottonwood dugouts, or boats of fibreglass or plywood construction), which may be motorised or propelled by oars (Aiken 1993). There has been an increasing movement towards the use of mechanised, fibreglass canoes over wooden construction (Aiken and Koslow 1992; Espeut 1992; Espeut and Grant 1990). Inshore fishers base their operations out of landing sites located on the mainland of Jamaica.

The offshore fisheries are typified by the use of canoe-type vessels as in the inshore fishery, but the fishers are based more or less permanently on the offshore cays which form part of the Morant and Pedro Banks. These resident fishers are serviced by larger packer boats operating from the mainland, carrying supplies to the cays and the daily catch to the mainland markets. In addition to the offshore artisanal fishers, there are larger independent industrial fishing vessels which operate offshore or along the Pedro and Morant Banks (Aiken 1993; Grant and Blythe 1995; Haughton 1988). The separation of offshore and inshore fisheries as presented here, however, is a simplification. For example, mainland motorised canoe fishers from the south coast are known to fish the Pedro Banks (Aiken 1990; Munro and Thompson 1983).

Jamaican fishers target a wide range of marine species, including ocean pelagics, coastal pelagics, deepslope finfish, coral reef finfish, lobster, conch, and other marine invertebrates (most notably, shrimp, crab, and oysters; Fisheries Division, Government of Jamaica, unpublished data). The coral reef-based fisheries are multi-species fisheries, with over 350 species known to have inhabited the waters of the Caribbean and over 200 species believed to be landed by Jamaican fishers (Espeut 1992; Munro 1983a).

The organisational structure of fishing activities (fishing, processing, marketing, etc.) vary according to the type of fishery (industrial versus artisanal, inshore versus offshore, as well as by target species). The typical biomass flows of the catches associated with the dominant fisheries are shown in Figures 2.2 through 2.8. Most fish caught by Jamaican fisheries are destined for domestic consumption as fresh fish, distributed largely through an informal yet highly functional marketing system (Espeut 1992; Bunce and Gustavson 1998). For the inshore artisanal fishers, the selling to retail "higglers" (who in turn sell to restaurants or the public in the markets or on the roadside), the selling to wholesale higglers (who buy on bulk and in turn sell smaller quantities primarily to retail higglers), and the direct selling to the general public on the roadside or on the landing sites, are the most common means used to market their product (Espeut 1992; Espeut and Grant 1990). The offshore artisanal cay fishers rely either on carrier vessels to carry their catch to the Kingston Fisheries Terminal for sale or on mainland-based canoe vessels to purchase and transport the catch to landing sites for resale to higglers or directly to consumers (Espeut 1992; Espeut and Grant 1990).

The most notable exception to the typical marketing structures evident in artisanal or mixed artisanal/ industrial fisheries in Jamaica (i.e. those providing fresh fish sales for domestic consumption) is that associated with the conch fishery (Figure 2.7). The main target of the conch fishery is the Queen Conch (*Strombus gigas*). Artisanal free-lung, hookah and scuba divers fish both the inshore and offshore areas, yet the dominant industrial interests in the fishery are concentrated on the Pedro Bank (Aiken 1993; Alleyne 1996; Grant and Blythe 1995). In 1994, the open access management regime was replaced. Following a stock assessment, a combined license and individual transferable quota (ITQ) system was instituted as a direct result of the need to demonstrate the sustainable management of the fishery to qualify for an exemption under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES; see Armstrong and Crawford 1998 for a description of the convention) to allow for the continued export of the product (Alleyne 1996). The industrial interests currently hold approximately 90% of the ITQ's, the product destined primarily for export (Alleyne 1996).

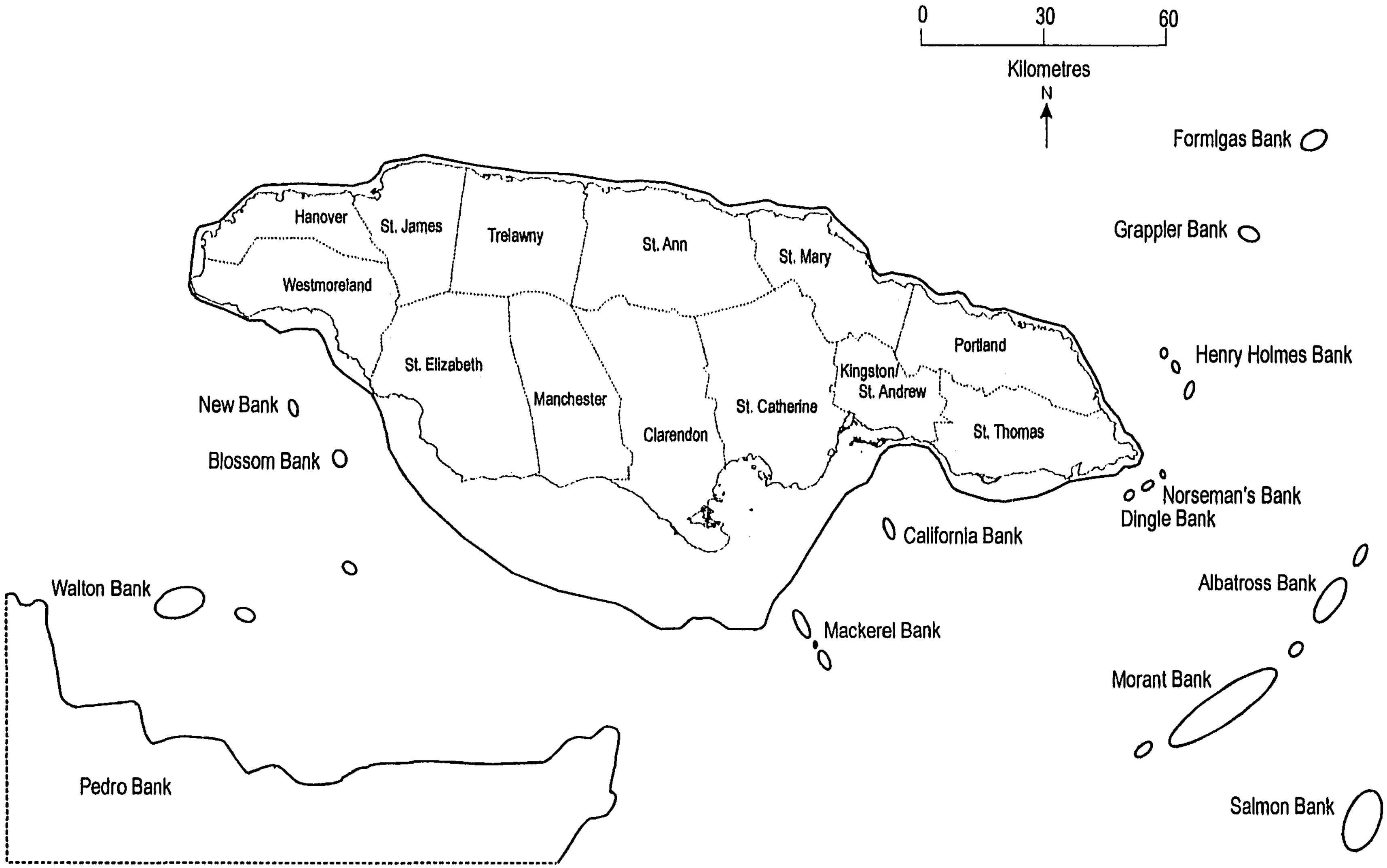

Figure 2.1. Shelf and banks, including the northern tip of the Pedro Bank, associated with the fisheries of Jamaica (note: shelf and bank representations are not intended to indicate precise shape or location; adapted from Espeut 1992 and Haughton 1988).

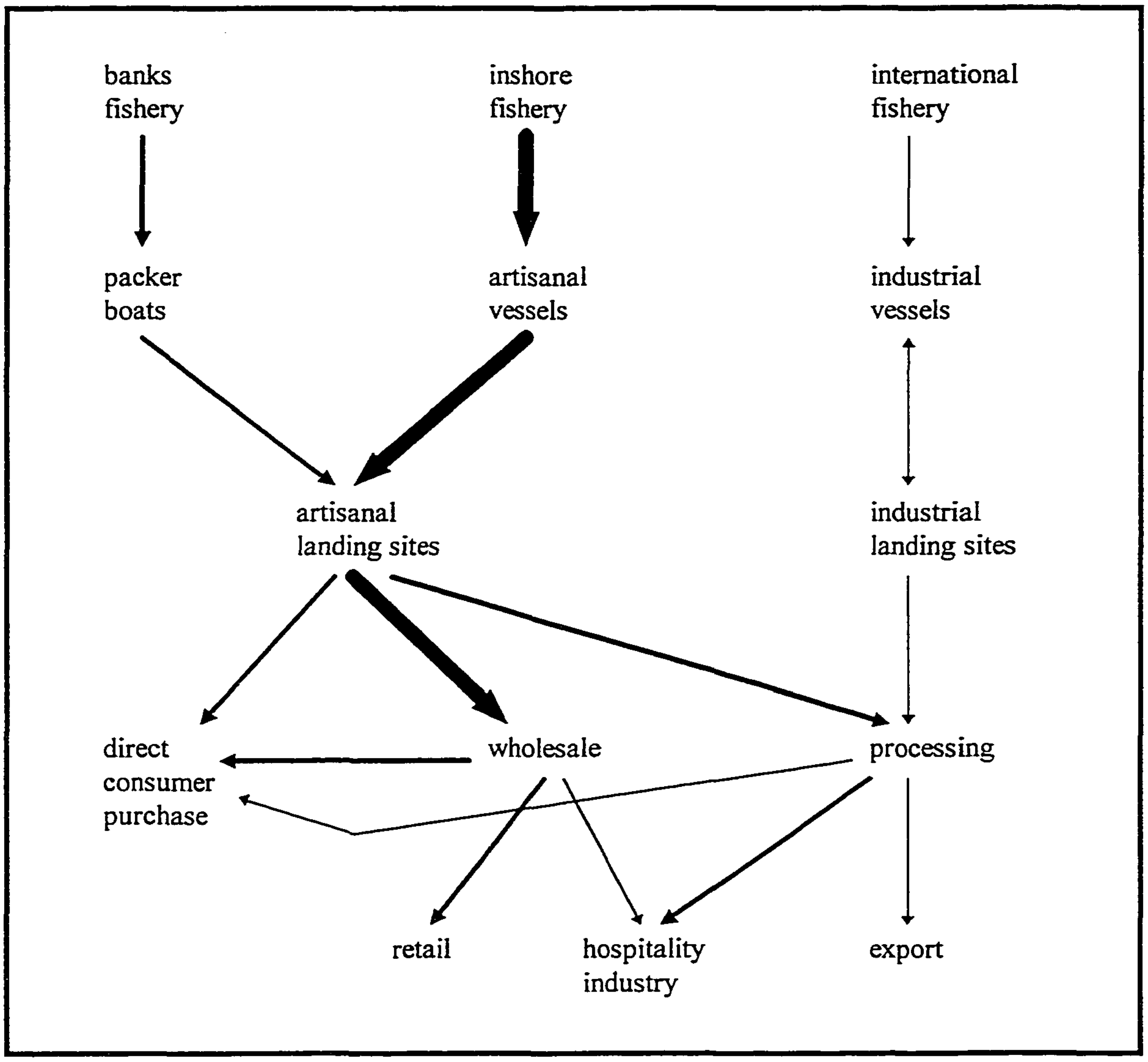

Figure 2.2. Economic source and destinations typical of the coral reef finfish fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

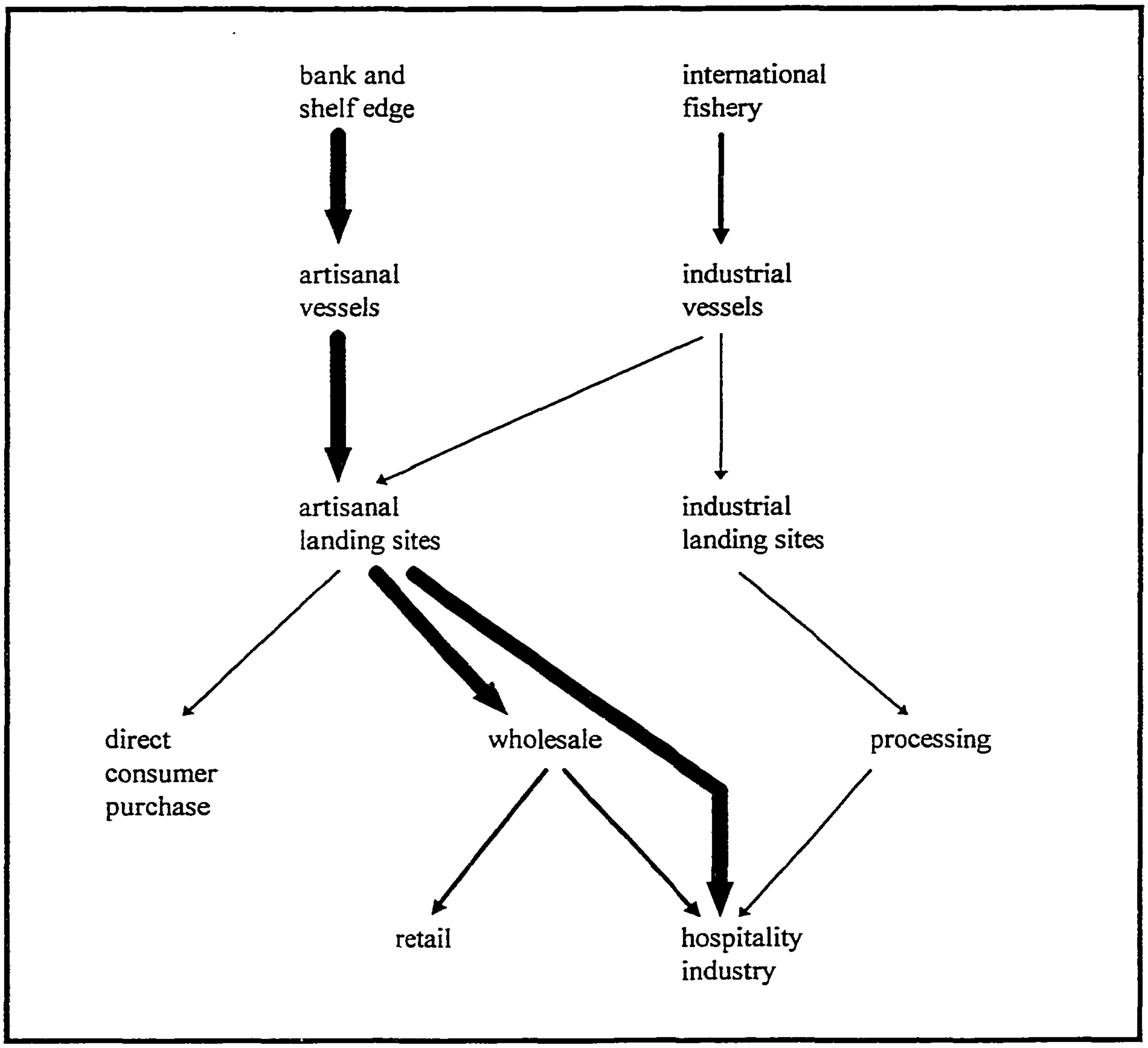

Figure 2.3. Economic source and destinations typical of the deepslope finfish fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

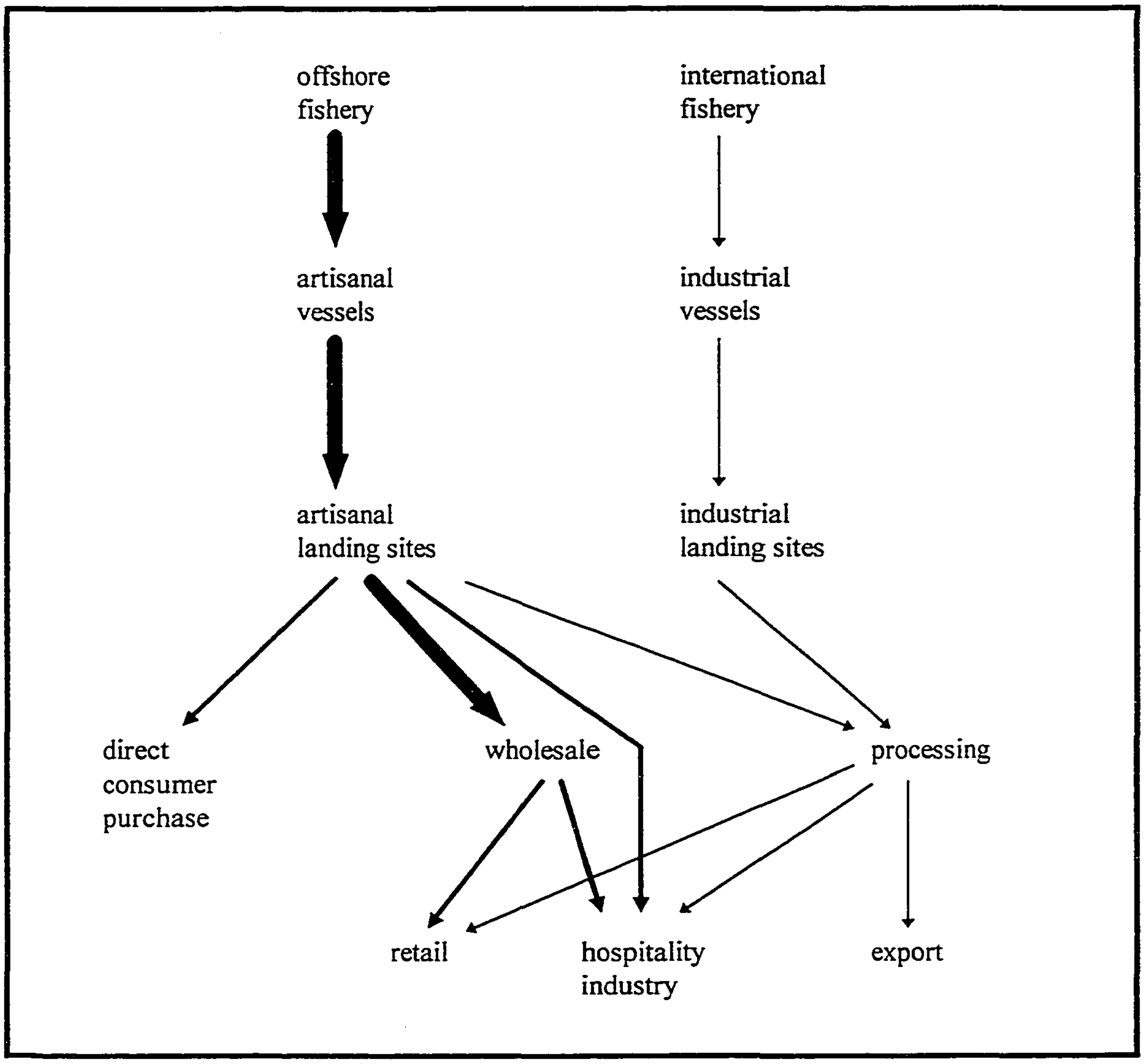

Figure 2.4. Economic source and destinations typical of the offshore pelagic finfish fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

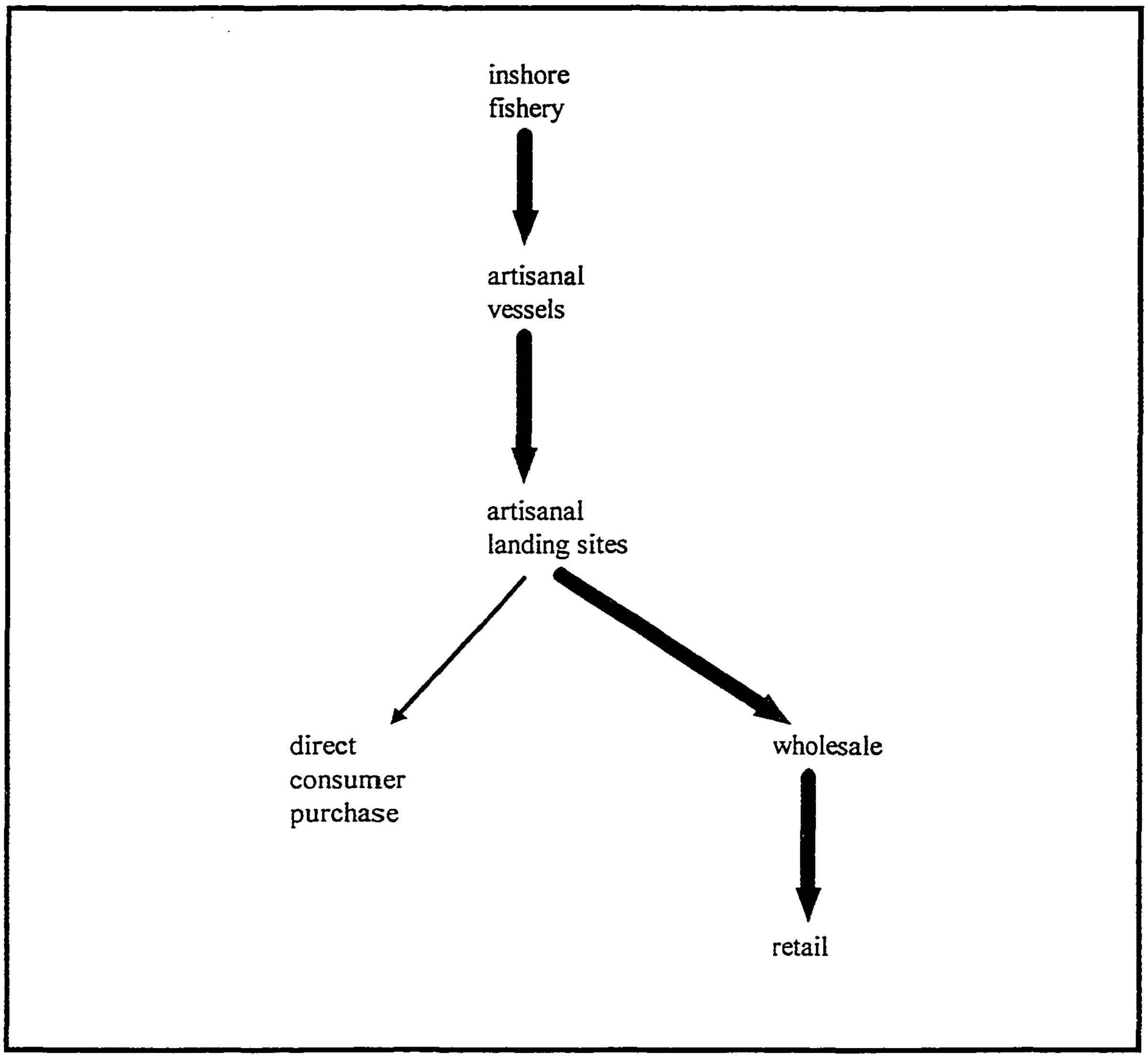

Figure 2.5. Economic source and destinations typical of the coastal pelagic finfish fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

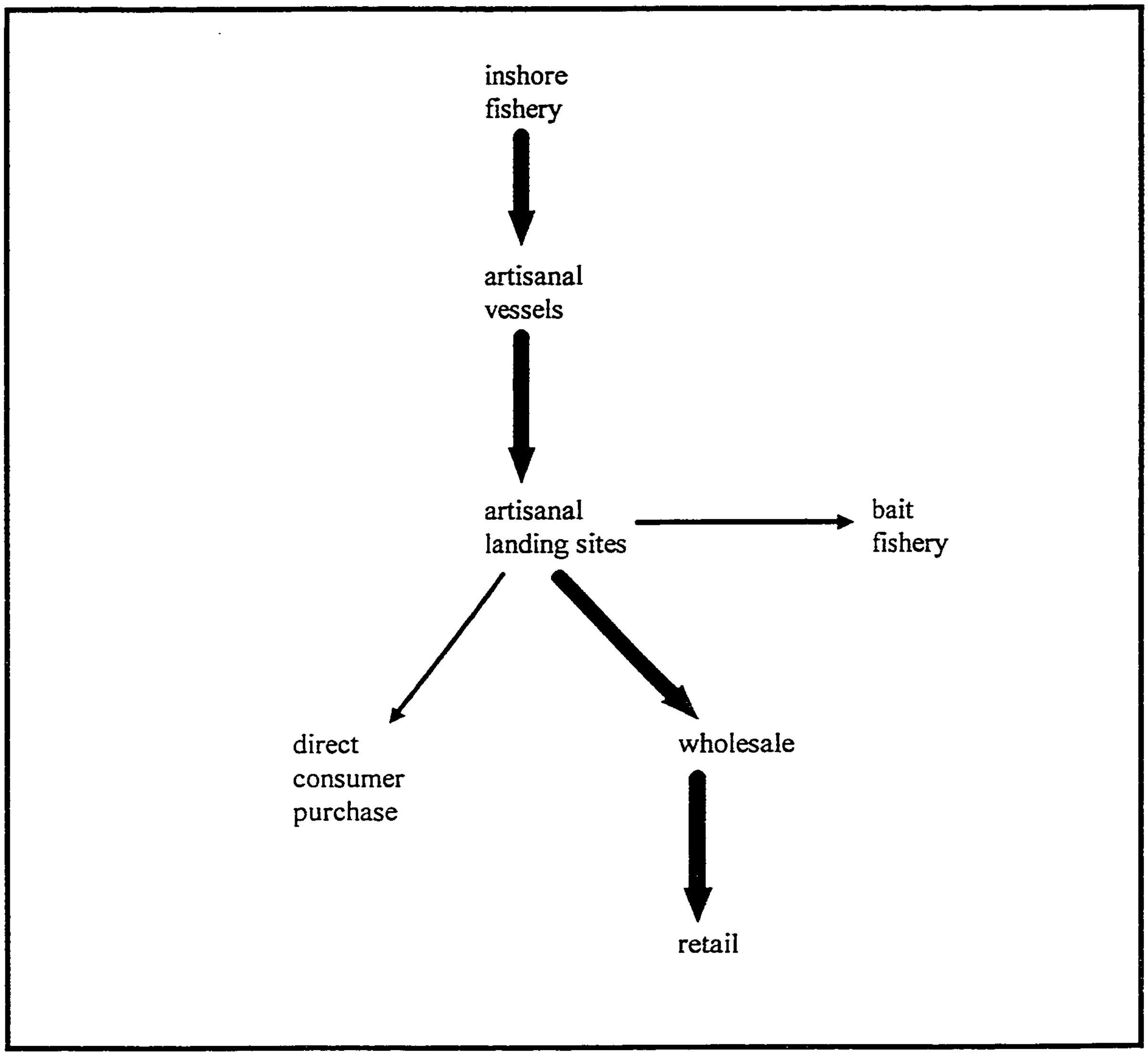

Figure 2.6. Economic source and destinations typical of the shrimp fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

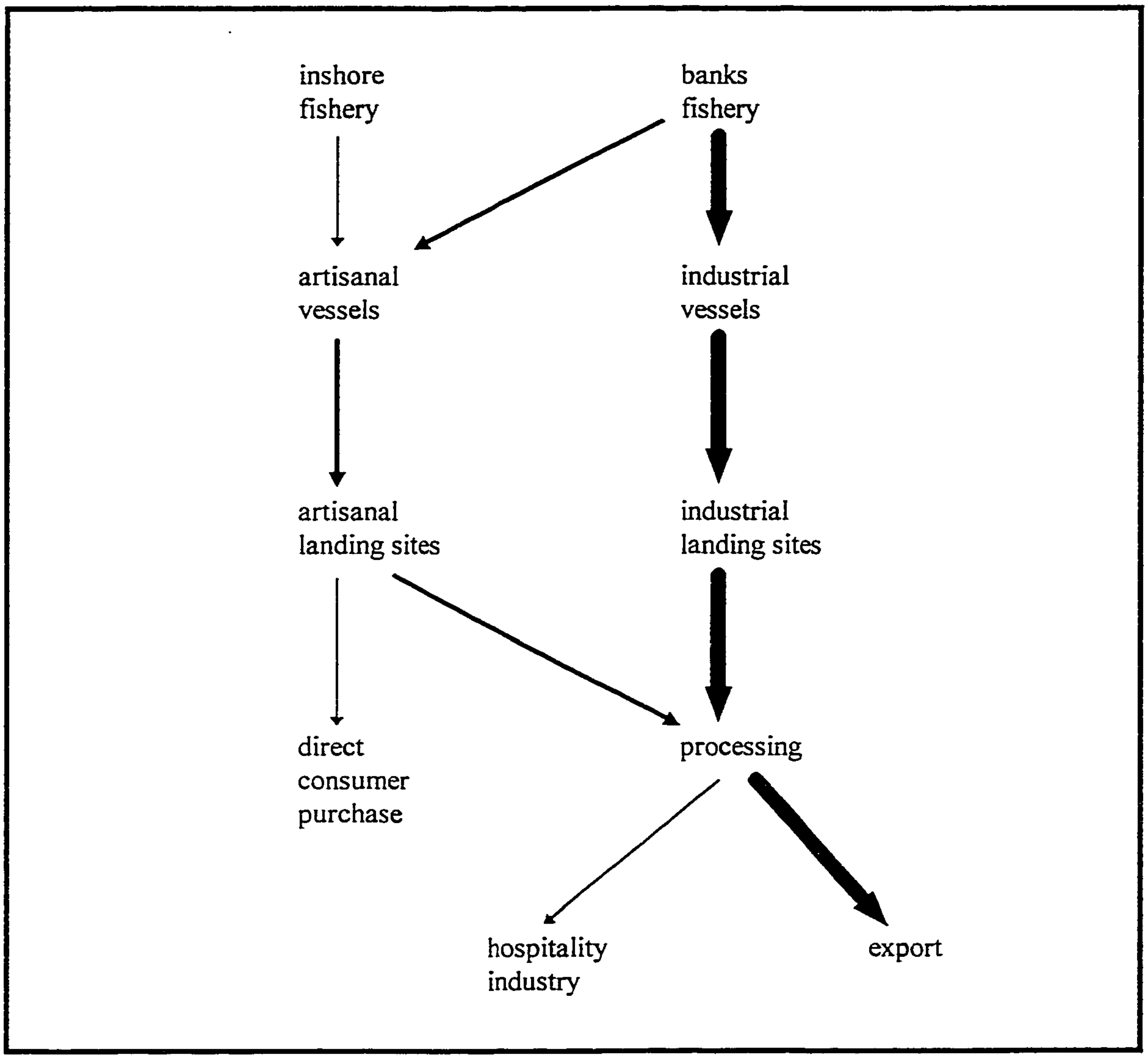

Figure 2.7. Economic source and destinations typical of the conch fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

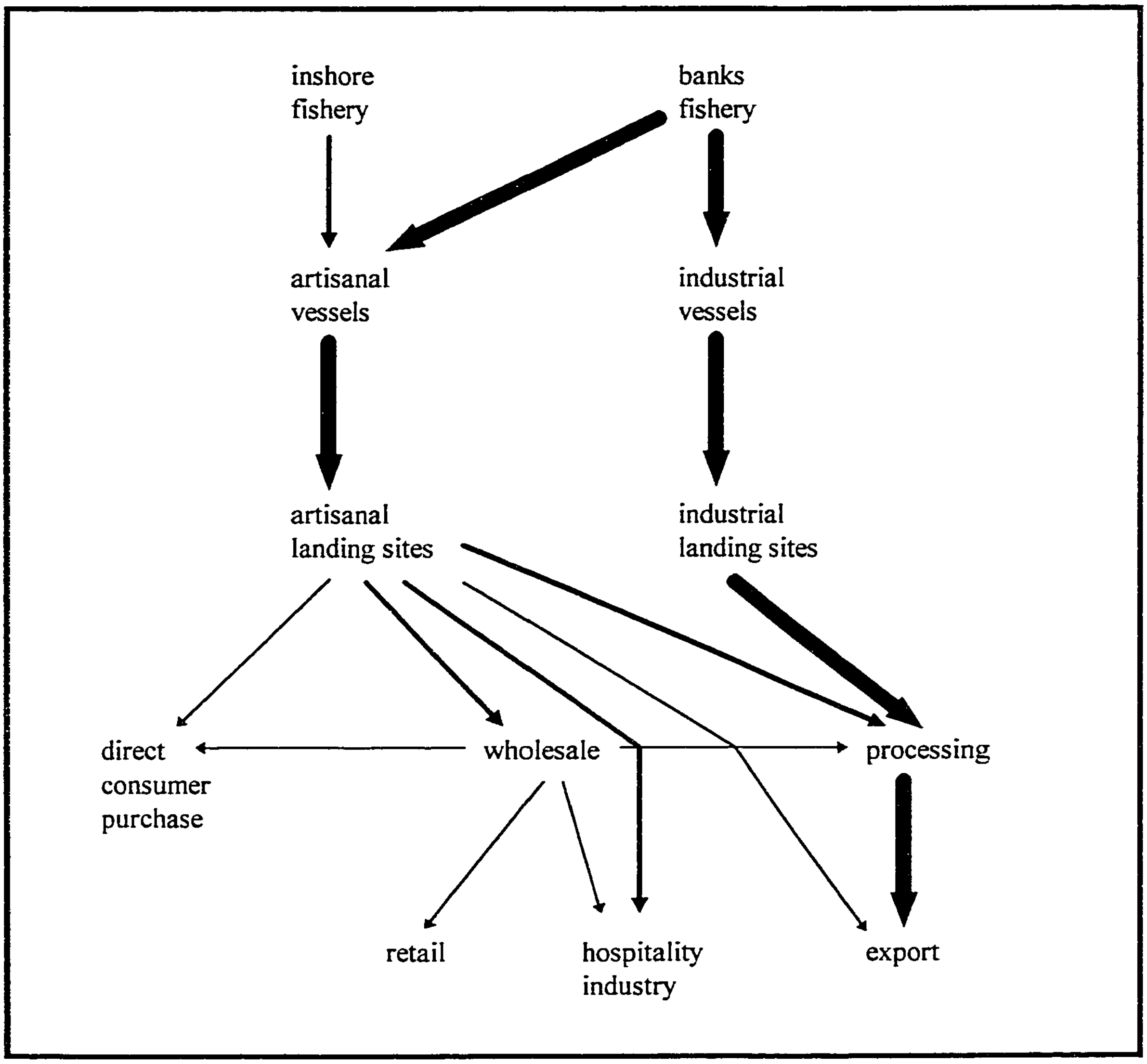

Figure 2.8. Economic source and destinations typical of the lobster fisheries of Jamaica. Light-weight lines represent <10% of biomass flows, medium-weight lines represent 10-30% of flows, and heavy-weight lines represent >30% of flows (source: unpublished notes of S. Grant, Fisheries Division, Government of Jamaica).

The Inshore Reef Fisheries of Jamaica

As of 1997, there were approximately 167 known landing sites with 2,962 vessels on the mainland of Jamaica from which the shelf and proximal bank fisheries are based (vessel statistics excluding carrier vessels or packer boats; Fisheries Division, Government of Jamaica, unpublished data). This is in comparison to the 184 known sites with a total of 3,760 vessels noted in the last government survey in 1981 (Sahney 1982a, 1982b). Figure 2.9 shows the current distribution of landing sites by parish.

As of late December 1998 there were approximately 10,733 registered fishers (boat owners, captains, crew members, and divers) operating from the mainland landing sites (Registration of Commercial Fishermen Database, Fisheries Division, Government of Jamaica). There is likely a large contingent of unregistered fishers in addition to this (e.g. see Chapter 3 concerning spearfishing activities in the Montego Bay area). Before 1975, there were no official records kept of the number of fisher or their activities. The Fishing Industry Act (1975) of Jamaica and the regulation created under the act made the licensing of all fishers and boats a requirement; unfortunately, the ineffective administration of the licensing program prior to the creation of the Registration of Commercial Fishermen Database in 1995 make estimates from licensing data extremely unreliable (i.e. the inability to track those who left the industry or to note the issuance of multiple licenses to the same individual; there were further indications of a large number of unlicensed fishers continuing to operate in violation of the regulations; Espeut 1992; Espeut and Grant 1990).

Inshore fishers use a variety of fishing methods, targeting coral reef finfish, ocean pelagics, coastal pelagics, deepslope finfish, and invertebrates (in particular, conch, lobster and shrimp). Using the unpublished information contained in the initial compilation of the Registration of Commercial Fishermen Database for 1995, the proportion of fishers are summarised according to: (i) target fish type (Table 2.1); and, (ii) method of fishing (Table 2.2). In general, specific methods of fishing target certain species groups - for example, traps, hand lines and spearguns are used primarily to target coral reef finfish, troll lines and rod and reel are used primarily to target ocean pelagics, and China or Trammel nets are used primarily to target coastal pelagics. It must be noted that fishing by more than one method is quite common among inshore fishers, usually combining the use of traps with other more "active" fishing methods (e.g. hook and line, trolling, etc.) while the traps are left to soak (Clemetson 1992; Espeut 1992; Espeut and Grant 1990). The statistics shown in Tables 2.1 and 2.2 only reflect the primary means of fishing as indicated by the fishers during registration.

Table 2.1. Proportion of fishers by target fish type as registered for Jamaican landing sites (source: derived from Registration of Commercial Fishermen Database 1995, Fisheries Division, Government of Jamaica, unpublished).

target species group	proportion of total	target species group	proportion of total
reef finfish	0.375	lobster	0.069
coastal pelagics	0.189	shrimp	0.020
ocean pelagics	0.164	conch	0.015
deepslope finfish	0.153	other[a]	0.015

[a]Includes crab, oysters, welks, and bait fish.

Table 2.2. Proportion of fishers by primary method used as registered for Jamaican landing sites (source: derived from Registration of Commercial Fishermen Database 1995, Fisheries Division, Government of Jamaica, unpublished).

primary method	proportion of total	primary method	proportion of total
trap	0.289	trammel net	0.038
hand line	0.234	scuba/ free lung	0.036
troll line	0.136	speargun	0.033
China net	0.079	rod and reel	0.026
drop line	0.071	other[a]	0.058

[a]Includes long line, hookah, palanca, and various types of nets not elsewhere noted.

Comprehensive estimates concerning the total production of Jamaican inshore fisheries have been hampered by the lack of available data. Sample surveys have been conducted on an irregular basis (see Aiken 1993; Williams 1995). Government of Jamaica surveys were conducted in 1962, 1968, 1973, and 1981. Since September of 1995, the Fisheries Division of the Government of Jamaica has been involved in a regular, monthly catch and effort sampling programme of mainland landing sites. This has further allowed for the generation of annual production estimates by the government for the years 1996 and 1997.

Aiken (1993) and Aiken and Koslow (1992) summarised the available figures for estimated total landings prior to the implementation of the Fisheries Division Catch and Effort Data Collection Programme. The results are shown in Table 2.3. The average for the years reported was found to be 7,600 t (SD = 1,400 t) for the inshore fishery, with the offshore fishery contributing an estimated additional 2,000 t annually (Aiken 1993). It must be noted, however, that the sampling and estimation methodologies are reported to vary considerably between sample surveys; thus, comparisons of data between surveys must be interpreted with caution. In particular, the high estimates for the years 1956 to 1962 have been criticised for employing a sampling methodology which may have significantly upwardly biased the results (Munro and Thompson 1983).

Table 2.3. Estimates of production for Jamaican inshore fisheries (source: Aiken 1993; Aiken and Koslow 1992 as summarised from various sources).

year	estimated inshore fisheries production ($t\ yr^{-1}$)
1945 - 1949	5,450
1950 - 1954	4,990
1955	6,580
1956	7,720
1958	10,260
1959	9,900
1960	10,300
1962	10,990
1968	6,630
1970	6,260
1971	7,080
1973	7,300
1975-1978	7,300
1981	7,220

Using the data from government sample surveys of the fishing industry for the years 1968, 1973, and 1981, Haughton (1988) attempted to improve on the estimated total production of the Jamaican shelf and proximal banks fisheries by introducing a correction factor to compensate for broad differences in the fishing capabilities of vessels, as well as for the proportion of mainland fish landings which were not shelf-based fisheries. This involved extrapolating the sample survey results while considering the total number of mechanised versus unmechanised vessels in the survey compared with the whole of the inshore fishery to adjust for differences in capacity, and deducting an estimate of the ocean pelagic catch. The following notable results were obtained:

- total yield had increased from an estimated 5,840 t in 1968, to 6,611 t in 1973, to 6,757 t in 1981, yet the number of "effective" (mechanised-equivalent) canoes fishing the shelf had similarly increased, from approximately 0.33 canoes km^{-2} in 1968, to 0.46 canoes km^{-2} in 1973, to 0.65 canoes km^{-2} in 1981; and,

- the catch per mechanised canoe had steadily declined, from an estimated 4.23 t $canoe^{-1}$ yr^{-1} in 1968, to 3.46 t $canoe^{-1}$ yr^{-1} in 1973, to 2.49 t $canoe^{-1}$ yr^{-1} in 1981.

Thus, while total inshore fisheries production has apparently shown slight increases from 1968 through 1981, an increase in the fishing effort (as measured by the number of canoes involved in the fishery) has reduced fishing efficiencies (see also Aiken 1985b). The effect of more complex extrapolation adjustments from the sample survey data to account for other structural features of the fleet, such as accounting for different fishing methods or technologies employed, have not been explored. The results of Haughton (1988) demonstrate the underlying rough nature of the production estimates as reported in Table 2.3, and their sensitivity to the chosen extrapolation technique in addition to known sensitivities to the surveying methodology employed.

From the Catch and Effort Data Collection Programme, the Fisheries Division (Government of Jamaica) estimated the total 1996 inshore reef fish catch for artisanal vessels to be 7,590 t, with total production from all fisheries to be 14,496 t (Fisheries Division 1997). In 1997, only an estimated 3,923 t of reef fishes were captured, with a total production from all fisheries of 7,747 t (Fisheries Division, Government of Jamaica, unpublished). These results are obtained through simple extrapolation of the mainland landing site sample catch statistics for each year, considering the proportion of actively fishing boats noted during survey days and the total number of boats registered for all mainland landing sites.

In general, the south shelf fisheries are known to produce much larger catches than the north shelf, believed to be primarily due to the greater shelf area available (3,562 km^2 versus 608 km^2; Espeut 1992; Espeut and Grant 1990; Haughton 1988) and the more prominent use of motorised vessels in the south which increases the fishing capabilities of the vessels (Sahney 1982a). However, average fishery productivity for a given unit of shelf area is believed to be significantly greater on the north shelf (Munro 1983d).

Estimates from Haughton (1988) for the total 1981 landings for the inland and proximal bank fisheries are 5,475 t on the south coast and 1,282 t on the north (a similar pattern is evident for data available for 1968). Canoes are also much more heavily concentrated in the north, with an estimated 1.55 mechanised-equivalent canoes km^{-2} working the north shelf and 0.50 mechanised-equivalent canoes km^{-2} working the south in 1981 (again, a similar pattern is evident for the year 1968; Haughton 1988). However, the different fishing capabilities of north versus south coast vessels, such as the use of more fishing equipment per vessel in the south, may account for much of the differences in the regional estimates as Haughton (1988) could not take this factor into account (e.g. south coast trap fishers are known to set more traps per canoe; Nicholson and Hartsuijker 1982).

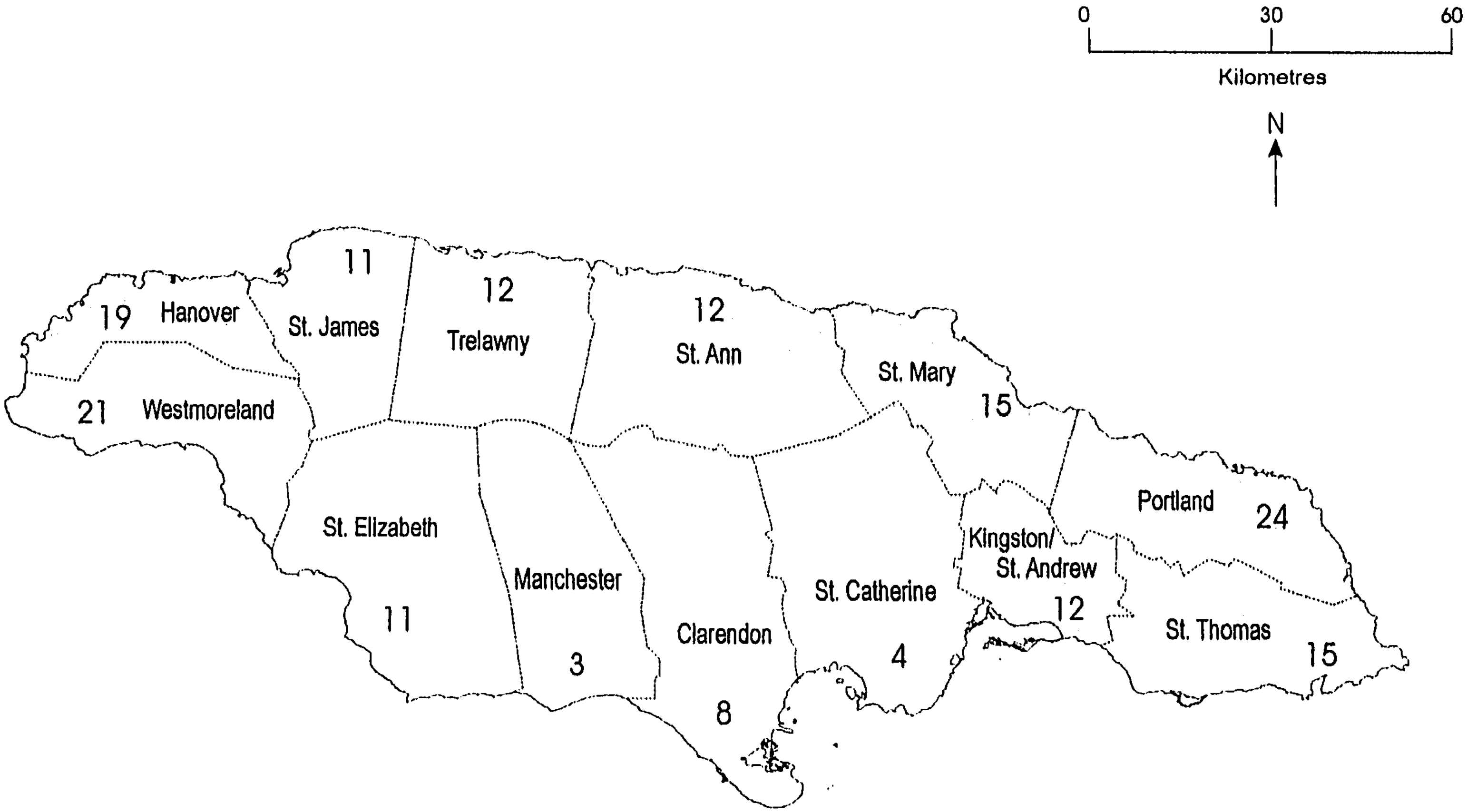

Figure 2.9. Number of landing sites for mainland Jamaica by parish as of October 1998 (source: Fisheries Division, Government of Jamaica).

Estimates of the Potential Yields from Jamaican Inshore Fisheries

Catch and effort data from sample surveys can be utilised to provide estimates of the potential physical yields from Jamaican fisheries through the application of area-based surplus production models (Fox 1970). Results of such applications are to be found in Munro and Thompson (1983), Haughton (1988), and Koslow *et al.* (1994; also in Aiken and Koslow 1992; Clemetson 1992).

Based on a re-analysis of the government data for the years 1968, 1973, and 1981, Haughton (1988) estimated that a maximum total yield (excluding ocean pelagics) of 2.2 t km^{-2} yr^{-1} may be possible for the inshore fisheries, with an associated fishing intensity of 1.0 to 1.2 mechanised canoes km^{-2} of shelf area. This translates to approximately 9,200 t yr^{-1} assuming a mainland shelf and proximal bank area of 4,170 km^2 (as reported in Aiken 1993 and Haughton 1988, using a 100 fathom isobath; alternatively, an estimate of 7,500 t yr^{-1} is derived, assuming a total shelf and proximal bank area of 3,421 km^2 using a 200 m isobath as reported in Munro and Thompson 1983).

The results of Haughton (1988) improved on the estimates of Munro and Thompson (1983), who estimated a maximum potential yield of 4.1 t km^{-2} yr^{-1}, with a fishing intensity of approximately 3.2 canoes km^{-2}, by making a broad adjustment to account for variations in fishing vessel capabilities. However, again the available data did not allow for the estimates to compensate for different fishing capabilities by method of fishing or equipment used. Furthermore, the estimations of the fishing intensities (i.e. number of canoes km^{-2}) corresponding to the maximum potential yield reported here do not take into account possible gains in fishing efficiencies that may be realised by adjusting fishing practices (e.g. see discussion concerning the operation of traps in Munro 1983c).

As noted by Aiken and Koslow (1992) and Koslow *et al.* (1994), the use of area-based surplus production models from catch and effort data, as applied by Munro and Thompson (1983) and Haughton (1988), assumes the following:

- habitats and the biological productivity of fishing grounds are not spatially variable;
- the technical or physical characteristics of fishing do not vary between fishing areas;

- the relationship between catch and effort for all sampled areas is at equilibrium; and,
- the fish stock subjected to the fishing effort is effectively contained in the designated area (i.e. no variations in migration or recruitment rates across management boundaries).

There may be further problems with application of the model if the chosen measure of fishing effort is not linearly related to *in situ* fish abundance (Aiken and Koslow 1992; Munro 1983c; see below). More generally, the effect of fishing on the dynamics and relative abundance of species in highly complex coral reef communities can not be expected to be accurately estimated through an assumed all-encompassing simple logistic relationship that underlies the models (May *et al.* 1979; Munro and Williams 1985). Yet, the models have been relied upon to provide approximations of maximum sustainable yield for the lack of better data which would allow for the use of more sophisticated models (for a review of potential alternative methods available, see Appeldoorn 1996; Die and Caddy 1997).

Both the studies of Haughton (1988) and Munro and Thompson (1983) may be criticised for potentially violating all of the assumptions listed above. For example, spatial differences in ecological productivity may have a profound effect on the potential biological yield estimates. As noted by Nicholson and Hartsuijker (1982), substrate type can have a significant effect on the distribution of demersal fish stocks. Exclusion in the estimates of potentially less productive areas than those sampled would be justified both on a broader regional basis and on a larger spatial scale, such as the differences which may be evident between reef and sandy benthic zones. Total productivity estimates based on the simple extrapolation of sample catch statistics to the whole shelf area may thus be biased.

Koslow *et al.* (1994; see also Aiken and Koslow 1992; Clemetson 1992) attempted to improve on the previous productivity estimates (in particular by trying to control for the productive area underlying each sampled fishery and the heterogeneity of the fishery as indicated by the species fished) but were unable to derive a significant catch-and-effort relationship for Jamaica. This may be due to persistent violations of the model assumptions as outlined above, such as species migration or the mixing of stocks, heterogeneity of the technical characteristics of the fisheries between areas, or disequilibrium between catch and effort (Koslow *et al.* 1994). Koslow *et al.*

(1994) suggest that it is the disequilibrium evident in the Jamaican fisheries as demonstrated by the declining catch rates over time that have likely led to significant upward bias in potential yield estimates using surplus production models.

Targeted sample fishing surveys, with the specific intent of assessing the potential biological yield for Jamaican fisheries development, have been periodic (see Aiken 1985a, 1985b, 1993; Munro 1983a; Nicholson and Hartsuijker 1982; Oswald 1963). Most notable for inshore fisheries are a 1969-1973 study reported in Munro (1983a) and a joint USSR-Jamaica Fisheries Research Project (1979-1980).

From data collected by trawl of demersal fish stocks on the south Jamaican shelf and the Pedro Bank, the USSR-Jamaica survey estimated total potential biological yields of 6,800 t yr^{-1} from the south shelf alone (Aiken 1985b). It has been suggested, however, that these results are an overestimate (Aiken 1985b). Munro (1983d), utilising a Beverton-Holt yield model (Beverton and Holt 1957) applied to trap fishery survey results, examined changes in yields which may result from changes in the mesh size and the fishing effort. Extrapolating from these results and utilising a surplus production model developed from the 1968 government sample survey data, the potential production of the inshore fisheries (mainland shelf and proximal banks, excluding ocean pelagics) was estimated to be from between 7,300 t yr^{-1} to 8,500 t yr^{-1}. The estimates of Munro (1983d) have also been criticised as being overly optimistic (Nicholson and Hartsuijker 1982).

All potential production estimates discussed only address potential biological yields (i.e. maximum sustainable yield) and ignore the economics involved, both in terms of costs associated with fishing and the prices that can be obtained for the catch. As variations in economic costs with fishing effort are considered and social policy objectives reflected in the management regime, desirable optimal fishing efforts will undoubtedly shift (see discussion to follow). There are also potential dynamic shifts in price or cost schedules which may have a profound effect on the desired level of effort. For example, decreases in the quality of the fish caught in Jamaican fisheries have been documented (e.g. Aiken and Haughton 1987; Clemetson 1992; Haughton 1988; Koslow *et al.* 1989; Munro 1983a; note that "quality" fish typically include fishes from the families Lutjanidae (snappers), Serranidae (groupers), and Mullidae (goatfishes), lower quality "common" fish include

Scaridae (parrotfishes), Haemulidae (grunts), and Carangidae (jacks), while "trash" fish include small Scaridae, Acanthuridae (surgeonfishes), Holocentridae (squirrelfishes), and Balistidae (triggerfishes) - see Aiken and Koslow 1992). This may result in a dramatic reduction in economic returns with increases in fishing effort.

Fisheries Economics

The general principles of fisheries economics have been summarised by Anderson (1977), Clark (1985, 1990), and Munro and Scott (1985). In order to establish the context for the methodology selected to derive models of fisheries production in Jamaican artisanal fisheries, the essential elements of bioeconomics are described, focusing on the elementary economic principles as presented by Gordon (1954) and Schaefer (1957). Brief comments are then made regarding limitations and some possible existing extensions from the primary bioeconomic model. In particular, elementary extensions as they relate to the use of capital theory and the application of multi-species or mixed species fisheries models are outlined as these are the most relevant to the present study. The application of a bioeconomic model, it is argued, is inappropriate here.

The Gordon-Schaefer Bioeconomic Model - the Case of an Isolated Single-Species Fish Stock

The bioeconomic (or bionomic) model originally envisioned by Gordon (1954) and Schaefer (1957) assumes the exploitation of a single species fish stock in which the rate of renewal of the fished stock is dependent on the population level of that stock. The model examines the long-run equilibrium relationships between indefinitely sustainable harvest levels and fish population levels, the value of the harvests, and the costs associated with the fishing effort. The physical environment as it affects fish population levels through the rate of natural increase is, on average, assumed invariant. Catch per unit of effort is proportional to the biomass level of the fished population, the fished population is uniformly distributed, fishing gear can not be saturated, and there is no congestion of fishing vessels. The model further assumes that there are insignificant interactions or dependencies on other fisheries stocks, such that the bioeconomics can be analysed in isolation of other fisheries. The demand for fish and the supply of fishing effort (through labour and capital) are perfectly elastic, and there are assumed to be no externalities associated with either costs or benefits (Clark 1985, 1990; Munro and Scott 1985).

The "fishable" biomass of the stock is the result of a balance between recruitment, growth, natural mortality, and fishing mortality (or harvesting). Any changes in the target fish population biomass is thus described by

$$\frac{dN}{dt} = \frac{dz}{dt} + \frac{dg}{dt} - \frac{dm}{dt} - \frac{dH}{dt} + \eta \qquad (2.1)$$

where N = fishable biomass;
z = recruitment;
g = growth;
m = natural mortality;
H = fishing mortality (harvesting); and,
η = randomness (with mean of zero).

Recruitment, growth and natural mortality are assumed to be a function of the biomass (N), while harvesting is assumed to be a function of both the biomass (N) and the fishing effort (E):

$$\frac{dN}{dt} = z\{N\} + g\{N\} - m\{N\} - H\{N, E\} + \eta \qquad (2.2)$$

The fish population biomass and the level of fishing effort may vary over time, although the original analysis of Gordon (1954) and Schaefer (1957) assumed the static case. At any point in time, an equilibrium (defined as $dN/dt = 0$) at a sustainable rate of harvest is achieved when the fishing harvest rate (as determined by $H\{N,E\}$) is equal to the net natural rate of increase in the population biomass (as determined by the sum of $z\{N\} + g\{N\} - m\{N\}$).

The inability to reasonably and accurately measure or independently estimate population recruitment, growth and natural mortality has severely hampered the determination of appropriate biological yields based on the model shown by equation 2.2. There are also further practical problems with the inability of fishing efforts to be cohort-specific within one species (Munro and Scott 1985). In order to help simplify the biological model for practical application, the terms defining the rate of natural population increase in equation 2.2 are combined, and it is assumed that one rate can be applied to the population as a whole. Schaefer (1957) approximated the rate of natural increase as

$$r\{N\} = k_1 N(K - N) \tag{2.3}$$

where k_1 = a constant;

N = population biomass; and,

K = the carrying capacity of the environment in terms of population biomass.

The harvest production function is taken to be a simple linear function of population biomass and fishing effort:

$$H\{N, E\} = k_2 NE \tag{2.4}$$

where k_2 = a constant.

Equating the harvest rate $H\{N,E\}$ with the rate of natural increase $r\{N\}$ (equations 2.3 and 2.4) and re-arranging the terms, we arrive at a production function describing the harvest rate as it relates to a simplified logistic model of fish population levels:

$$H = k_2 E(K - \frac{k_2}{k_1} E) \tag{2.5}$$

If we further assume that the price received per unit for the catch remains constant, then the production function also describes the value of the catch (where the value of the catch is equal to the biomass of the harvest multiplied by a constant price factor); the price may be expected to vary with harvest levels (depending on the ability of the harvest to influence market prices or the timing of other exogenous price-influencing factors) but as long as the price elasticity of demand remains greater than unity, the function describing the relationship between fishing effort and harvest levels will retain the same general shape and the level of effort corresponding to maximum production value will not vary (Schaefer 1957).

The biological production function described by equation 2.5 must next be considered in conjunction with the economic costs associated with harvesting. The level of fishing effort which corresponds with the maximum production value of the harvest does not alone allow one to examine the *net* economic benefits associated with the fishery in question. As noted by Gordon (1954), "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." Assuming that the costs are in direct proportion to the level of fishing effort, and that the level of fishing effort does not affect the prices of labour and capital as factors of production (aggregated and sufficiently indexed by the chosen measure of fishing effort) as these are determined in the larger economy as a whole, then the following is achieved:

$$C = \alpha E \tag{2.6}$$

where C = cost of fishing effort; and,

α = a constant.

Maximum net economic yield (MEY) is of course obtained where the marginal costs (through equation 2.6) are equated with the value of the marginal product (through equation 2.5), or where $dC/dE = dH/dE$ (assuming a constant unit product price):

$$\alpha = k_2(K - 2\frac{k_2}{k_1}E) \tag{2.7}$$

Equation 2.7 thus defines the economic optimum degree of utilisation of the fishery for any point in time. As outlined by Gordon (1954), a renewable resource with an open access management regime will be "overexploited" to the point where all rent has dissipated - that is, where total costs of harvest equal total revenues generated. This is a direct result of the individual fishers making harvest decisions based on the value of the *average product* in relation to *average costs*, rather than on the value of the *marginal product* - each fisher will continue to fish until the average total product just equals the average total costs (where costs include the actual costs associated with fishing, such as fuel, gear and supplies, the interest and depreciation on capital, and the opportunity income of the fishers; Clark 1985; Gordon 1954). The Gordon-Schaefer model, well explored, criticised and expanded upon in the literature over its forty year history, describes the fundamental bioeconomic features behind the essential economic difficulties encountered in fisheries management.

Limitations to the Gordon-Schaefer Model

Given the "base case" Gordon-Schaefer model, there are numerous limitations which make its practical application in most instances invalid, or at least highly questionable. These limitations are not comprehensively summarised here (e.g. see Clark 1985, 1990; Munro and Scott 1985), but can be derived from the assumptions and structure of the model. In short, the focus on an isolated static fishery by Gordon (1954) and Schaefer (1957) is the source of the limitations. Describing an economic production process involving an ecological resource, which is a multi-species complex system whose behaviour has spatial and temporal dynamics, using comparative statics is not ideal (e.g. see Andersen and Sutinen 1984 for an introduction to stochastic bioeconomic methods). Clark (1977) summarises the model's primary limitations as follows:

- an inability to incorporate fish population development or recovery phases;
- an inability to incorporate exogenous changes to economic and biological parameters;
- an inability to examine optimal harvesting equilibriums using anything other than the implicitly defined zero social discount rate; and,
- an inability to consider issues of stability.

The Gordon-Schaefer model is highly simplified, yet has provided a solid basis for understanding the economics of fisheries and the expected results of management efforts. As is the case with any model, specific assumptions may be relaxed in order to focus on the effect of specific factors. Model variations may even predict end results which are in radical departure from the essential conclusions to be drawn from the Gordon-Schaefer model. For example, as noted by Munro and Scott (1985), zero or negative net economic benefits may *not* necessarily result in certain fisheries, even with resource mismanagement (e.g. with non-linearities introduced by relaxing the model assumption of perfect elasticity of supply and demand). Despite the use of what may seem to be a highly restrictive, unrealistic set of assumptions, the Gordon-Schaefer model has proved to be invaluable to understanding the economic issues concerning the management of an open-access common property resource.

The Neo-Classical Concept of Fisheries Rent and the Use of Capital Theory in Fisheries Economics

Aside from the potential problem of rent dissipation as discussed above, the second elementary principle from natural resource economics, which may be applied to fisheries, concerns appropriate investments over time and the issue of valuing the stream of benefits to be derived from the resource. Fisheries essentially represent capital investments which will yield a particular return over time, dependent in a large part on the harvest decisions. Examining the biological investment required in the fish stocks to determine appropriate levels of harvest over time requires use of a dynamic model.

Capital theory, in essence, examines the optimal level of investment in and exploitation of the resource as a function of time. Fisheries management is seen, as with any problem in natural resource economics, as a problem concerning capital investment and use (e.g. Clark 1977; Clark and Munro 1975). As is the case with the use of any dynamic model in fisheries management, a need to move towards models which could better capture the important aspects of the interactions between fisher and fish, as well as the responses of the fish populations, fisher investment decisions, and fishing behaviours, is recognised (Munro and Scott 1985).

In order to illustrate the application of capital theory to fisheries, Munro and Scott (1985; see also Clark 1977, 1985; Clark and Munro 1975) present a simple deterministic model that maintains the basic assumptions of the Gordon-Schaefer model. Specifically, the same general production function is maintained, and the demand for fish and the supply of fishing effort (through labour and capital) are assumed perfectly elastic. In their model, they further assume that the price of fish, the unit costs of fishing effort, and the social discount rate are all independent of time. The fish population biomass, $N\{t\}$, is controlled through the harvest rate, $H\{t\}$, such that the net present value (NPV) is maximised. Resource rent earned at any given point in time can be expressed as

$$\prod\{N,H\} = (p - c\{N\})H \tag{2.8}$$

where p = unit price of fish;

c = unit costs of harvesting stated as a function of N; and,

H = rate of harvest.

(unit costs of harvesting can be derived by substituting the harvest production function as shown by equation 2.4 within the total fishing effort cost equation as shown by equation 2.6, and removing H to reflect unit harvest costs). The objective function is

$$NPV = \int_0^\infty e^{-\delta t} \prod(N\{t\}, H\{t\})dt \qquad (2.9)$$

where δ = the social discount rate.

The state equation is taken from the basic Gordon-Schaefer model as

$$\frac{dN}{dt} = r\{N\} - H\{t\} \qquad (2.10)$$

with the constraints of

$$N\{t\} \geq 0 \qquad 0 \leq H\{t\} \leq H_{max} \qquad (2.11)$$

The optimal equilibrium biomass (N^*) then becomes

$$r'\{N^*\} - \frac{c'\{N^*\}r\{N^*\}}{p - c\{N^*\}} = \delta \qquad (2.12)$$

It implicitly follows that given $\delta > 0$, the optimal level of harvest will be at a higher level of harvest than with the static Gordon-Schaefer model which assumes a zero social discount rate. Thus, investment in the fisheries stock resource below that required for maximising economic rent isolated at any one point in time may be called for, simply because a future harvest is valued less than a current one.

Of course, this base capital theory model has many possible extensions, such as the exploration of various exogenous functions and different social discount rates. However, as presented it illustrates the essential neo-classical treatment of natural resource rents as applied to fisheries.

Multi-Species Models and Mixed Species Fisheries

The coral reef fisheries addressed in this study can be characterised as highly diverse mixed species fisheries (that is, many species are typically caught for any given single fishing effort event). Different species or stocks of fishes may also be inter-related biologically, not simply technologically due to the nature of the fishing methodologies. Modelling of such systems requires consideration of the multi-species dynamics. Bioeconomics have explored ways in which adequate descriptions of such systems may be provided. For examples of relatively simple inter-related fisheries models, see Agnello and Anderson (1977) and Strand and Hueth (1977).

As soon as one considers the quantitative bioeconomic modelling of multi-species or mixed species fisheries, and their associated complex ecological relationships, the problem becomes extremely complex and generally unmanageable. It is possible to explore general statements of elementary functions describing the interaction of two fished populations or describing the related issues (e.g. see Clark 1985, 1990), but detailed quantitative modelling, particularly involving more than two species, has not proven fruitful. As one can appreciate, the general assumptions necessary for a single-isolated species model are compounded and become untenable in the multi-species case. As noted by Agnello and Anderson (1977), assumptions concerning equilibrium, growth, inter-temporal relationships and lags, and spatial aggregations are necessary for bioeconomic modelling. For example, increases in fishing effort within a multi-species fishery (e.g. coral reef fisheries) is expected to result in a decrease in natural mortality rates due to a reduction in the number of predators and competitors of any given species - this may result in physical yields from fishing reaching a maximum at a higher rate of effort (Munro 1983e). The consideration of dynamic complex systems models becomes a necessity.

Complex simulation modelling (e.g. McClanahan 1995), particularly through the use of qualitative information in conjunction with quantitative information, may hold particular promise. However, the successful integration of a complex multi-species or ecosystem simulation model as part of a bioeconomic fisheries production model remains for future research endeavours. The recent development of ecosystem steady-state mass-balance models also hold some promise (e.g. the ECOPATH series of models; Christensen and Pauly 1992), yet economic considerations have been non-existent or rudimentary at best.

Economic Fundamentals of Production Function Models

The Chosen Functional Form

The fishery production function which shows the relationship between the harvest rate and a simplified fish population model, as described by equation 2.5, is fundamentally different from many more generally utilised neo-classical economic production functions due to the defined relationship between the inputs and the production output. Specifically, many economic production functions are concerned with describing either constant or increasing returns and, after a point, diminishing returns while some maximal asymptote is approached. Equation 2.5 describes diminishing returns to fishing effort over the whole range of the production function (at $N/2$ marginal returns become zero and thereafter are negative until total returns reach zero at N=0). This is a fundamental result of harvest being a function of both the fishing effort and the fish population biomass (equation 2.4).

What is further unique in the case of a fishery, as noted by Anderson (1976), is that the fisher (producer) does not have direct control over her or his output - the amount of fish harvested. Fishers only have direct control over the amount of fishing effort, which in turn interacts with the fish population level and the overall level of fishing effort exerted by other fishers to determine the amount of catch (equation 2.4). Similar to a firm in a competitive market being a price taker, a fisher in a competitive fishery can be viewed as a "catch rate taker", having to accept the average rate of catch per unit of effort, assuming relatively homogeneous fishing vessel technologies. This has ramifications for the specification of fisheries production at the level of the firm (e.g. see Anderson 1976).

Fisheries bioeconomics has traditionally assumed a quadratic biological production function (equation 2.5). The harvest production function (equation 2.4) can be written in the following form (Munro and Scott 1985):

$$H\{t\} = qE^{\alpha} N^{\beta} \tag{2.13}$$

where q = constant;

α = production elasticity associated with fishing effort (E); and,

β = production elasticity associated with the fish population biomass (N).

Equation 2.4 is a special case of a Cobb-Douglas production function model (a generalised power function) in which $\alpha = \beta = 1$. Assuming that equation 2.13 is the correct functional form, any changes in the constant or elasticities over time, across geographical regions, or between production processes reflect structural changes and/or disequilibrium in the chosen production process. It is possible to relax the restrictive assumptions of this highly simplified production function model (most notably the imposition of homogeneity, strong separability, and elasticity of substitution of one) and employ other general models.

In selecting the algebraic form of the production function, it is necessary to consider the physical nature of the production processes and the economic properties ultimately of interest. When such information is generally lacking, it is best to select a functional form which is less restrictive. A translog function (transcendental logarithmic function; Christensen *et al.* 1973, 1975) is often assumed in industrial production studies as a general form of a power function with relatively limited restrictions imposed due to the structure of the function itself. The form analogous to equation 2.13 would be

$$\ln C = \ln \alpha_0 + \alpha_E \ln E + \alpha_N \ln N + \frac{1}{2}\beta_{E,N} \ln E \ln N \qquad (2.14)$$

Equation 2.14, however, requires the use of more sophisticated statistical analyses to estimate. Modelling using such a functional form can easily become overly complex to solve if anything but the simplest underlying ecological relationships are assumed.

The Selection of Inputs and the Measurement of Fishing Effort

In all models presented thus far, we have assumed harvest rates to be a function of fishing effort. How does one measure fishing effort? In strict adherence to standard fisheries economics practice, fishing effort is defined as an index of the capital and labour inputs to the production process. The particular form of the index should be selected based on relevance to the fishery in question. Clark

(1985), in reviewing possible operational definitions of fishing effort, arrives at a general definition for the instantaneous rate of catch as it relates to fishing effort:

$$C = \varepsilon a E \rho \tag{2.15}$$

where C = instantaneous catch rate (mass per unit time, or H as specified previously by equation 2.4);

ε = gear selectivity factor, or the proportion of fish in a given volume of water captured by the fishing gear (dimensionless factor, where $0 \leq \varepsilon \geq 1$);

a = constant representing the volume of water fished per time period for a given standardised vessel-gear (volume per unit time);

E = nominal fishing effort, or the number of standardised vessel-gear units actively fishing for a given time (standardised vessel-gear units); and,

ρ = density, or concentration of fish in a given volume of water (mass per unit volume).

As indices of fishing effort are normally applied to bioeconomic modelling in which there is an explicit and assumed relationship with fish population dynamics, and it is often desirable or even necessary to be able to deduce fish population levels from fisheries catch and effort data, the above relationship ultimately applies (i.e. fishing effort as it is related to the rate of catch is not seen to occur separate and apart from the population of fish). This can alternatively be stated as due to a general focus of fisheries economists on the specification of the relevant fish population-linked production function (i.e. through equation 2.5 or a functional equivalent) rather than a focus on other production issues (e.g. see McGaw 1981 for an alternate analysis of endogenously determined fishing effort from the perspective of the supply behaviour of the industry).

Fishing effort can be generally defined as the rate of screening of the fishing ground (Clark 1985). This definition emphasises the overarching characteristic of most measures of effort - there is a direct proportionality between the catch rate per unit of effort and the *in situ* density or abundance of the fish stock (Clark 1985). Following this line of reasoning, any measure of effort should seek to maintain this proportionality *for a given fishing unit for a given point in time.* The specific measure of effort chosen must also accurately reflect the capital and labour inputs into the production process.

This means that the measure of effort employed must accurately reflect the physical realities associated with the mechanism of fishing - that is, the availability of the fish. For example, in a typical Antillean-type trap the number of fish retained will approach an asymptote over time as rates of ingress become balanced with escapement (Munro 1983b; Nicholson and Hartsuijker 1982). In other words, the bioeconomic modelling of trap fisheries can not use an effort index based directly on the amount of time the traps are soaked. Munro (1983b) described the relationship as

$$A = \frac{pC_s}{r\left(1 - e^{-Rs}\right)} \tag{2.16}$$

where A = availability value;
C_s = catch taken after a trap soak of s days;
p = probability of escape;
r = rate of retention; and,
R = coefficient of the rate of retention.

As noted by Clark (1985), based on equation 2.15 there are difficulties associated with aggregating catch data over time and across space, particularly since fish densities are not likely to be constant everywhere. Equation 2.4 can be specifically related to equation 2.15 through a "catchability coefficient" (Clark 1985):

$$k_2 = \frac{a\varepsilon\rho}{N} \tag{2.17}$$

where k_2 = constant as defined in equation 2.4; and,
N = fish population biomass as previously defined.

Importantly, it is now evident that the coefficient (k_2) from equation 2.4 is necessarily a function of the population biomass (N) and the fish population density (ρ). Assuming that the population density is some function of the population biomass, then the catch per unit effort (or harvest rate, *H*) is necessarily directly proportional to the marginal stock density (Clark 1985). The total harvest function as defined in the Gordon-Schaefer model (equation 2.4) assumes that the gear selectivity factor (ε) and the concentration of fishes (ρ) is constant throughout the fishing area in question. The assumption of equal fish population density throughout the fishing area is physically unrealistic but may be defensible if one assumes that fishers will continually seek out areas with the highest fish densities; this in effect leads to equal densities for the population fished at any one point in time (Clark 1985). The model thus also assumes the specific case where the catch per unit effort is directly proportional to the population biomass and that there is then a direct and linear relationship between population biomass and population density. As outlined by Clark (1985), particular modelling challenges appear where there is not a direct proportionality between density and biomass (e.g. pelagic schooling fishes).

Using a concept of effort as traditionally defined, it is thus necessary to consider the possible relationship between fish population biomass and fish population density for the fishery in question before one can apply with any confidence an index in a fisheries production function that is related back to the dynamics of the fished population. The specification of the functional form relating fishing effort to harvest is key to the success of bioeconomic modelling efforts. There is no one definition of fishing effort which is appropriate in all or even most circumstances, as it will depend on the type of gear and equipment used, the nature of the fishing method, and the characteristics of the fished population.

For practical application, fishing effort is usually an aggregate of the time spent fishing weighted by the fishing power of the vessel in question to adjust for productivity differences. More specifically (Anderson 1978),

$$E = \sum n_i p_i e_i \tag{2.18}$$

where n_i = number of boats in the *i*th class;

p_i = fishing power coefficient of boats in the *i*th class; and,

e_i = number of days fished (or traps set, etc.) per boat in the *i*th class.

In addition to the above features imposed on the measure of fishing effort due to the bioeconomic basis in the model, there are additional elementary economic principles regarding the creation of indices (such as fishing effort) through the aggregation of factors of production within a production function that should be considered. As noted by Squires (1987), these principles are often ignored or assumed unimportant in fisheries economics.

Separability in inputs (either weak or strong separability depending on the chosen form of the production function) is a necessary condition for the aggregation of the factors of production into a single index within a production function (Berndt and Christensen 1973). It further follows that homotheticity of the function using an aggregated measure of fishing effort is a necessary and sufficient condition if the analysis is to involve optimisation (Squires 1987). Considering the typical neo-classical production model with labour and capital as separate inputs, the aggregation of capital assumes that the marginal rate of technical substitution (MRTS) of one form of capital for another, where both (or multiple) forms of capital are measured by one index number, is independent of the amount of labour in use (Solow 1955). Similarly, the aggregation of differing forms of labour into a single index number assumes intra-index factor substitution independence from the amount or mix of capital in use. As it specifically applies to an aggregated index of effort and the production function shown by equation 2.5, the MRTS between capital and labour within the index of effort (or any other pair of inputs deemed relevant and necessary for incorporation into the effort index) must be independent of the level of the fish stocks and any changes in the level of stocks (Huang and Lee 1976).

More generally, to meet the condition of weak separability, any changes in the relative proportions of the components of any one index number (MRTS) within a production function must be independent of any changes in the levels any other production function variables not within that index (Berndt and Christensen 1973). Weak separability is a necessary and sufficient condition for the use of input indices within production functions in the form of $F\{X\}=F\{X_1, X_2, ..., X_n\}$ where X_n are input indices (Berndt and Christensen 1973, 1975). Whenever this condition does not hold, the validity of the aggregation is in question and thus collapse of production function variables into a set of indices should be avoided. To meet the more restrictive condition of strong separability (necessary when $F\{X\}=F\{X_1+X_2+...+X_n\}$ where X_n are input indices; e.g. the multi-factor CES function), the MRTS between any two components of any two separate index numbers within a production function must be independent of any changes in any other production function variables not within either of those two indices (Berndt and Christensen 1973). In the limited case of a Cobb-Douglas production function for which there are only two input indices or subsets involved, weak separability implies strong separability (Berndt and Christensen 1973).

An additional primary consideration in the selection of inputs and the specification of a function is the separation of disparate and independent production into consistent and meaningful processes. Where multi-species (or multiple output) fisheries are involved, the researcher must ask whether or not a specified set of inputs is required to produce all outputs (joint-in-inputs production; Squires 1987). If joint production does not hold, then a separate production function should be specified for each set of outputs for which there is a reasonably independent set of inputs required. In other words, joint-in-input production implies that associated with one product there are external effects on other outputs and, thus, all necessary inputs should ultimately be consolidated into one consistent production process accounting for the complete set of outputs.

Joint production has long been recognised in economic theory, and arises either through inherently interdependent production processes or through fixed or quasi-fixed factors of production (e.g. see Kurz and Salvadori 1995). As noted by Squires (1987), fisheries managers are also guilty of assuming the opposite case - that the described fisheries is nonjoint-in-inputs. This leads to the erroneous assumption that the fishery can be adequately analysed and managed as a separate production process when in fact other fisheries outputs are necessarily involved.

The Position

The field of fisheries bioeconomics, as it has been defined, has held tightly to the notion of attempting to explicitly model natural fish population dynamics embedded within an economic production model. Such an approach at first only seems logical and necessary given that it is intuitively obvious that production in a fishery must necessarily depend on both the level of fishing effort and the natural supply of fish. However, this approach can only be quantitatively approached or adequately described given the integration of the simplest ecological and economic models. The complexity of many ecosystem-fishery relationships, particularly concerning coral reef fisheries, makes the truly meaningful application of bioeconomics untenable. Moreover, as noted above, in the pursuit of sufficient bioeconomic models many of the elementary fundamentals of microeconomic production function construction have been neglected.

I suggest that advances can be made in the field of fisheries economics and management if one takes an alternate approach and backs away from the complexities associated with trying to develop an integrated, dynamic bioeconomic model and returns to a primary consideration for economic modelling as a description of a process separate, initially at least, from the ecosystem dynamics. This represents a return to an analysis of the production process as an exertion of fishing effort *at the level of the firm* (e.g. see also McGaw 1981; Squires 1987).

It is not the intent of this study to present a bioeconomic model in which production is ultimately linked to natural fish population dynamics, nor to subsequently address questions of maximum net economic yield or the sustainability of current (or alternative) fishing practices. This dissertation is unique in that rather than attempting to address the *availability* of the supply of fish for the production process through a relationship which is assumed to effectively describe the fish population dynamics, it will first address the nature of the production process independent of the unknown availability of fishes. The direct *contribution* of coral reef ecosystems to the value of the final product will be explored separately through the development and use of a biophysical index of captured ecosystem value (Chapter 4).

To summarise, this approach is adopted for the following reasons: (i) to statistically explore the nature of firm-level inputs as they vary by fishery within Jamaica; and, (ii) to derive production functions which describe the application of fishing effort and are valid and adequate descriptions of the production processes.

Methods

Information readily available from published sources describing fisheries in Jamaica (see Introduction) has not sufficiently described either the level and type of fishing activities or the economics involved in fishing to allow for specific economic modelling or the placing of fishing activities in the context of an ecological-economic model. It was necessary, in order to fulfil the objectives of this study, to obtain more detailed information. Unpublished data was solicited from secondary sources in Jamaica, including academic institutions, marine management agencies, and government agencies during field research in the months of January, February, and September 1998. Information was solicited primarily from the two institutions with the greatest local involvement - the Fisheries Division (Ministry of Agriculture, Government of Jamaica), which is mandated with the responsibility of managing fisheries in Jamaica, and the Department of Life Sciences and the Centre for Marine Science at the University of the West Indies at Mona, which is the leading local academic institution concerned with the study of Jamaican fisheries. In addition, regional information was sought from the Discovery Bay Marine Laboratory (the primary marine research facility in Jamaica, operating under the auspices of the University of the West Indies at Mona) and the Montego Bay Marine Park (the oldest marine park in Jamaica, with a history of involvement in fisheries and marine research).

The Nature of Existing Information

The Fisheries Division (Ministry of Agriculture, Government of Jamaica) offered access to their existing databases for this project. The Division currently does not examine or analyse the economics of its fisheries in any consistent or comprehensive way, nor do any academic or other government agency personnel specifically address Jamaican fisheries economics issues on a regular basis. Reports have been prepared which describe the socio-economics of specific fisheries, under the auspices of both academic and government institutions, but these have been infrequent and with a narrow analytical and geographical focus (see Introduction, specifically Aiken 1990; Alleyne 1996; Grant and Blythe 1995; Espeut 1992; Espeut and Grant 1990; Haughton 1988; Nicholson 1994; Nicholson and Hartsuijker 1982).

Table 2.4 shows all information that is currently collected and compiled on Jamaican fisheries and held by the Fisheries Division. One source was primarily utilised for the analysis - the Catch and Effort Data Collection Programme database. The Registration of Commercial Fishermen Database and the fishery flow chart information (a preliminary analysis compiled by Fisheries Division staff, showing the economic relationships between actors in a number of fisheries) and was used to describe the nature of the fisheries (see Introduction) and thus place the results of the catch and effort modelling component within a broader context of fishing activities. The economic costing information which is collected, with the exception of the limited yet targeted sampling for the analysis of the commercial shrimp and conch fisheries, is only currently done so on an opportunistic basis as the information is volunteered by the fishers participating in the catch and effort surveys. The existing information is sporadic and unreliable, and could not be used further for this present analysis.

Table 2.4. Information currently collected and compiled by the Fisheries Division, Government of Jamaica, for Jamaican fisheries.

programme	information	level of collection	frequency of collection	temporal coverage
registration of commercial fishers	• for each registered fisher: fishing beach, role in fishery, number of boats, type of boats, time working in fishery (full-time/ part-time/ no-time), marine area fished, type of marine fish, and gear type.	• collected by fishing beach.	• ongoing.	• initiated 1994 and 1995, with periodic updates depending on external special information requests and registration of new fishers.
Biological Data Collection Programme (CFRAMP)	• wet weights for conch; carapace length for lobster; fork length for herring sprat, dolphinfish, and skipjack tuna.	• 500 sample size for conch; 200 per month for herring sprat, dolphinfish, and skipjack tuna.	• ongoing.	• initiated 1997 (various start dates).
Catch and Effort Data Collection Programme (part of CFRAMP)	• catch (wet weight) by species, effort (crew size, boat type, equipment type, equipment size, and hours fished), and price by boat and landing beach.	• various landing beaches sampled (e.g. 18 in 1996; 14 in 1997) periodically throughout the year.	• ongoing.	• initiated September 1995.

Table 2.4. (continued). Information currently collected and compiled by the Fisheries Division, Government of Jamaica, for Jamaican fisheries.

programme	information	level of collection	frequency of collection	temporal coverage
fishery flow chart	• flows by weight on destination of catch for reef finfish, deepslope finfish, offshore pelagics, coastal pelagics, shrimp, conch, and lobster fisheries.	• variable, depending on the fishery.	• one-time.	• 1996.
economic costing	• fuel expenses, wages, maintenance expenses, food expenses, and prices received for catch (from production figures can estimate gross receipts).	• active collection for shrimp (preliminary economic analysis by Economics Department, Ministry of Agriculture) and conch (see final report by Alleyne 1992); others "opportunistic" on Catch and Effort Data Collection Programme (information recorded when volunteered by fisher during survey).	• one-time for shrimp and conch; ongoing and variable for all fisheries depending on information volunteered by fisher during Catch and Effort Data Collection Programme surveys.	• 1996 for conch; 1997 for shrimp; since 1995 for "opportunistic" information collection.

Estimation of the Fisheries Production Functions

Functions of fishing effort were derived using the catch and effort database for the years 1996 and 1997. Figure 2.10 shows the locations of all landing beaches surveyed during 1996 and/or 1997. Table 2.5 outlines the characteristics of those beaches, as well as the extent of the sampling (excluding industrial conch and lobster surveys). The raw data provided by the Fisheries Division for this analysis has not elsewhere been analysed or reported.

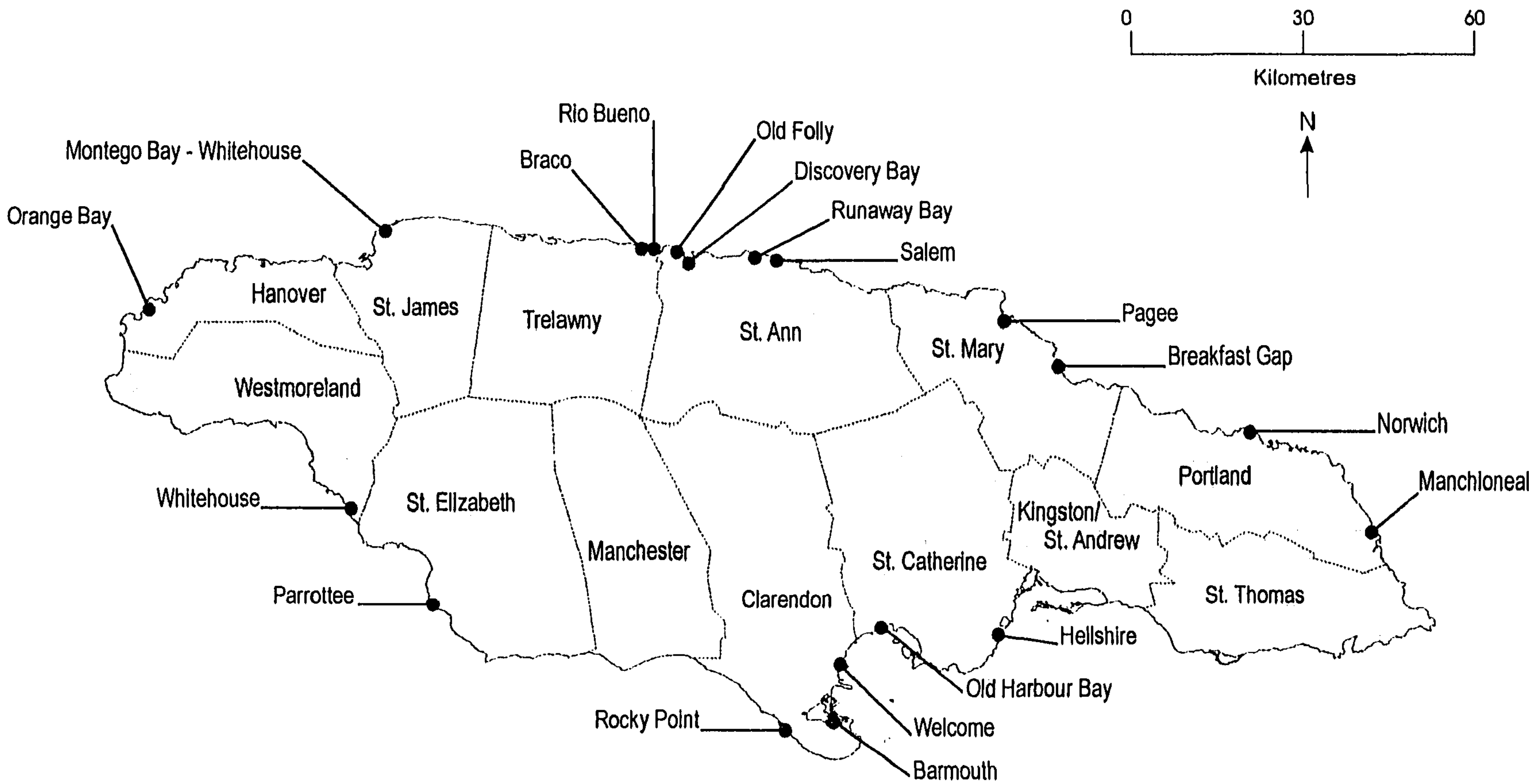

Figure 2.10. Locations of landing sites surveyed for the 1996 and/or 1997 Catch and Effort Data Collection Programme as conducted by the Fisheries Division, Government of Jamaica.

Table 2.5. Characteristics of landing beaches surveyed for the Catch and Effort Data Collection Programme and extent of surveys (excludes industrial conch and lobster surveys).

landing beach	number of boats on beach (1997)	main gear types	main target fish type	total number of catch and effort surveys		total number of months represented in catch and effort surveys	
				1996	1997	1996	1997
Barmouth	34	traps; China net; hand line	reef	47	64	6	6
Breakfast Gap	15	traps; hand line	reef	1	0	1	0
Braco	6	traps; hand line	reef	34	28	8	7
Discovery Bay	24	traps; hand line	reef	65	25	7	6
Hellshire	24	traps; China net; hand line	reef	0	3	0	2
Manchioneal	31	traps	reef (offshore banks)	131	96	12	10
Montego Bay - Whitehouse	15	trolling	pelagics	17	35	6	7
Norwich	4	traps; hand line	reef	2	0	1	0
Old Folly	11	traps; hand line	reef	88	20	11	8
Old Harbour Bay	138	traps; China net; hand line	reef	108	284	6	10

Table 2.5 (continued). Characteristics of landing beaches surveyed for the Catch and Effort Data Collection Programme and extent of surveys (excludes industrial conch and lobster surveys).

landing beach	number of boats on beach (1997)	main gear types	main target fish type	total number of catch and effort surveys		total number of months represented in catch and effort surveys	
				1996	1997	1996	1997
Orange Bay	7	traps; hand line	reef	1	0	1	0
Pagee	33	trolling	pelagics	118	88	10	8
Parrottee	10	traps; China net; hand line	reef	6	0	1	0
Rio Bueno	25	traps; hand line	reef	99	34	11	6
Rocky Point	156	traps; China net; hand line	reef	82	121	7	5
Runaway Bay	15	traps; hand line	reef	37	39	9	9
Salem	16	traps; hand line	reef	120	58	11	9
Welcome	14	traps; China net; hand line	reef	7	0	2	0
Whitehouse	60	traps; China net; hand line	reef	75	22	10	6
total	638	n/a	n/a	1038	917	12	12

Before the production functions may be estimated, it is necessary to: (i) select the necessary and relevant variables; (ii) select the algebraic form of the production functions; and, (iii) select the estimation technique. As stated above, the specification of the fisheries production functions focuses on deriving functions of effort as they relate to catch. From the existing Catch and Effort Data Collection Programme Database (see Table 2.4 and Discussion below), the following relevant variables were selected:

- total catch (weight);
- crew size;
- soak time or time gear fished; and,
- quantity of gear used.

In addition, information on the fishing beach and the date on which individual surveys were conducted was retained to inform the production function estimates. The database (Table 2.5) was audited with respect to the completeness of the records. Specifically, individual records were eliminated if any of the above variables were not recorded. The number of samples eliminated through the audit is shown in Table 2.6. A total of 523 samples were available from 1996, while a total of 664 were available from 1997.

Table 2.6. Number of catch and effort survey samples eliminated through data quality audit and number available for production function modelling.

landing beach	number of samples eliminated		number of samples available for modelling	
	1996	1997	1996	1997
Barmouth	3 (6.4%)	0 (0.0%)	44	64
Breakfast Gap	0 (0.0%)	n/a	1	0
Braco	34 (100.0%)	28 (100.0%)	0	0
Discovery Bay	65 (100.0%)	25 (100.0%)	0	0
Hellshire	n/a	0 (0.0%)	0	3
Manchioneal	21 (16.0%)	9 (9.4%)	110	87
Montego Bay - Whitehouse	0 (0.0%)	1 (2.9%)	17	34
Norwich	0 (0.0%)	n/a	2	0
Old Folly	88 (100.0%)	20 (100.0%)	0	0
Old Harbour Bay	12 (11.1%)	21 (7.4%)	96	263

Table 2.6 (continued). Number of catch and effort survey samples eliminated through data quality audit and number available for production function modelling.

landing beach	number of samples eliminated		number of samples available for modelling	
	1996	1997	1996	1997
Orange Bay	0 (0.0%)	n/a	1	0
Pagee	12 (10.2%)	8 (9.1%)	106	80
Parrottee	0 (0.0%)	n/a	6	0
Rio Bueno	99 (100.0%)	34 (100.0%)	0	0
Rocky Point	16 (19.5%)	4 (3.3%)	66	117
Runaway Bay	37 (100.0%)	39 (100.0%)	0	0
Salem	120 (100.0%)	58 (100.0%)	0	0
Welcome	1 (14.3%)	n/a	6	0
Whitehouse	7 (9.6%)	6 (27.3%)	68	16
total	515 (49.6%)	253 (27.6%)	523	664

The surveys were aggregated according to the use of similar production technologies (i.e. methods of fishing). Table 2.7 lists the total number of samples available for modelling by fishing technology.

Table 2.7. Number of catch and effort survey samples by fishing technology available for production function modelling.

fishing technology	number of samples available for modelling		
	1996	1997	1996 and 1997
trap (fish pot)	205	225	430
troll	128	109	237
speargun/ scuba	23	36	59
hand line	32	17	49
long line	1	4	5
palanca	22	40	62
bottom line	7	5	12
China net (gill net)	88	220	308
trammel net (drag net)	4	2	6
sprat net	4	0	4
beach seine	4	6	10
hookah	1	0	1
trawl	4	0	4

Various models for describing fishing effort were explored, yet all were restricted within two families of production functions:

1. a Cobb-Douglas (C-D) production model, using crew size, quantity of gear used, and soak time as the maximum set of independent variables. The C-D model is described as

$$C = \alpha_0 L^{\alpha_L} G^{\alpha_G} F^{\alpha_F} \tag{2.19}$$

where C = total catch (kg);
L = crew size (labour);
G = number of gear units; and,
F = soak time or fishing time (hours or days).

2. a translog function (transcendental logarithmic function; Christensen *et al.* 1973, 1975; Kim 1992). The base translog assumed for this study took the following form:

$$\begin{aligned} \ln C &= \alpha_0 + \alpha_L \ln L + \alpha_G \ln G + \alpha_F \ln F \\ &+ \frac{1}{2}\beta_{L,G} \ln L \ln G + \frac{1}{2}\beta_{L,F} \ln L \ln F + \frac{1}{2}\beta_{G,F} \ln G \ln F \end{aligned} \tag{2.20}$$

The extended translog form assumed for this study was

$$\begin{aligned} \ln C &= \alpha_0 + \alpha_L \ln L + \alpha_G \ln G + \alpha_F \ln F \\ &+ \frac{1}{2}\beta_{L,G} \ln L \ln G + \frac{1}{2}\beta_{L,F} \ln L \ln F + \frac{1}{2}\beta_{G,F} \ln G \ln F \\ &+ \alpha_R R + \alpha_Y Y + \sum_{j=1}^{11} \alpha_{M_j} M_j \end{aligned} \tag{2.21}$$

where Y = dummy variable for the year 1997;
M_j = dummy variable for the month (j = 1 through 11 represents the months February through December); and,
R = dummy variable for north shelf fisheries: Norwich, Breakfast Gap, Pagee, Salem, Runaway Bay, Discovery Bay, Old Folly, Rio Bueno, Braco, Montego Bay - Whitehouse, and Orange Bay (as opposed to south shelf and banks fisheries).

Limits to the data available for some months and regions (e.g. note that in many instances 100% of the samples from north coast landing sites were eliminated through the data auditing procedure;

Table 2.6) restricted application of the extended translog model (equation 2.21). The translog function, although more complex in its structure, is assumed as a more general form of a power function with relatively fewer restrictions imposed due to the structure of the function itself (i.e. no *a priori* restrictions on the elasticities of substitution or returns to scale). The lack of cost and price information did not allow specification of cost or profit functions, or the use of simultaneous estimation methods; thus, only the physical production processes were examined.

For C-D models, an ordinary least-squares (OLS) linear multiple regression estimation after log transformation of the data was used to derive distinct production functions for all homogeneous sets of fisheries technologies. The number of independent variables incorporated into the C-D production descriptions (equation 2.19) was reduced if an individual term's coefficient was not significant (H_0: $\beta_i = 0$, t-statistic, $\alpha(2) = 0.05$). For translog models, estimation was made via non-linear multiple regression using sequential quadratic programming. Parameter estimates from C-D models were used to inform starting values. Individual terms were eliminated from the base and extended translog models again if the coefficient was not significant (H_0: $\beta_i = 0$, t-statistic, $\alpha(2) = 0.05$). Elimination of equation terms was conducted sequentially starting from the full model and removing the least nonsignificant terms one at a time, recalculating t-statistics at each stage in the process. All statistical procedures were conducted using SPSS® (version 7.5 for Windows) statistical software package. Limits to the sizes of the available samples (Table 2.7) necessarily dictated that production function estimates could only be performed for six fisheries technologies: i) China net fishing; ii) trap fishing; iii) hand line fishing; iv) palanca fishing; v) speargun fishing; and, vi) trolling.

Results

China net fishing

A significant C-D production function (equation 2.19) was derived, yet provided little explanatory power:

$$\ln C = 1.34 + 0.78 \ln L + 0.31 \ln F \tag{2.22}$$

($r^2 = 0.079$; $p < 0.001$)

or as transformed into its exponential form,

$$C = 1.34 L^{0.78} F^{0.31} \tag{2.23}$$

where C = catch (kg);
L = crew size (number of individuals); and,
F = soak time (hours).

Gear quantity was excluded from the model as the term's coefficient was not significant ($t = 0.460$, $df = 303$, $p > 0.05$). This is likely a reflection of the survey samples being largely dominated with fishers using only a single net. Applying the base translog model (equation 2.20) did not yield a statistically significant L-F interaction coefficient ($t = 0.231$, $df = 303$, $p > 0.05$), thus reducing to the C-D model (equation 2.23).

Applying the extended translog model (equation 2.21),

$$\begin{aligned} \ln C &= 1.46 + 0.75 \ln L + 0.30 \ln F \\ &-0.39 M_{apr} - 0.44 M_{jun} + 0.92 M_{sep} \end{aligned} \tag{2.24}$$

($r^2 = 0.132$)

Note that the absence of data from the months of November and December prevented the estimation of factor adjustments to the model for fishing activities during those months.

Trap fishing

For trap fishing, a significant C-D production function was derived:

$$\ln C = 0.99 + 0.83 L + 0.42 G \tag{2.25}$$

($r^2 = 0.317$; $p < 0.001$)

or as transformed into its exponential form,

$$C = 0.99 L^{0.83} G^{0.42} \tag{2.26}$$

where C = catch (kg);

L = crew size (number of individuals); and,

G = gear quantity (number of traps).

Fishing time (F) was excluded from the model as the term's coefficient was not significant (t = 0.423, df = 426, $p > 0.05$). A large number of cases (365) were eliminated during the data audit procedure (Table 2.6) solely because the number of crew was not recorded during the survey. The data set with the additional 365 cases (for a total of 795 cases) was subsequently used to explore the derivation of a C-D model without L to determine if the inclusion of the cases would provide substantially different results. The following best-fit model was obtained:

$$\ln C = 0.28 + 0.77G + 0.14 \ln F \tag{2.27}$$

($r^2 = 0.388$; $p < 0.001$)

or as transformed into its exponential form,

$$C = 0.28 G^{0.77} F^{0.14} \tag{2.28}$$

The latter C-D model (equation 2.28) provides greater explanatory power and more realistically reflects the passive nature of trap fishing (i.e. labour is not involved in an active fishing process as traps are left unattended). Applying the base translog model, excluding L as an independent variable in the production process and utilising the more extensive data set, provided the following results:

$$\ln C = 0.48 + 0.68 \ln G + \frac{1}{2}(0.12) \ln G \ln F \tag{2.29}$$

($r^2 = 0.389$)

Similarly applying the extended translog model,

$$\ln C = 1.53 + 0.44 \ln G + \frac{1}{2}(0.09) \ln G \ln F$$
$$+ 0.26 M_{oct} - 0.94 R_{north} \tag{2.30}$$

($r^2 = 0.476$)

Hand line fishing

For hand line fishing, the following C-D production function was derived:

$$\ln C = 0.21 + 1.78 \ln L + 0.45 \ln F \tag{2.31}$$

($r^2 = 0.498$; $p < 0.001$)

or as transformed into its exponential form,

$$C = 0.21 L^{1.78} F^{0.45} \tag{2.32}$$

where C = catch (kg);
L = crew size (number of individuals); and,
F = soak time (hours).

Gear quantity (G) was excluded from the model as the term's coefficient was not significant ($t = 0.771$, $df = 45$, $p > 0.05$). Applying the base translog model (equation 2.20) did not yield statistically significant interaction coefficients (for *L-G* interaction, $t = 1.801$; for *L-F* interaction, $t = 1.891$; for *G-F* interaction, $t = 0.458$; $df = 42$; $p > 0.05$), thus not improving on the C-D model (equation 2.32).

Applying the extended translog model,

$$\ln C = 0.54 + 1.83 \ln L + 0.39 \ln F - 0.74 Y \tag{2.33}$$

($r^2 = 0.571$)

Note that the absence of data from the month of December prevented the estimation of factor adjustments to the model for fishing activities during that month.

Palanca fishing

For palanca fishing, a significant C-D production function was not found. Applying the base translog model (equation 2.20) similarly did not yield a single significant variable coefficient.

Applying the extended translog model,

$$\ln C = 2.33 + 2.87\, M_{aug} \tag{2.34}$$

$(r^2 = 0.138)$

Note that the absence of data from the months of November and December prevented the estimation of factor adjustments to the model for fishing activities during those months.

Speargun fishing

A significant C-D production function was derived for speargun fishing:

$$\ln C = 0.11 + 1.22 L + 0.62 \ln F \tag{2.35}$$

$(r^2 = 0.536;\ p < 0.001)$

or as transformed into its exponential form,

$$C = 0.11 L^{1.22} F^{0.62} \tag{2.36}$$

where C = catch (kg);

L = crew size (number of individuals); and,

F = soak time (hours).

Gear quantity (G) was excluded from the model as the term's coefficient was not significant ($t = 1.814$, $df = 55$, $p > 0.05$). Applying the base translog model (equation 2.20) did not yield statistically significant interaction coefficients (for L-G interaction, $t = 0.891$; for L-F interaction, $t = 0.459$; for G-F interaction, $t = 1.137$; $df = 52$; $p > 0.05$), thus not improving on the C-D model (equation 2.36).

Applying the extended translog model,

$$\begin{aligned}\ln C &= 1.39 + 0.54\ln G + 0.51\ln F \\ &+1.71 M_{nov} - 1.40 R_{north}\end{aligned} \tag{2.37}$$

($r^2 = 0.744$)

Note that the absence of data from the month of December prevented the estimation of factor adjustments to the model for fishing activities during that month.

Trolling

For trolling, the following C-D production function was derived:

$$\ln C = 1.01 + 0.43G + 0.62\ln F \tag{2.38}$$

($r^2 = 0.185$; $p < 0.001$)

or as transformed into its exponential form,

$$C = 1.01 G^{0.43} F^{0.62} \tag{2.39}$$

where C = catch (kg);
G = gear quantity (number of lines); and,
F = soak time (hours).

Crew size (L) was excluded from the model as the term's coefficient was not significant ($t = 1.592$, $df = 233$, $p > 0.05$). Applying the base translog model provided the following results:

$$\begin{aligned}\ln C &= 1.50\ln L + 1.03\ln F \\ &-\frac{1}{2}(1.18)\ln L\ln F + \frac{1}{2}(0.38)\ln G\ln F\end{aligned} \tag{2.40}$$

($r^2 = 0.226$)

Applying the extended translog model,

$$\ln C = 1.44\ln L + 1.01\ln F$$
$$-\frac{1}{2}(1.16)\ln L\ln F + \frac{1}{2}(0.39)\ln G\ln F \tag{2.41}$$
$$+0.89M_{aug} + 0.55M_{sep}$$

$(r^2 = 0.251)$

Discussion

Models of Fishing Effort and Fisheries Production in Jamaica

The utility in formally defining the nature of functions of fishing effort within Jamaican fisheries is in understanding the determinants of catch. By examining the above functions, one can explore the effect of applying varying levels of effort (as measured by labour, gear, and fishing time), as well as the influence of month, region and year. Although in some instances the catch and effort data yielded less then satisfactory model fits, some patterns emerge.

First, in the extended translog models for trap fishing (equation 2.30) and speargun fishing (equation 2.37) a decrease in the catch with a given level of effort was associated with fishing the north coast shelf (R_{north} coefficient significant and negative in both instances). The nonsignificant effect of north shelf fishing on catch is not surprising for trolling (equation 2.47) as it is an offshore activity, the target species for trolling being ocean pelagics. The troll fishery is not primarily dependent on the productivity of the north coast shelf, and thus would not be directly subject to its biophysical limitations. The nonsignificant effect of north shelf versus south shelf and bank fishing on the productivity of China net, hand line, and palanca fishing efforts is perhaps more surprising given the documentation in the existing literature (see Introduction) and the lack of an obvious explanation as to why these would differ from trap and speargun fishing.

Second, there is a positive seasonal influence on catches during the late summer or fall (August through November) for five out of six fisheries examined, although the specific months in which the peak influence occurs and the relative size of the influence varies by fishery. Palanca fishing catch rates (equation 2.34) increase during the month of August, while China net fishing (equation 2.24) is similarly influenced during the month of September. Trolling (equation 2.41) enjoys a seasonal increase in productivity during August and September, although the influence appears stronger in August. Trap fishing (equation 2.30) and speargun fishing (equation 2.37) show an October and November peak, respectively. A significant negative influence on China net fishing catch rates (equation 2.24) during the months of April and June is also evident. The driving forces behind these monthly variations and why they may differ between fisheries remains to be sufficiently explained. Seasonal variations in the availability of fishes (e.g. fall "runs") is the most likely explanation.

Differences in fishing effort productivity between the years 1996 and 1997 was only detected in one fishery - hand lining (equation 2.33) - for which 1997 shows a significant reduction. In all other fisheries examined, year was not a significant influence.

Specific comments regarding the derived production functions can be made regarding each fishing technology examined. Very little of the variance in catch levels for China net fishing could be explained by the models. A more satisfactory model fit could not be found. The extended translog model (equation 2.24) was able to increase the explanatory power of the C-D model ($r^2 = 0.132$ versus 0.079; equations 2.23 and 2.24) by including parameters to account for the months of April, June, and September. It is also evident that labour is the most significant component of effort affecting catch levels, with variations in fishing time having notably less influence. Again, the lack of variation in the quantity of gear utilised as represented in the catch and effort surveys did not allow for sufficient modelling of the influence of that variable - the use of a single net, rather than multiple nets, dominates among fishers sampled.

The examination of trap fishing catch and effort data yielded more satisfactory models. The C-D model (equation 2.28) explained over 38% of the variance, a result not substantially improved upon by moving to the base translog model (equation 2.29). The relatively strong effect of the number of traps employed is not surprising (elasticity = 0.77; equation 2.28). The very low effect of changes in the fishing time on catch levels (elasticity = 0.14) likely reflects the known physical asymptotic relationship between catch levels per trap and time (e.g. Munro 1983b). In other words, fishers seem to be soaking traps for a length of time after which catch levels do not increase substantially. Fishers may even be "over-soaking" traps such that a given amount of gear could be more efficiently used if traps were emptied more frequently. The extended translog model (equation 2.30) explains approximately 10% more of the variance by accounting for increased catches during the month of October and the negative influence of fishing the north shelf waters.

The significant results of the C-D model for hand line fishing (equation 2.32) revealed the dominant influence and increasing returns through the amount of labour applied (elasticity = 1.78). The amount of fishing time seems to have less of an effect (elasticity = 0.45), while gear quantity variations were nonsignificant. The C-D production model shows that at the level of the firm, at which fishing decisions are made, there is an incentive for increasing levels of fishing effort specifically through increasing crew size. There may be many underlying reasons, however, why such an increase in productivity occurs. The causal factors that may underlie these effects (e.g. changes in the way in which labour is applied in the production process) remain to be explored. The extended translog model provided the best fit ($r^2 = 0.571$; equation 2.33). Overall, variations in the month or region fished appear to have a nonsignificant influence over the catch levels, while there was a notable reduction in productivity for 1997.

An examination of palanca fishing catch and effort data did not yield a significant C-D or base translog model fit. Application of the extended translog model did not fair much better, although approximately 14% of the variation in catches could be explained by fishing during the month of August (equation 2.34). Underlying reasons why palanca fishing, in particular, would yield models with such large residual variances, apart from the relatively small sample size available (n = 62; Table 2.7), are not readily apparent.

A highly significant C-D model describing the influence of the application of labour and fishing time on catch levels, explaining over 53% of the variance, was derived for speargun fishing (equation 2.36). It is notable that the model shows increasing returns, with labour having the strongest influence (elasticity = 1.22). The extended translog model (equation 2.37) explained approximately 74% of the variance in catches, showing a strong positive regional and month (November) effect. It is also notable that the translog model excludes labour as a significant variable once region and month are taken into account. This indicates that there are regional and monthly variations in crew sizes which are incidental to catch levels.

A significant C-D model for trolling was derived, yet did not provide a great deal of explanatory power ($r^2 = 0.185$; equation 2.39). Changes in two components of effort (gear and fishing time) make significant contributions to changes in catch levels, although the translog models (equations 2.40 and 2.41) indicate a significant role played by the amount of labour applied and a less prominent role played by the gear quantity employed. There appears to be a pronounced seasonality to catches, with greater catch rates during the months of August and September (equation 2.41), although it must be borne in mind that the extended translog model still leaves a great deal of the variation unexplained.

Additional questions remain regarding the structure of fishing activities within Jamaican fisheries and the effect of various qualities and quantities which may be characteristic of fishing efforts. In all but a few of the models, a large amount of the variation in catch levels remains unexplained. It was not possible to model many factors which may have a significant influence due to the limitations of the available data. For example, it was not possible to distinguish between motorised and unmotorised vessels, or more generally between vessels of different size or power. Although a given set of fishing technologies may be relatively consistent across Jamaica (e.g. traps are of a reasonably consistent size and design), unaccounted differences in gear may also have a significant influence. Daily natural influences, such a unusual weather, may also significantly affect the results of individual surveys conducted on specific dates, resulting in the data set not being truly representative. Alternatively, stochastic variations may simply dominate.

Perhaps the greatest influencing factor which could not be accounted for in the modelling was that of individual fishing skill. For example, recent work in the Montego Bay area (Chapter 3; Bunce and Gustavson 1998) indicate that variations in levels of skill between fishers may have a profound effect on catch levels. It would be desirable in future modelling efforts to employ a methodology to account for such differences (e.g. accounting for age or years of experience among the crew). Of course, as exemplified by bioeconomic modelling approaches, it may also be necessary to more finely account for regional and temporal differences in the availability of fishes rather than the more broadly factoring of annual, monthly, and regional patterns as in the extended translog models employed here.

Fisheries Management in Jamaica

Jamaican fisheries are largely overexploited, resulting in low stock levels and reduced catch rates (Aiken and Haughton 1990; Clemetson 1992; Haughton 1988; Kaslow *et al.* 1994; see also Chapter 3). This is a direct result of the open access management regime. Licensing for inshore artisanal fishers is not restricted, as any individual who is able to produce the nominal fee of J$150 (approximately US$4.25, current as of January 1998) to cover the costs of providing the required identification card is granted a license; similarly, the licensing of all vessels is required, yet there is no fee and few restrictions attached (regulations as specified under the Fishing Industry Act of 1975; Bunce and Gustavson 1998; Espeut 1990; Espeut and Grant 1992; note that higher fees are charged for offshore cay fishers).

The notable exception to the open access regime is the conch fishery on the Pedro Banks, for which a more comprehensive management plan with an established quota system has been put in place. As noted previously, the specific management focus on the conch fishery has, to a large extent, been a result of the requirements for exception under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES; see Armstrong and Crawford 1998 for a description of the convention) to allow for the continued export of the product by the industrial fishery. There are also Jamaican-wide seasonal closures of the lobster fishery to protect gravid females and a minimum size restriction imposed on catches (regulations specified under the Fishing Industry Act of 1975; Aiken and Haughton 1990), yet the effectiveness of these measures and the status of the stocks have not been sufficiently reported.

Contrary to the claims of Berkes (1987), there are no effective common property management structures (in the sense of Ciriacy-Wantrup and Bishop 1975) evident in Jamaica. The distribution of artisanal landing sites along the mainland coast does result in the concentration of individual fishing activities close to home beaches, and individual fishing patterns may be well developed; however, fishing communities do not enforce any weak perception of territory that may exist. The spatial extent of the fishing activities seems to be only limited by the capabilities of the individual vessels or the abilities of the individual fishers (e.g. fishers with non-motorised canoes will often travel as far as they can row in a day; Montego Bay fishers with canoes equipped with outboard engines will set traps as far east down the coastline as Falmouth or further, ignoring the possible territory of other fishers; Bunce and Gustavson 1998). In other words, patterns of fishing are not determined through a set of social norms or a commons management rule base, but are necessarily dictated through the capabilities of specific technologies and the decisions of individual fishers.

Entry into the inshore artisanal fisheries is exacerbated by the extent of poverty in Jamaica and the high unemployment rates (Chapter 3; see Statistical Institute of Jamaica 1998). Fishing provides the primary means of subsistence for many individuals as there are few employment options available (Bunce and Gustavson 1998; Chapter 3). This has lead to what Pauly (1997) has termed Malthusian overfishing - the continuing entrance of new fishers highly dependent on the fishery for their survival, continually increasing fishing effort and resorting to ever-increasing destructive fishing methods in order to meet their needs. It is these newer entrants which largely fish unlicensed and operate increasingly independent from the established fishing communities. There is a general willingness of artisanal fishers in Jamaica to shift into other sectors of the economy (Bunce and Gustavson 1998; Espeut 1992), yet there are few opportunities to do so. The willingness to shift provides some hope in the potential to restrict activity in the fisheries; however, the strong personal attachments to fishing and the independent lifestyle it affords may require that developed alternatives be sensitive to existing values.

Current fisheries management in Jamaica is not only fighting a battle against the poor economic conditions in the country and the financial and resource limitations imposed on the management agencies as a result of government spending restrictions, but also against a general lack of appreciation by fishers of the effect that their own activities can have on the marine resources. Opinions of fishers either vary widely with respect to the cause of catch declines, or they are quick to blame competing resource users (e.g. the tourism sector; Bunce and Gustavson 1998; Espeut 1992). An effective educational campaign will be a necessary component of any management programme as many fishers simply do not recognise the dangers of biological overexploitation.

The numerous complexities associated with multi-species fisheries management, and in particular coral reef fisheries management in the developing tropics, have been discussed in the literature (e.g. Birkeland 1997; Mahon 1990; Munro and Williams 1985; Pauly *et al.* 1989; various contributions in Polunin and Roberts 1996). These provide a starting point and useful guidance, but will not be elaborated upon here. The remainder of the discussion will focus on developing an initial approach towards the economic management of coral reef fisheries in Jamaica. "Economic management" as used here is taken to mean *the management of fisheries with the explicit intent of controlling or guiding the extent of the private and social economic benefits and costs, as well as the distribution of those benefits and costs.* To begin, evidence concerning the general effectiveness of the various available economic instruments, based on the more extensive international experience from temperate fisheries, is briefly reviewed, drawing primarily from the OECD (1997), Squires *et al.* (1995), and Townsend (1990).

Economic Management of Fisheries

Despite the development of fisheries economics over the last fifty years, it is often primarily the concern for biological overexploitation of fisheries which results in management measures being taken to restrict or limit fishing efforts. When the economics of fisheries are considered, it is usually with limited information and analyses (e.g. see OECD 1997 survey). Management measures often aim at restricting fishing effort or capacity through limits on the total allowable catch (TAC), limits on the number of fishing vessels, or reductions in the efficiency of fishing operations (e.g. gear restrictions), yet have not generally had successful results when enacted in isolation (OECD 1997). More generally, management instruments include either restrictions on the *production output* (e.g. total allowable catch, quotas, vessel catch limits), restrictions on *production inputs* (e.g. limited licences, individual effort quotas or restrictions, gear and vessel restrictions), or through the use of *technical measures* (e.g. time closure, area closures, size and sex selectivity; OECD 1997).

Specific economic prescriptions to prevent rent dissipation have fallen in and out of favour. Perceived desirable economic instruments usually aim at establishing property rights (e.g. see various contributions in Neher *et al.* 1989, specifically Scott 1989). The general failure of programmes which previously emphasised the use of limited licensing, as demonstrated primarily by the characteristic overcapitalisation in the industry and continued "race to the fish" which generally results, has led to a shifted emphasis on the use of quotas (e.g. Arnason 1991; Neher *et al.* 1989; Squires *et al.* 1995). In many instances, particularly for temperate fisheries involving developed nations, the use of individual transferable quotas (ITQs) which allocate a portion of a TAC among fishers (either fixed quantity or proportional denomination; Squires *et al.* 1995) likely offers the best hope, yet can not be assumed desirable in all cases (e.g. Scott 1988, 1989). Most notably, high transaction and enforcement costs may make the use of ITQs undesirable. Highgrading (the discarding of lower-valued catch in favour of higher-valued catch), as well as the discard of species until the quota for certain species is reached, have been documented (OECD 1997; Squires *et al.* 1995). Thin or fragmented ITQ markets may also decrease economic efficiencies (Squires *et al.* 1995). It is further difficult to conceive of an effective ITQ management regime for complex multi-species fisheries given current technical knowledge and management capabilities.

Fisheries management efforts have had such a dismal record in part because they often ignore or downplay the economic, social and political incentives of the fishers and the management authorities. The particular economic characteristics of a common pool resource in conjunction with the incentives of individual fishers (not only directly tied to resource use, but often dependent on a larger socio-economic context; e.g. see Chapter 3), even with established property rights, can result in both the biological and economic overexploitation of the fisheries. In other words, net economic returns are not maximised because a socially scarce good is treated as a free good by individuals (Gordon 1954). The result is the excessive depletion and eventual elimination of the net social benefits that can be derived from the resource and the emergence of excessive fishing capacity. With excessive capacity, a given amount of fish are harvested with more inputs than is necessary, resulting in "wasted" inputs which would potentially be better utilised in alternate production processes (Gordon 1954; Munro and Scott 1985).

As noted as early as Schaefer (1957), however, the goal of maximising net economic yield in fisheries may not be the only valid social policy goal. Management efforts which aim to maximise sustainable biological yield have been long entrenched in policy and seek to maximise total physical production or production value directly from the resource. Policies may desire to surpass both maximum economic and biological yield levels of harvest in order to satisfy employment or community development goals (Charles 1988). The latter may be seen to be particularly important in the management of developing nation fisheries where there are few alternate employment opportunities. The debate may be academic in practice, however, as challenging theoretical problems may prevent the successful achievement of maximised economic rent even if desirable, due to such factors as variations in cost and price structures, the differing nature of long-run versus short-run externalities, varied biological characteristics of fish populations evident in multi-species fisheries, temporal and spatial variations in fish population characteristics, and overall uncertainty (OECD 1997; Schaefer 1957; Townsend 1990).

Figure 2.11 presents a conceptual framework which summarises the major elements which determine the effectiveness of fisheries management regimes. The complexities of the management environment, in conjunction with the implementation of multiple instruments for any one or a related set of fisheries, often make even the appropriate use of measures with a theoretical suggestion of success doubtful.

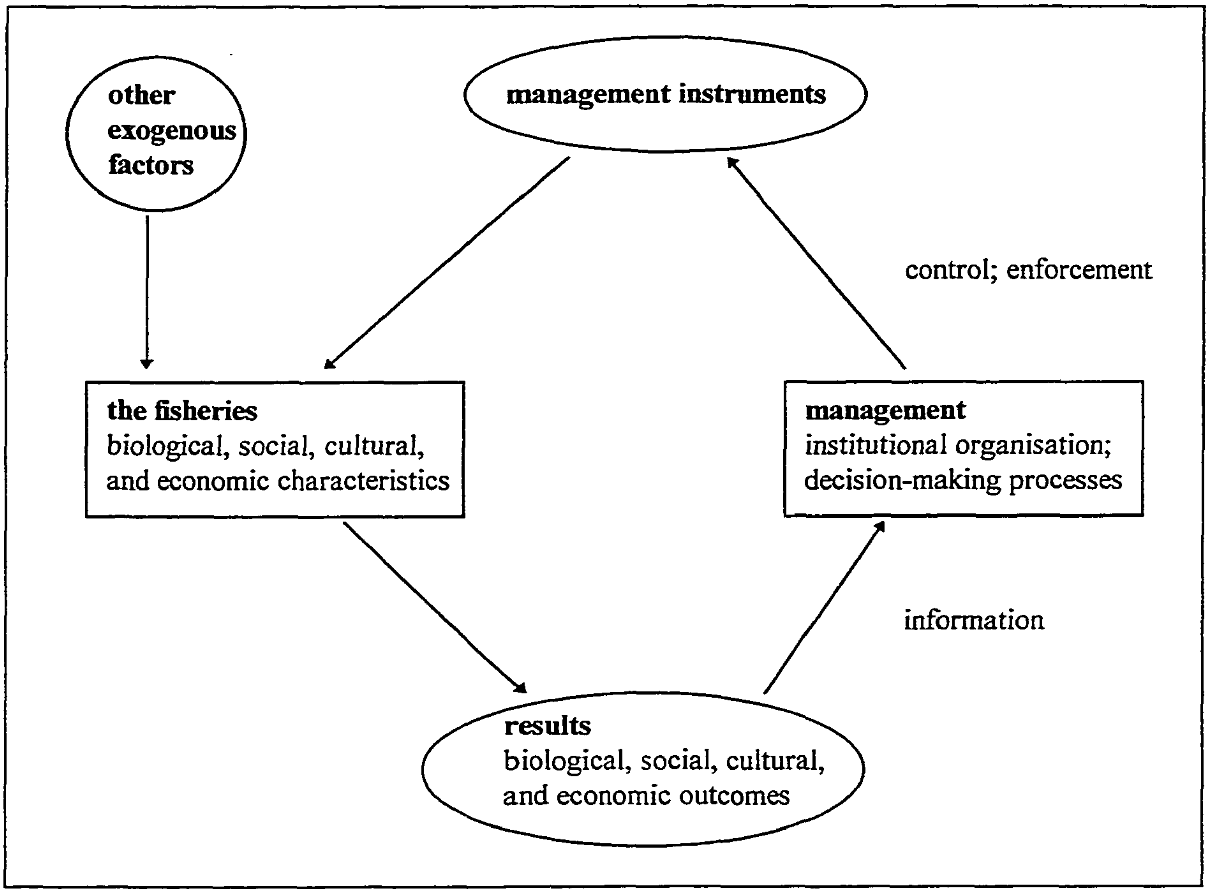

Figure 2.11. Conceptual framework for the analysis of the effectiveness of fisheries management regimes (adapted from OECD 1997).

The allocation and enforcement of property rights (e.g. ITQs, exclusive area user rights), often in conjunction with co-management regimes (e.g. community based management), are likely necessary if effective management is to be achieved as evidenced by the limited but demonstrated success in temperate fisheries (OECD 1997). The application of such instruments to the developing tropics, however, will require consideration of region-specific socio-economic and cultural conditions. Such measures may effectively require fishers to take into account all of the costs of their activities, and avoid problems associated with overcapacity or over-capitalisation in the fishery. The required legal framework, additional administrative and management costs, and structural adjustments among the user groups may make such changes difficult (OECD 1997). In addition, the social welfare consequences of management regimes, such as changes in the distributions of incomes and individual wealth and the level and forms of employment, should not be ignored.

A few comments are warranted concerning common property management. As noted by Stevenson (1991), there may be difficulties associated with the allocation of individual property rights to common pool resources due to: i) the physical difficulty of doing so (i.e. technology does not allow for effective exclusion); and, ii) the potential for adverse impacts on wealth and income distribution (e.g. there may be a high degree of general community dependence on the resource). Establishing common property involves the vesting of property rights within a framework of group control. Common property management has all of the following characteristics (Stevenson 1991):

- a resource with bounds that are well defined by physical, biological, and social parameters;
- a well-delineated group of users, distinct from those excluded;
- multiple users participating in resource extraction (to distinguish from private property);
- explicit and/or implicit, well-understood rules regarding rights and duties to one another regarding resource extraction;
- joint, non-exclusive entitlement shared by users to the *in situ* or fugitive resource prior to its use (again, to distinguish from private property);
- competition for the resource, users thereby imposing negative externalities on one another, although the extent of the externalities is controlled (to distinguish from a corporation); and,
- a well defined group of rights holders, which may or may not coincide with the group of users (to distinguish from property tenure).

Common property management strategies for the effective management of tropical coral reef fisheries have received considerable recent attention, given the difficulties of attaching individual property rights in developing country settings. In specific instances, particularly where there are long-established aboriginal fishing populations with developed traditional knowledge, commons management structures may be the most effective manner in which to treat coral reef fisheries when allowed to function without the disrupting influences of government interventions (e.g. see Birkeland 1997; Johannes 1997). Such is not the case in Jamaica, where there is no evidence of persistent social structures or a traditional knowledge base associated with fishing activities which could readily accommodate a commons management regime. The success of common property management depends on *group enforcement* and *assurance of others' behaviours* (Stevenson 1991), conditions which do not lend themselves to the current realities within Jamaican fisheries.

Co-management of a fisheries, in which there is non-government or quasi-government organisational involvement at the local level with management decisions and actions in conjunction with higher or federal level government management, may reduce government costs, increase the flow and quality of local information available to managers, increase the flexibility and timeliness of management decisions, and increase local level compliance. In the Montego Bay area, there is an interesting and potentially fruitful relationship being forged between the Montego Bay Marine Park authorities and the Fisheries Division (Government of Jamaica) to work together towards the effective management of the fishing activities of the local fishers (J. Williams, Montego Bay Marine Park, pers. comm.). It is hoped that a continued co-operative relationship between the NGO which governs the Park and the federal government, in conjunction with a programme to build trusting, working relationships with the local fishing communities, will lead to the more effective management of the fisheries. Although in its initial stages, the programme may serve as a model for other such ventures. However, the conditions of a multi-species fisheries utilising diverse fishing technologies and methodologies involving many distinct users, some not connected to local fishing communities, may make such co-management options more difficult to successfully implement (OECD 1997). The effective limitation of fishing effort and catch levels through an institution of co-management involving the fishers must most notably overcome difficulties associated with disincentives to forming contractual working arrangements due to the lack of information, asymmetric information, and the unwillingness to confront wealth distribution questions (Scott 1993).

With or without co-management, the perceived legitimacy by fishers of the management options depend on the following (OECD 1997):

- the existence of a common understanding of the problem;
- a perception that the procedures used to arrive at management decisions and actions are fair;
- a perception that the outcomes of the management measures are fair; and,
- a perception that the resulting measures will be effective.

These elements are important to achieve if management authorities are in turn to gain compliance by the fishers.

Towards the Economic Management of Jamaican Fisheries

Specific fisheries management options that have been suggested for Jamaica include gear limitations (e.g. minimum mesh size requirements) and area closures (e.g. creation of fish sanctuaries and reserves), accompanied by resource enhancement (e.g. artificial reefs), fisher education and publicity campaigns, and improved institutional co-ordination and involvement through a multi-agency management advisory council (Aiken and Haughton 1987, 1990). Gear limitations have received perhaps the most attention, with a targeted equipment exchange programme on the north coast (Discovery Bay) to replace existing traps with larger mesh traps resulting in a significant improvement in the catches (Sary *et al.* 1997). All of these suggested measures, however, do very little to directly address either the economic conditions of the fishers or the institutional and economic factors contributing to the overexploitation of the marine resources.

This is by no means to suggest that answers are readily apparent. Before appropriate instruments are selected, it is essential that more be known concerning the economic and socio-cultural structures and conditions associated with the fisheries. The implementation of economic management instruments can not be assumed *a priori* to be feasible or socially efficient (Wilson 1982) given our current knowledge of the system. Understanding the economic incentives involved is particularly important. To begin the process of information collection and analysis need not require an unrealistic input of financial resources, yet will require a concerted effort. The monetary flows associated with fishing activities are largely unknown. Indeed, collection of the data may prove difficult. For example, many fishers will fear that divulging financial information to government officials will simply result in their income becoming subject to tax (Espeut 1992).

Fisheries research programmes in Jamaica are poorly funded - a direct result of not only the government's overall financial difficulties, but the low national priority of fisheries management initiatives (Aiken and Haughton 1990). The low socio-economic position and continued marginalisation of artisanal fishers within society is a notable hindrance to effective coral reef fisheries management in developing countries (Pauly 1997). Government subsidisation of Jamaican fishing activities without the political will to address the critical management issues further promotes the economic overexploitation of the resource (e.g. the provision of duty-free gasoline by the government and the operation of co-operative stores which also provide supplies duty-free; Aiken and Haughton 1990; Espeut 1992; Espeut and Grant 1990).

The collection of descriptive information concerning the nature of fisheries production processes, the existence of resource rents (be they positive, zero, or negative), and the distribution of the economic costs and benefits is necessary. The above analysis, which formally derives artisanal fisheries production functions, represents a first step towards the economic management of the resource. Due to data limitations, rent analysis was not possible at the time of this study, yet steps have been taken to collect the necessary information.

Expansion of the Jamaican Catch and Effort Data Collection Programme towards the Economic Management of the Fisheries

The Fisheries Division (Ministry of Agriculture, Government of Jamaica) expressed an interest in a programme for the establishment of baseline economic information on Jamaican fisheries, and subsequent analysis and modelling of that information to assist with government decision-making and the establishment of policy. The opportunity arose to assist with the establishment of such a programme. The collection of supplemental information would provide much of the production costing necessary to conduct a rent analysis of the coral reef fisheries.

The Fisheries Division is ultimately interested in examining "acceptable" potential output of the fisheries or the net economic value that can be realised through fishing, including considerations of government management costs (Andre Kong, Fisheries Division, pers. comm.). This is not an easy task given the current lack of economic information on the Jamaican fisheries. As a start, the Division requires documentation of the contribution of fisheries to the economy and descriptions of the rents captured by the fishers. The economic analysis and modelling programme should ultimately have the following end products in mind, as all are essential components in the first steps toward the economic management of the fisheries:

- production function descriptions;
- rent analysis; and,
- analysis of the potential effects of management alternative on economic yields.

The fisheries economics programme should be established within the Fisheries Division to collect the needed economic information and increase the capacity of the Fisheries Division for analysis, yet not be an excessive burden on the limited available government resources. Due to the importance of first establishing a methodology which would be effective yet efficient, it was necessary to initially narrow the focus of the data collection. The artisanal coral reef fishery was selected for a pilot project. A survey was designed which complemented and expanded on the current Catch and Effort Data Collection Programme, with data collection beginning October 1998 by Fisheries Division staff on a limited number of landing beaches. The co-ordination of the economic sampling programme with the catch and effort programme will ensure that the two data sets are commensurable for subsequent analyses.

Expansion of the Existing Data Collection Programme

The analyses possible with the currently existing catch and effort database, in conjunction with the fisheries flow chart information and the "opportunistic" economic costing information (see Table 2.4), does not provide sufficient information for the economic management of the fisheries. Much of the necessary costing information is not currently collected; thus, it is necessary to supplement the existing database. Specifically, the Catch and Effort Data Collection Programme survey (Table 2.8) was expanded with a focus on providing additional information on:

- operating costs associated with fishing (including bait, ice, and food and beverages);
- the direct use of energy in the fisheries (i.e. the value of fuel used);
- the distribution of incomes among those involved in fishing (i.e. the income sharing arrangement); and,
- the loss of equipment through fishing.

Table 2.9 shows the supplementary form developed for use in conjunction with the pre-existing Catch and Effort Collection Form to provide the necessary information as the basis for a fisheries economics data collection programme.

Table 2.8. A reproduction of the Catch and Effort Collection Form for data collection specific to landing site and date of collection (Fisheries Division, Ministry of Agriculture, Government of Jamaica).

boat registration #								
boat name								
boat length								
crew size								
time/ date depart								
time/ date return								
weight type (V,F,W)								
gear	1		2		1		2	
gear type								
soak								
depth fished								
# gear used								
mesh size								
net weight								
fishing ground								
catch by species		wt(lb)	wt(lb)	price		wt(lb)	wt(lb)	price

Table 2.9. Supplemental Catch and Effort Data Collection Form developed for data collection to support an economic analysis programme (to be appended to form shown on Table 2.8).

boat registration #								
boat name								
total amount of fuel used[a]		1		2		1		2
price paid for fuel ($ per gal)[b]								
income sharing arrangement (%)[c]	captn	crew	crew	owner	captn	crew	crew	owner
total operating costs (other than fuel)[d]								
equipment loss[e]								
comments								

[a]Record total amount of fuel used for the specific surveyed fishing trip; units unimportant, but **record** the units used (e.g. $ or gallons).

[b]Record the price (in $ per gallon) paid for the fuel used for the specific surveyed fishing trip.

[c]Record the arrangement for the share of the income from the catch for the specific surveyed fishing trip (in % distribution); record in the **comments** section if the owner of the boat is also the captain.

[d]Record the total operating costs **not** including fuel (e.g. bait, ice, food and beverage); do **not** include any cost associated with maintaining or repairing equipment (e.g. fishing line, net repairs, etc.) or the purchasing of equipment; record in the **comments** section how operating costs are paid (e.g. costs of fuel and other operating costs taken off the top before income is shared?).

[e]Record any loss of equipment or significant damage to equipment during the specific surveyed fishing trip (e.g. lost traps, lost fishing line and lures, etc.); also record the monetary value of the loss if the information is provided.

Information on the value of the produced capital in the fisheries (i.e. the value boats, engines and fishing equipment) was collected concurrent to the sample survey through the Census of Fisheries conducted by the Government of Jamaica during the fall of 1998. In addition to the survey items of Table 2.9, the economic database should ideally also maintain and update as required the current market value of the following capital items:

- fishing vessels and engines (data coded by vessel name and registration number); and,
- fishing gear (by type of gear).

It is important to be able to link the estimated market values of the vessel and engine(s) to the specific vessels interviewed for each catch and effort data collection survey. For fishing gear, however, it will be sufficient to obtain a more general estimate of gear market prices and apply this to the specific vessel based on the type and quantity of gear employed. This will allow for the calculation of rents through the estimation of an annuity factor for all capital contributions to costs, and may also allow for a more informed aggregation of fishing vessels into "like" activities for production function analyses.

Economic Analysis and Capacity Building within the Fisheries Division

It is essential that capacity is built within the Fisheries Division itself to maintain the collection, analysis and interpretation of a fisheries economics programme to the extent possible. This need not and likely can not require a significant input of financial resources to achieve. The following should be explored during development of the programme to allow for self-maintenance to the fullest extent possible:

- development of standards and protocols for data collection, analysis and interpretation;
- acquisition of software, hardware, and/ or other tools for improved database management and analyses;
- information sharing workshops with management and staff; and,

- identification of external resources and access to expertise.

To assist towards this end and in addition to the design of the supplemental survey, it is suggested that a manual be produced and made available to Fisheries Division staff and management which outlines principles of fisheries economic data collection, analysis and interpretation for the programme. This should focus on a suggested set of standards and protocols to help ensure that the data collection and storage would facilitate further economic analyses in the future.

Chapter 3: The Economics of a Small-Scale Commercial Fishery: the Artisanal Fisheries of Montego Bay, Jamaica

The following presents the results of a focused field study of the fisheries utilising the waters of the Montego Bay Marine Park, Montego Bay, Jamaica. The work was part of a socio-economic assessment of the three primary user groups (fishers, watersports operators, and hoteliers) and an overall local use economic valuation study of the Park, the results of which have been reported in Bunce and Gustavson (1998) and Gustavson (1998). The socio-economic assessment and local use valuation studies were in turn part of a larger World Bank study that is exploring and testing valuation methodologies for the estimation of benefits derived from coral reefs in the developing tropics. The overall objective of the broader World Bank project is to assist policy makers in managing and protecting coral reefs through improved estimates of benefits, and is to be used in conjunction with a management interventions least-cost model being developed concurrently (Huber 1998; final reports forthcoming).

Historic, systematic and reliable information on the size of the nearshore, artisanal fisheries of Jamaica is not available (see Chapter 2). Reliable records of the number of fishers and the method of fishing began when the Fisheries Division of the Government of Jamaica began a Registration of Fishermen Database in 1995. Economic information regarding fisheries in Jamaica is even more limited. Espeut (1992) and Espeut and Grant (1990) provide information, yet it is not directly applicable to the nearshore fisheries in the Montego Bay area. Nicholson (1994) conducted a spring 1994 socio-economic survey of fishing activities, including the sampling of catches, in the Montego Bay Marine Park. His study represents the only pre-existing source of economic information for the area and will be used in conjunction with the information provided in the World Bank studies to develop a profile of fishing activities.

Some brief comments should be made regarding notable limitations to the Nicholson (1994) study. First, the survey was conducted over a limited time period in the spring, yet there may be a strong seasonality in the fishing effort and the characteristics of the catch (Chapter 2). As noted by

Nicholson (1994), only about 60% of the seaworthy vessels at the River Bay landing beach (the study site) during the time of the study were directly involved in fishing - fishing likely has a variable and demand-adaptable component. Further, most marine species of commercial importance will exhibit seasonal behaviour patterns which affect the availability within the fisheries. The information contained in the studies of Gustavson (1998) and Bunce and Gustavson (1998) may also suffer from many of the same limitations due to being conducted over a limited time frame. However, these latter studies relied more heavily on existing annual statistics and, despite being conducted over the two months of January and February only, interviews were designed to reveal more general annual and seasonal information.

Fishing Activities by Landing Beach

The primary means of fishing within the Montego Bay Marine Park are trap fishing (using z-traps or antilean-type traps), spearfishing, hand line fishing (also called hook-and-line), and net fishing (including trammel or drag nets, and China or gill nets). As of early 1998, there were approximately 400 registered fishers operating from 126 boats using five landing beaches located within or just outside the Park boundaries (Figure 3.1 and Table 3.1). In contrast to the total number of *registered fishers operating from Montego Bay area landing beaches,* there are an estimated 380 fishers actively *fishing within Park waters* (this figure is based on the summation of the following: an assumed 150 unregistered spearfishers, 10% of all registered fishers from Whitehouse landing beach, and 100% of all registered net, spear, trap, hand line, and "other" fishers in the other four landing beaches; these adjustments to the registration figures were made based on the results of interviews with the local fishers as reported in Bunce and Gustavson 1998). The number is a significant increase from 1995 registration statistics, in which 237 fishers are recorded as fishing from 83 registered boats. The large increase in the numbers over the last three years shown by the statistics is likely a result of two factors - an increase in the number of boats and fishers, and the registration database itself becoming more complete over time. The most recent previous registration statistics from a survey in 1981 recorded 159 boats (Sahney 1982).

As of 1998, over half of the registered fishers launch from the River Bay landing beach and almost a third launch from Whitehouse landing beach. Fishers based out of three minor landing beaches -

Reading, Bogue, and Spring Gardens - also use Park waters (a fourth minor beach outside the eastern most edge of the Park, known as Rum Bottle, is used as a base from which fishers out of Great River sell their catch; during interviews, fishers from River Bay noted that Great River fishers will do a small amount of hand line and trap fishing within the eastern most area of the Park, yet Rum Bottle fishers are excluded from the analysis here). There are few registered spearfishers operating out of the five landing beaches in the Montego Bay area; however, there are a large number of unregistered commercial spearfishers illegally fishing park waters, launching from indiscriminate points along the shoreline (a rough estimate of approximately 100 to 200 is provided through interviews with local fishers; Bunce and Gustavson 1998).

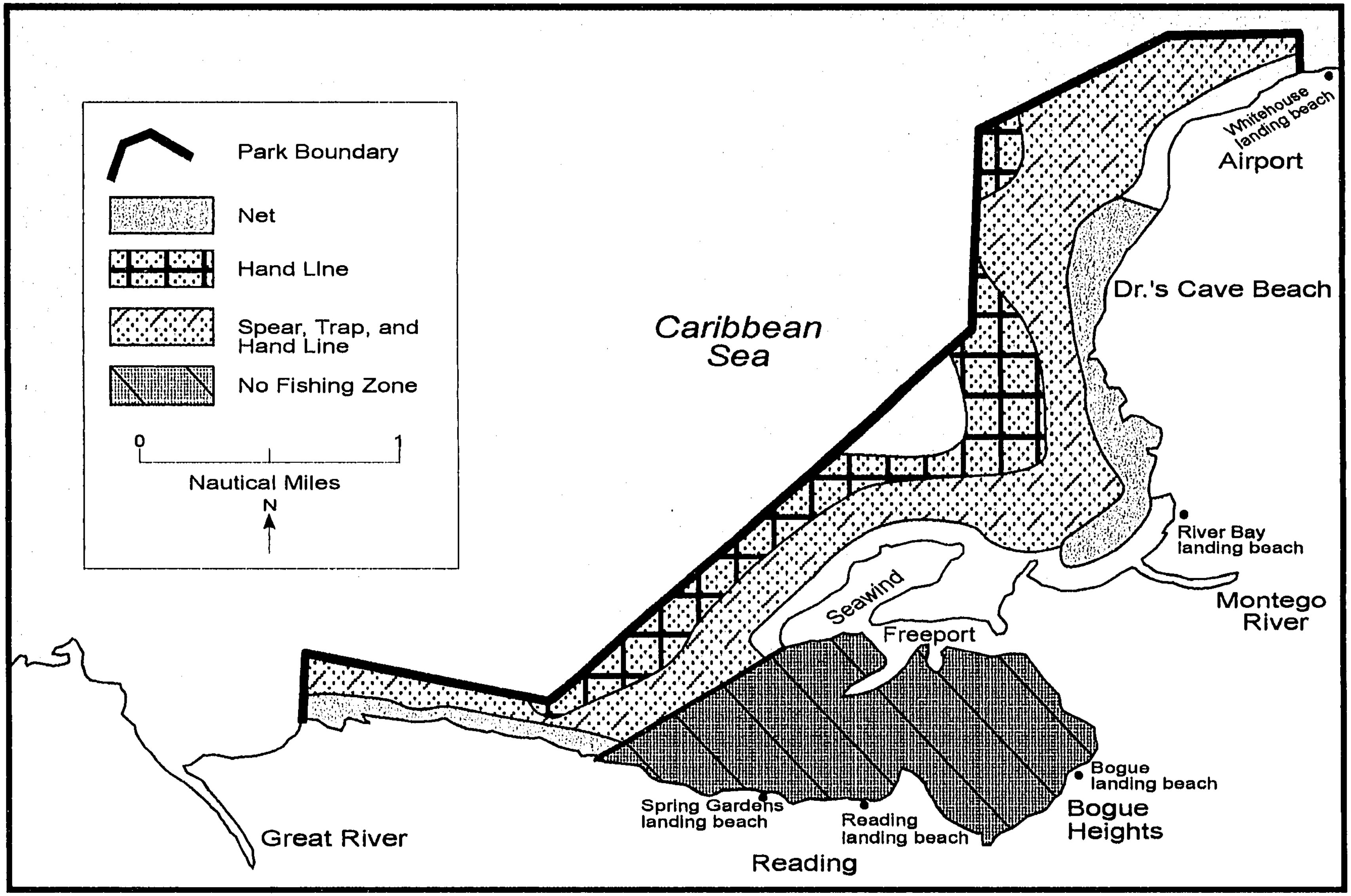

Figure 3.1. Location of fishing activities within the Montego Bay Marine Park, Montego Bay, Jamaica.

Table 3.1. Total number of fishers and boats by landing beach and by year (Sahney 1982; Registration of Fishermen Database, Fisheries Division, Government of Jamaica, 1995 and 1998).

landing beach	number of boats			number of fishers	
	1981	1995	1998	1995	1998
River Bay	92	48	64	128	202
Whitehouse	30	16	44	70	146
Bogue	10	5	1	6	8
Reading	12	10	12	22	33
Spring Gardens	8	4	5	11	8
Kerr Wharf[a]	7	—	—	—	—
total	159	83	126	237	397

[a]No longer operational.

The Fisheries Division's 1995 registration statistics indicate that hand lining and net fishing are the predominant forms of fishing out of River Bay, comprising 53% and 17% of the fishers, respectively (Table 3.2). In contrast, the River Bay fishers noted during interviews that there are currently *five* types of fishing occurring at significant levels: trolling, hand line fishing, trap fishing, net fishing, and spearfishing (troll fishing occurs outside of the boundaries of the Montego Bay Marine Park; Bunce and Gustavson 1998). Registration statistics also indicate that trolling and hand line fishing are the predominant forms of fishing from Whitehouse. However, the interviews revealed that trap fishing is also a common fishing technique among Whitehouse fishers, more so than the 6% registered in 1995. This discrepancy between interview results and government statistics may be primarily due to the way in which government registration statistics are recorded. Specifically, the Fisheries Division only registers fishers by their primary means of fishing. In the Montego Bay area, many of the troll fishers also set traps.

According to 1998 registration statistics compiled by Fisheries Division, there are a total of 18 boats and 49 fishers operating out of Bogue, Reading and Spring Gardens landing beaches, which is significantly fewer than Whitehouse and River Bay (Table 3.1). Although the individual landing beaches have varied in the number of fishers and boats since 1995, overall the total number of boats has remained approximately the same (in fact decreasing by one) and number of fishers has increased by only 10 individuals. 1995 registration statistics indicate that hand lining and trap fishing are the predominant forms of fishing out of these three beaches.

Table 3.2. Total number of fishers by landing beach and by method of fishing for 1995 (Registration of Fishermen Database, Fisheries Division, Government of Jamaica, 1995).

landing beach	hand line		trap		troll		net	
River Bay	68	(53%)	8	(6%)	9	(7%)	22	(17%)
Whitehouse	36	(51%)	4	(6%)	24	(34%)	0	(0%)
Bogue	6	(100%)	0	(0%)	0	(0%)	0	(0%)
Reading	11	(50%)	4	(18%)	3	(14%)	0	(0%)
Spring Gardens	6	(55%)	4	(36%)	0	(0%)	0	(0%)
total	127	(54%)	20	(8%)	36	(15%)	22	(9%)

landing beach	beach seine		spearfishing		other/ none/ unknown	
River Bay	3	(2%)	3	(2%)	15	(11%)
Whitehouse	0	(0%)	1	(1%)	5	(7%)
Bogue	0	(0%)	0	(0%)	0	(0%)
Reading	0	(0%)	0	(0%)	4	(18%)
Spring Gardens	0	(0%)	0	(0%)	1	(9%)
total	3	(1%)	4	(2%)	25	(11%)

Types and Patterns of Fishing

The general pattern of fishing within the Montego Bay Marine Park is shown in Figure 3.1. According to Fisheries Division 1995 statistics, trap fishing is not a very common means of fishing. Only 20 of 237 fishers are registered as trap fishers (Table 3.2). However, interviews with fishers indicate that trap fishing is common as both a primary and secondary means of fishing (again, the discrepancy may be because, for many fishers, trap fishing is secondary to other means of fishing; Bunce and Gustavson 1998). Trap fishing involves rowing or motoring to varying locations, setting and leaving a series of z-traps (wire mesh supported by a wooden frame, taking on a characteristic z shape) for approximately one week, after which they are emptied and reset. Using either oranges or eel as bait, these fishers primarily catch coral reef finfish, deepslope finfish, coastal pelagics, and crabs and lobsters. Trap fishers typically check their traps on Sunday during the morning hours. They usually leave between 5 am to 7 am and return between 9 am to 12 pm. Consequently, it takes between 2 hrs to 5 hrs of time to travel to the various locations, haul and reset the traps. The number of traps that each boat sets is variable, but interview respondents noted an average of 20 to 30 traps per boat (Fisheries Division 1995 registration data record an average of 33 per registered trap fisher).

Trap fishing typically occurs within a mile of shore and at varying depths. River Bay fishers noted that they typically set traps at less than 60 ft. Some fishers set as deep as 80-90 ft to avoid vandalism by spearfishers, but this was noted to reduce catch rates. In contrast, Whitehouse fishers stated that they typically set traps in 30 ft of water or less in sand near the reefs. At times, Whitehouse fishers also reportedly set their traps deeper in order to avoid vandalism, but this again may have the effect of reducing the catch rates.

According to Fisheries Division 1995 statistics there are only 22 net fishers, all based out of River Bay (Table 3.2). This relatively small number is confirmed by interviews and personal observations (Bunce and Gustavson 1998) which indicate that River Bay is the main launching beach for net fishing. Net fishing typically occurs in sandy areas, often between the reefs and the shore. The primary net fishing sites are behind Dr.'s Cave reef, along River Bay landing beach, and east of Spring Gardens landing beach (Figure 3.1). There are two main types of nets in use - trammel (or drag) and China (or gill) nets. China nets operate by snagging the fishes when they encounter a filamentous mesh, while trammel nets operate by entangling the fishes between two layers of nylon mesh. Fishers locate schools of fish and drive them into the set nets (usually beating a paddle on the water to scare the fish) where they become snagged or entangled. Trammel nets are commonly used in favour of China nets. Net fishers set their nets in the evening, leaving shore at 4 pm to 5 pm and returning by 9 pm (i.e. involving from 4 hrs to 5 hrs of time), during which time various nets are worked. Net fishers fish an average of three times per week, with some fishers only going out once per week. Net fishers target coral reef finfish and coastal pelagics.

Approximately 50% of fishers use hand lines (Table 3.2). Hand line (or hook-and-line) fishing occurs from offshore to shallow areas and subsequently overlaps with many of the other fishing activities, particularly trap fishing (Figure 3.1). River Bay hand liners, for example, focus on relatively shallow areas operating at a depth of about 60 ft, but are known to fish deeper (Bunce and Gustavson 1998). Hand line fishing, which involves dropping one or two hooked lines off the side of an anchored boat, occurs predominantly at night. Fishers typically leave at 7 pm or 8 pm and return to the beach anytime between 4 am and 8 am (i.e. involving between 8 hrs and 13 hrs of time). Hand liners primarily catch coral reef finfish, coastal pelagics, and deepslope finfish.

The number of spearfishers recorded from registration information is small compared to the number of spearfishers estimated through the interviews (Bunce and Gustavson 1998). According to Fisheries Division data there were 4 registered spearfishers in 1995 operating out of Whitehouse and River Bay (Table 3.2). There is believed to be a large number of unregistered spearfishers currently operating within Park waters, with estimates between 100 and 200 provided. The number of spearfishers is believed to have increased drastically over recent years. During interviews conducted by Bunce and Gustavson (1998), one respondent noted that approximately 15 to 20 years ago, one could count the number of spearfishers in Montego Bay “on your hands”, whereas today there are so many it is difficult to arrive at an accurate figure (note again that spearfishers are predominantly unregistered and fishing illegally within Park waters; consequently, it is not possible to rely on statistical information regarding numbers of individuals and the nature of their activities). Fishers from both Whitehouse and River Bay indicated that currently the majority of spearfishers that operate out of landing beaches use Bogue, Reading and Spring Gardens. Although some spearfishers are based in one of the landing beaches, it is believed that most do not characteristically operate exclusively from any one location. Typically, when a spearfisher wants to fish, he or she simply travels to the shore and enters the water at a convenient location, which can be anywhere along the shoreline of the Park.

Spearfishers fish to a depth of up to 60 ft and up to about half a mile offshore. Within the Park waters, the most popular spearfishing areas are the reef from the airport to the Dr.'s Cave, the reef along the north-western side of Seawind, and westward along the coast to the Great River (Figure 3.1). It was reported during interviews (Bunce and Gustavson 1998) that spearfishers do not fish in the Bogue lagoon (respecting the marine sanctuary as it is seen as an important fish breeding ground), inside the Montego Bay proper (near River Bay beach) or near Pier 1 (avoiding locations near the Park office). Coral reef finfish and deepslope finfish are the target species.

Most spearfishers fish five to seven days per week. Approximately 90% of all spearfishers in the Montego Bay area are believed to be free-lung divers. Scuba equipment is seen as something that is bothersome and costly to maintain, and obtaining dive certification is difficult for most fishers. Obtaining tank air fills can also be difficult, but is usually done through friends who work in the local diving industry. Spearfishers predominantly fish at night because it is easier to catch the reef fish at the time when many are less active (particularly the parrotfish, which can be found "resting" on the bottom at night), and presumably to lessen the chance of detection by Park authorities and other user groups. The need to fish at night was also attributed to the low reef fish stocks and the difficulty of finding high enough concentrations of fish to make day spearfishing worthwhile (Bunce and Gustavson 1998). When fishing at night, spearfishers will fish for a maximum of 8 hrs if free-lung fishing, but only a maximum of 4 hrs if using scuba equipment. When fishing during the day, spearfishers are reported to be usually out from sun-rise to sun-down.

The final significant method of fishing used by fishers who operate from the five landing beaches discussed here is trolling, which typically occurs one to four miles offshore and outside of Park boundaries. According to Fisheries Division 1995 statistics, approximately 15% of fishers are registered trollers (Table 3.2). The predominant equipment used is either rod and reel or troll line (a single line wrapped on a spool). Offshore pelagics are the primary target species. Trolling occurs an average of four days a week, with fishers leaving early in the morning (6 am to 7 am) and returning late in the day approximately 8 hrs to 15 hrs later.

Target Species and Catch Rates

Fisheries Division's 1995 registration data indicate that deepslope finfish, coral reef finfish, and coastal pelagics are the predominant target fishes, with 38%, 32%, and 14% of the fishers targeting those species, respectively (Table 3.3). These statistics reflect the importance of reef-related and near-shore coastal areas for the fishers.

Table 3.3. Total number of fishers by landing beach and by fish targeted for 1995 (Registration of Fishermen Database, Fisheries Division, Government of Jamaica, 1995).

landing beach	coral reef finfish	coastal pelagics	deepslope finfish	offshore pelagics	other/ unknown
River Bay	40 (31%)	22 (17%)	48 (38%)	9 (7%)	9 (7%)
Whitehouse	14 (20%)	5 (7%)	35 (50%)	9 (13%)	7 (10%)
Bogue	5 (83%)	1 (17%)	0 (0%)	0 (0%)	0 (0%)
Reading	12 (55%)	2 (9%)	4 (18%)	3 (14%)	1 (5%)
Spring Gardens	5 (45%)	2 (18%)	3 (27%)	0 (0%)	1 (9%)
total	76 (32%)	32 (14%)	90 (38%)	21 (9%)	18 (7%)

Catch rates are believed to be extremely variable. As one fisher noted during interviews by Bunce and Gustavson (1998), "Every day is a fishing day, but everyday is not a catching day." Skill and experience are considered to have a great deal to do with the individual catch rates, with the older fishers being the more "skilled".

Bunce and Gustavson (1998), during interviews with local fishers, tried to identify the range of typical catch rates. Whitehouse fishers noted that troll and trap fishes often only catch 10 to 20 lb of fish in a week; however, at other times each boat can catch up to 200 lb in a single week.

Inquiries into catch levels indicated that hand line fishers average 10 to 20 lb per landing, and net fishers average 10 to 15 lb. Spearfishers reportedly average 10 lb per outing. Regardless of the type of fishing, estimates of average catches never exceeded 30 lb per fishing trip. These estimates are confirmed by Nicholson (1994) who reported an average of 15.7 lb per landing per boat during a spring 1994 survey of catches. Specific estimated catch weights per landing by type included 8.3 lb for net fishing, 16.9 lb for trap fishing, 13.2 lb for hand line fishing, and 19.3 lb for spearfishing (Nicholson 1994).

Fishing activities and catch levels are expected to have an annual seasonality. Fishers also noted that they will catch different species of fish depending on the phases of the moon (Bunce and Gustavson 1998). Certain types of equipment, whether net, hand line, or trap, will be more successful than others at different times of the month. This variability has ramifications for fishers who rely on one type of method and, consequently, experience significant monthly variations in catches. It is possible for fishers who use more than one type of equipment to strategically choose when to utilise each method so as to maximise their catch. Unfortunately, given the limited amount of information on fishing activities in the Montego Bay area, it is not possible to quantify the annual or monthly variations in catch rates, or specifically how this varies by the equipment used (although see Results in Chapter 2).

Reef fishing is reported to be poorest from January through April due to the colder water and the often poor weather conditions (Bunce and Gustavson 1998). During this time of the year, deep-sea trolling is often more successful. Coastal catches peak during the months of September and October. At this time, there are more people fishing than at any other time of the year, as individuals who do not normally fish will come down to the beaches to assist with the catches. Many fishers noted, however, that this typical fall peak in catches did not occur in 1997 for some unknown reason (some fishers speculated that a change in the water current and temperature patterns resulted in the fish bypassing the north coast of Jamaica; Bunce and Gustavson 1998). As the fall is the largest money making time for the fishers, a small “run” can make it difficult for them to cover many annual expenses such as equipment maintenance.

Characteristics of Employment and Incomes

Fishing in the Montego Bay area is largely subsistence artisanal fishing. During interviews, all fishers expressed difficulty with being able to make a living - as one fisher noted, "[We] want to make a living fishing but we can't. Mostly fish just to eat." (Bunce and Gustavson 1998). In contrast to the south coast of Jamaica where there are more significant commercial fisheries, the Montego Bay landing beaches have relatively small amounts of fish sold directly to the public along the road and on the landing beaches.

When questioned regarding the extent to which they are dependent on fishing for a source of income, fishers' responses varied widely; however, all stressed the reliance on fishing as either their primary or sole source (Bunce and Gustavson 1998). River Bay interviewees estimated that between 75% and 95% of fishers rely on fishing as a sole source of income. These estimates are roughly consistent with 1995 Fisheries Division registration statistics which report that 77% of Whitehouse fishers, 70% of River Bay fishers, and an average of 69% of fishers for all landing beaches are full-time (Table 3.4). In contrast, statistics available for 1994 indicate that 93% of fishers were full-time (Nicholson 1994). This difference is likely attributed mainly to the differences in data collection techniques, particularly the limited and selective sampling of the Nicholson (1994) study in contrast to the information provided by a larger cross-section of fishers to the Fisheries Division through registration. Although the statistics indicate that most fishers likely do not have other significant sources of income, interviewees noted that many fishers take other jobs in order to survive (Bunce and Gustavson 1998).

Table 3.4. Total number of fishers by time worked in fishing and landing beach (Registration of Fishermen Database, Fisheries Division, Government of Jamaica, 1995).

landing beach	full-time		part-time		no-time	
River Bay	90	(70%)	37	(29%)	1	(1%)
Whitehouse	54	(77%)	16	(23%)	0	(0%)
Bogue	3	(50%)	1	(17%)	2	(33%)
Reading	15	(68%)	6	(27%)	1	(5%)
Spring Gardens	2	(18%)	9	(82%)	0	(0%)
total	164	(69%)	69	(29%)	4	(2%)

Typical of artisanal fisheries in Jamaica, there is an income share arrangement between the crews, captains, and owners of the boats (see also Espeut 1992; Espeut and Grant 1990). Sharing arrangements are informal and often flexible, but well established and reflect a relationship of trust built among fishers who operate from the same vessel. Understanding this share arrangement is critical to understanding the distribution of the economic benefits.

Usually, fishers retain some of the catch for personal consumption. A portion of the gross sales is taken off the top to cover the operating expenses, and the remaining gross sales are distributed among the boat owner, the captain, and the crew. The arrangement at Whitehouse and River Bay (Bunce and Gustavson 1998) typically involves 50% of the gross value of the catch or weight of the catch (at times distribution is based on the weight of the catch not the income from sales, particularly if the catch is used primarily for sustenance) going to the owner of the boat to cover operating expenses, equipment maintenance expenses, and as a return for the capital investment. The remaining 50% is distributed equally among the captain and crew who operated the fishing vessel (the captain is usually, but not always, also the owner of the boat). Assuming the most common arrangement of having two individuals fishing from one boat, this usually means that each captain and crew takes 25% of the catch. Where there are more than two fishers, the income is accordingly less (e.g. if there is one captain and two crew members, individuals take one third of 50% of the catch, or approximately 17%).

This share arrangement, however, is varied at times. With the use of certain types of fishing equipment, such as traps, the catch will be distributed according to each fishers' proportional contribution to the total amount of equipment used (e.g. if one fisher own 40% of the traps deployed, he or she will received 40% of the catch distributed to all fishers). Owners may also decrease the percentage share retained for the boat when the catches are low in order that the crew receive higher incomes (Bunce and Gustavson 1998). Owners of special gear (e.g. the owner of the net used) will also receive an extra share of the catch (Espeut 1992; Espeut and Grant 1990).

All fishers were understandably reluctant and evasive concerning inquiries into their average incomes during interviews conducted by Bunce and Gustavson (1998). Using the information provided, however, some estimates can be made. Based on the estimated number of fishing trips per week, estimated average catches, the average price of fish per pound, and the boats' usual sharing arrangements, weekly individual incomes (before taxes) per fishing activity are estimated as follows: J$750 to $2500 (US$21 to $70) for trolling (conversion from Jamaican dollars to US dollars assumes, as elsewhere in this chapter, that J$35.5 = US$1 based on the average of the median bid and the median asking price in world markets on the first of the month for the first five months of 1998), J$250 to $500 (US$7 to $14) for trap fishing, J$750 to $2250 (US$21 to $63) for net fishing, J$750 to $2500 (US$21 to $70) for hand line, and J$5000 to $7000 (US$141 to $197) for spearfishing (Table 3.5). A realistic average weekly individual income (excluding taxes) for most fishers is between J$2000 and J$3000. Clearly, spearfishing is the most profitable means of fishing. Based on a survey of fishers in 1994, Nicholson (1994) reported average net returns per day per boat of J$400 to J$1000 in current 1994 dollars. For comparison purposes, these figures were converted into 1998 dollars using an average annual inflation rate of 28% per year to obtain an estimate of J$1074 to J$2684 in current 1998 dollars (the assumed annual domestic inflation rate of 28% is based on a 10 year average of the annual implicit price deflator for total GDP for Jamaica from 1987 through 1996; source: the Statistical Institute of Jamaica). This estimate appears consistent with the one provided by Bunce and Gustavson (1998), given that most boats will fish three to five days a week with, on average, two fishers per boat.

Table 3.5. Estimates of catches, gross incomes per boat, and individual incomes of fishers by method of fishing for 1998.

method of fishing	number of outings per week	approximate catch per outing (lbs)	approximate weekly gross income per boat assuming J$100/lb[a] (J$)	approximate weekly individual income[b] (J$)
troll	3 to 5	10 to 20	3000 to 10000	750 to 2500
trap	1	10 to 20	1000 to 2000	250 to 500
net	3 to 5	10 to 15	3000 to 7500	750 to 2250
hand line	3 to 5	10 to 20	3000 to 10000	750 to 2500
spear[c]	5 to 7	10 lbs	5000 to 7000	5000 to 7000

[a]In general, coral reef "table fish" (fish 1.5 lbs each and up) will sell for $J100 per lb, while "frying fish" (fish under 1.5 lbs) will sell for $J50 per lb in early 1998. Species caught by trolling command more specific prices: for example, $J100 per lb for dolphinfish, $J70 per lb for blue marlin, $J60 per lb for tuna (noted as one of the harder fish to sell), and $J100 per lb for kingfish. J$100 per lb was used for calculations, assuming that higher value fish are caught.

[b]Weekly individual incomes per fishing activity were estimated as 25% of the approximate weekly gross income per boat. This assumes a typical sharing arrangement and an average of one captain and one crew member per boat.

[c]Spearfishing has no sharing arrangement, with expenses and capital investments expected to be fewer than those required for other forms of fishing; thus, although their net will be less than the gross due to expenses, no adjustments were made to the gross incomes as reported on this table.

As mentioned, operating expenses and expenses associated with the maintenance of the capital investments are typically covered through retention of 50% of the value of the catch from fishing. General repairs to boats can be significant, reportedly costing on average J$10,000 to J$20,000 per year (Bunce and Gustavson 1998). Due to a lack of local supplies, this can mean the fishers having to make a trip to Kingston to get what is required (e.g. parts for engines). Fishers frequently commented during interviews that most savings had to go to boat and equipment repairs and replacement (Bunce and Gustavson 1998). Required maintenance of equipment varies by the type of fishing. For example, meshed wire is required for repairs and the maintenance of traps, with the average fisher reportedly having to buy one bundle every eight months at a cost of J$4,500 to J$5,000 (one bundle will make two traps). Sticks for the traps cost J$100 for a bundle of 12, requiring 1.5 bundles to make a trap. Traps will usually last for between six months and one year before requiring replacement. The greatest operating expense associated with trolling is the cost of gasoline (fisheries requiring anywhere between 5 gal and 20 gal, depending on the size of the engine and the time spent fishing, with gasoline costs reported to be between J$52 and J$55 per gal). However, Nicholson (1994) reported that mechanised boats had significantly higher catch rates, thus potentially compensating for the extra costs. Line and tackle can also be a significant expense, but frequency of replacement is highly variable.

Rent and Net Present Values Earned through the Montego Bay Fisheries

To arrive at the annual value of the contribution of the marine waters of Montego Bay Marine Park to direct economic benefits from fishing, the *net value* of those fishing activities was calculated. The net value is the remainder of the total monetary value of the benefits once all existing economic claims to the production have been deducted. This remainder is the economic production claim which can be attributed to the marine system. To calculate the net value associated with coral reef use, all variable costs which represent a claim on economic production must first be deducted from the gross receipts from fishing. This potentially includes the opportunity costs associated with labour, operating services sold to the fishers, repairs and maintenance, goods and materials, government license and registration fees, and insurance. It does not include such items as government taxes and subsidies (transfer payments), as these are not payments for activities which involve economic production. Similarly, any internal financial transactions, such as depreciation, or external financial transactions, such as bank interest payments, are not to be included (for a description of methodologies associated with the calculation of net values, particularly as they concern cost-benefit analysis, see Brent 1996; Nas 1996).

The net operating values are then translated to true net values (i.e. a full cycle analysis) by converting the value of capital investments or stocks to annual flow values to be deducted from the annual net operating values. The "equivalent annual capital cost" can be estimated through the use of an annuity factor:

$$E = \frac{C}{AF} \tag{3.1}$$

where E = equivalent annual capital cost;
C = value of capital at cost; and,
AF = annuity factor.

An infinite time horizon is assumed, such that $AF = 1/i$, where i is the discount rate used in the specific net present value (NPV) calculation (see below). Total capital investments by fishers was estimated as the value of boats and engines at cost. Information regarding the value of other capital equipment at cost (e.g. fishing equipment) was not forthcoming or possible to reasonably estimate.

For the next step in the calculation, we assume that a continuing, sustainable use is possible at the level of use for the given year and that the total value in which we are interested takes into account an infinite stream of net annual benefits. Thus, the *net present value* (NPV) of fishing activities is calculated. NPV can be simply thought of as the current equivalent net value associated with the use of the Montego Bay Marine Park waters for fishing, or the contribution of marine biodiversity to productive economic output summed annually over an infinite time stream. Future values are discounted in order to reflect the social time preference rate. To illustrate the sensitivity of the analysis to the chosen discount rate, three rates are separately assumed in the calculations: 5%, 10%, and 15% per annum.

The NPV is thus represented as

$$NPV = \frac{(R - C)}{i} = \frac{NV}{i} \tag{3.2}$$

where R = revenue;
C = costs;

i = discount rate (5%, 10%, and 15%); and,

NV = annual net value.

It must be emphasised that the derivation of NPVs as done here is not a cost-benefit analysis *per se*. In a cost-benefit analysis, one would compare the economic value of the resource after an intervention (e.g. a management strategy which would improve reef conditions) with the economic value before an intervention. This report does not consider the effect of possible management interventions on the economic value of fishing activities derived from the reefs of Montego Bay Marine Park, nor the changes in derived value with changes in reef quality or the quantities of fishes. The NPVs reported here represent the "value at risk"; in other words, it is the direct local use value associated with fishing which would be lost if the resource was completely degraded.

Bunce and Gustavson (1998) were not able to reliably determine through the interviews what portion of the 50% of all earnings from fishing, as retained by the owner of the boat, was consumed by the operating expenses and equivalent annual capital costs, and what portion would represent producers' surplus. The captains and crew are "hired hands" from the point of view of the owner of the boats; thus, it is the owners that would effectively be in the position to capture most of the fisheries rent. The exception to this is the spearfishers, for there is typically no sharing arrangement as they are primarily sole operators. For spearfishers, the vast majority of which are believed not to operate from a vessel, it is similarly important to deduct an estimate of the operating expenses and the opportunity cost associated with fishing.

For the methods of fishing for which we have economic information (Table 3.5), hand line, trap, net and spearfishing occur within Park waters (trolling occurs outside of Park waters). To ultimately derive the NPV from fisheries, we need to first deduct an estimate for the operating expenses, equivalent annual capital costs, and opportunity costs of labour from the gross incomes per boat for net, trap and hand line fishing. In the case of the spearfishing, operating expenses and the opportunity cost of labour should be deducted from the gross earnings.

Using the 1995 registration statistics for the total number of fishers by fishing method by beach and applying them proportionately to the 1998 estimated total number of boats and fishers by landing beach (Tables 3.1 and 3.2), we can arrive at an estimate for the number of boats and fishers using Park waters (Table 3.6; this excludes 13 fishers recorded as non/ unknown/ other method of fishing). We can then use these results to arrive at an estimate for the total number of "owners" (owners of boats for hand line, trap, and net fishing; sole operators for spearfishing). Table 3.7 shows these results.

Nicholson (1994) estimated that total operating costs for fishers (less labour payments) were between 11% and 34% of gross revenues. 25% was assumed here for the calculation of net operating values for all forms of fishing (indications from interviews with fishers during the field portion of the study of Bunce and Gustavson 1998 supported this approximation). In other words, approximately 75% of the gross receipts for net, trap, hand line, and spearfishing can be assumed to be operating surplus less the deduction for the payment to labour.

In 1996, the average hourly wage for large establishments, all sectors for Jamaica as a whole was J$56 (source: the Statistical Institute of Jamaica). This statistic is converted to a 1998 equivalent of J$92 assuming an annual increase in the wage rate of 28% reflective of the general inflation rate (again, the conversion assumes an average annual domestic inflation rate of 28% based on a 10 year average of the annual implicit price deflator for total GDP for Jamaica from 1987 through 1996; source: the Statistical Institute of Jamaica). Assuming a 40 hour work week (see Statistical Institute of Jamaica, *Statistical Yearbook of Jamaica*, for a breakdown of typical work week hours), an average weekly wage in Jamaica is J$3670. To arrive at an estimate for the opportunity cost for labour, this average weekly wage (assuming it reflects the value of the marginal product of labour) is discounted by 25% (to reflect the availability of alternate employment) to arrive at a final figure J$2750 per week per individual (note that the official 1996 unemployment rate in Jamaica was 16% according to the Statistical Institute of Jamaica).

J$2500 per week is the income which fishers indicated during interviews would have to be the minimum income earned through alternate employment for them to leave fishing, and the typical wage reportedly being offered by the tourism industry for fishers who have found employment in that sector (Bunce and Gustavson 1998); however, this being the "opportunity cost" of employment it must be noted that the reality for many fishers is that there are few opportunities for alternate employment. Based on $J2750 per week, the opportunity cost of labour is estimated to be

J$5500 per boat for net, trap, hand line fishing (assuming an average of one captain and one crew member per boat), and J$2750 per spearfisher. These costs are deducted from gross fishing earnings along with the previously cited estimate of 25% of gross for operating expenses. The derivation of the overall net operating values (i.e. a partial cycle analysis) are shown in Table 3.7.

Table 3.6. Total number of fishers and boats by landing beach estimated to be fishing in the waters of Montego Bay Marine Park in 1998 for which there is economic information (Bunce and Gustavson 1998; Registration of Fishermen Database, Fisheries Division, Government of Jamaica, 1998).

landing beach	number of boats[a]	number of fishers[a]
River Bay	51	161
Whitehouse	5	15
Bogue	1	8
Reading	8	23
Spring Gardens	5	8
unregistered spear fishers	---	150
total	70	365

[a]The totals exclude 13 fishers, and the proportionate number of boats, recorded as non/ unknown/ other method of fishing.

Table 3.7. Derivation of annual net operating values (current J$) by method of fishing for 1998.

method of fishing	weekly gross income	total number of owners	weekly net operating value per owner	total annual net operating value (without capital deduction)
trap	1000 to 2000	13	-4750 to -4000	$-3.21x10^6$ to $-2.70x10^6$
net	3000 to 7500	10	-3250 to 125	$-1.69x10^6$ to $6.50x10^4$
hand line	3000 to 10000	47	-3250 to 2000	$-7.94x10^6$ to $4.89x10^6$
spear	5000 to 7000	154	1000 to 2500	$8.01x10^6$ to $2.00x10^7$
total	n/a	224	n/a	$-4.83x10^6$ to $2.23x10^7$

Nicholson (1994) estimated the average value of the boat capital assets (including vessels and engines) to be on average J$58,000 (current 1994 dollars) per owner. This is approximately equivalent to J$156,000 in current 1998 dollars. The equivalent annual capital costs are thus J$7,800 for $i = 0.05$, J$15,600 for $i = 0.10$, and J$23,400 for $i = 0.15$ for each boat owner. These figures are then deducted from the annual net operating values for net, trap, and hand line fishing "owners" (but not for spearfishers) as shown in Table 3.7, yielding 1998 net annual values. Total deductions for annual capital costs are thus J$$5.46x10^5$ for $i = 0.05$, J$$1.09x10^6$ for $i = 0.10$, and J$$1.64x10^6$ for $i = 0.15$. The resulting net annual values are then converted to NPVs. The results are shown in Table 3.8.

Table 3.8. Net annual values and net present values (millions of current J$) for the fisheries of Montego Bay Marine Park, 1998.

	$i = 0.05$	$i = 0.10$	$i = 0.15$
net annual value	-4.83 to 21.8 (midpoint of 8.5)	-5.92 to 21.2 (midpoint of 7.6)	-6.47 to 20.7 (midpoint of 7.1)
net present value (NPV)	-96.6 to 436 (midpoint of 170)	-59.2 to 212 (midpoint of 76)	-43.1 to 138 (midpoint of 47)

Discussion and Implications for Further Research

The calculation of NPVs assumes that the level of use in the base year is sustainable - the benefits will continue to be received in perpetuity. The validity of this assumption must be checked against biophysical information regarding the conditions of the reefs in Montego Bay as they have changed over time. Moreover, any future or continuing changes in reef ecological conditions will necessarily have an effect on the current levels of local use. There is one notable documented ecological survey (Sullivan and Chiappone 1994) which examines reef conditions in the Montego Bay Marine Park. As well, there is additional information available on the reef conditions as perceived by the primary user groups; this latter information is outlined in Bunce and Gustavson (1998). This report will not attempt to make assumptions regarding the sustainable level of local use. The coral reefs of Montego Bay are part of a highly complex system, involving interactions between ecological components, user groups, and land-based activities. Although there are certainly negative ecological impacts associated with increases in the levels of local use, the relationship is not simple, nor can the ecological impacts be isolated from other coastal and land-based activities. The high degree of system uncertainty, as well as system links, synergisms and feedbacks, make assumptions regarding the sustainable level of use difficult.

The marine environment of the Montego Bay Marine Park is affected by a diversity of activities, including watersports, urban development, and artisanal fishing. In large part due to these activities, the Park's marine resources are most notably impacted by poor sewage and solid waste disposal practices, sedimentation from coastal construction, habitat loss from coastal infilling and mangrove destruction, intensive exploitation of the fisheries resources, and physical damage from boat anchors, as well as scuba and snorkelling activities. Further impacts have occurred by coral bleaching events, the massive die-off of the long spined black sea urchin (*Diadema antillarium*) in the early 1980s, Hurricane Allen in 1980, and Hurricane Gilbert in 1988 (see Discussion in Chapter 4).

The successful management of coral reef resources in the developing tropics requires not only consideration of the biophysical conditions that determine system structure and processes, but also an understanding of the social and economic conditions, contexts, and motivations that are associated with their use. It is through the consideration of the local use values, in conjunction with subsequent analyses regarding the ecological condition of the reef and the sustainable level of reef use, which will allow for management authorities to obtain more complete information on the extent of the reef-derived local economic use benefits at risk of being lost if conservation efforts prove inadequate.

Under the current open access management regime, one would predict that all rents associated with fisheries would have dissipated. Although the range of estimates (Table 3.8) certainly include zero and negative rents, midpoint estimates indicate that rent is being earned. Why might fishing rents be positive? Fishing rents are most likely maintained through socio-cultural and expertise barriers. The results of Bunce and Gustavson (1998) indicate that fishing activities are associated with a particular socio-economic class and that fishers themselves do not become proficient at fishing until they have gained the necessary experience. Those outside the fishing communities would likely find it difficult to fish profitably. It was even noted during interviews with Montego Bay fishers (Bunce and Gustavson 1998) that wealthier individuals not associated with the fishing communities will at times try fishing, but will soon cease operations due to low catch rates, being unfamiliar with how or where to fish. The experience gained by the older fishers seems largely to be passed on through persistent involvement in fishing and interaction within the fishing communities themselves. Spearfishers, who are more likely to enjoy the largest rents, are less tightly linked to the fishing communities and, thus, might be expected to be subject to fewer socio-cultural barriers of entry. However, experience and the unfamiliarity of many Jamaicans with the marine environment would still factor largely into their level of fishing success, and even their willingness to begin fishing in the first place. The overall effectiveness of any barriers of entry into fishing, however, is not absolute. More individuals are fishing (especially spearfishing) as is evidenced by the relatively recent and rapid increase in the number of fishers in Montego Bay. This increase in the number of fishers is expected to continue with the persistence of positive fishing rents.

In addition to direct marketed benefits such as fishing, in which components of the marine system input as commodities directly and explicitly into economic activity, there are also indirect local uses. Indirect benefits can be defined as ecosystem functional contributions to economic production value, providing implicit and integral support of economic activities. The issue of valuing the marine system structural and functional diversity can also concern itself with the creation of artificial markets (e.g. estimating existence and bequest values as revealed through contingent valuation), or the creation of new markets (e.g. bioprospecting values, which estimate system value through a distribution of profits or value-added associated with marine product development).

The direct local use values for fisheries utilising the waters of the Montego Bay Marine Park as reported here (Table 3.8) serve to illustrate a more traditional economic approach to resource valuation. As outlined in Chapter 1, the monetary value of the contribution of the natural environment to the final value of fisheries products is represented by the resource rent earned. However, this dissertation is ultimately concerned with the examination of biophysical values - the extraction of ecosystem function value in the coral reef artisanal fisheries of Jamaica. How is the biophysical structure and function value represented by harvested fishes measured and how does it compare with market use values? This is the question to which we turn in Chapter 4.

Chapter 4: Jamaican Coral Reef Fisheries and Captured Ecosystem Values

The trick is to loose
the wild bear
but hold tight
to the chain,
woe
when the bear
snatches up
the links
and the man dances.

Alden Nowlan, *Hymn to Dionysus*

Introduction

As argued in Chapter 1, the economic use of renewable natural resources involves the capture of the productive value generated by those natural assets. This chapter explores an alternative measure for the value of the contribution of ecosystems to production in the coral reef fisheries of Jamaica. A non-monetary index will be developed and explored which will measure the ecosystem function value *captured* by the economic process as *embodied* in the fishes. Such an index ideally must have a biophysical basis and reflect the functional characteristics of the utilised ecosystem. Does such an index of ecosystem function currently exist or can one be suitably adapted for the examination of captured ecosystem value by an economic process utilising a renewable natural resource base? If such an index is currently not available, how is one to be developed?

Isard (1969, 1972; see also Lonergan and Cocklin 1985) presents a conceptual framework for the analysis of the interactions or linkages between economic and ecological systems. It is based in a traditional commodity-by-industry input-output modelling framework describing linear interactions between sectors within the economic system (e.g. see Miller and Blair 1985), but extends the matrix to include ecological commodity inputs and outputs to economic activities, economic commodity inputs and outputs to ecological processes, and ecological commodity inputs and outputs to ecological processes. Each cell entry in the larger matrix represents an input of a commodity into (a positive cell entry) or use of a commodity by (a negative cell entry) a process or activity. Figure 4.1 depicts the framework as adapted from Isard (1969, 1972).

	economic activities	ecologic processes
economic commodities	economic system: intersector coefficients	ecologic processes: input and output coefficients re: economic commodities
ecologic commodities	economic sectors: input and output coefficients re: ecologic commodities	ecologic system: interprocess coefficients

Figure 4.1. Framework for the analysis of ecological-economic interactions as adapted from Isard (1969, 1972).

The most restrictive feature of the input-output style matrix is the assumed linear relationship between, in the case of Figure 4.1, the inputs and outputs that define the nature of the activities or processes (i.e. technical coefficients assume constant input proportions, describing constant returns to scale). Key non-linear relationships can not be explicitly defined within the matrix, and thus must be analysed externally. The model also assumes a level of information which, in reality, may not be sufficient. Despite the significant drawbacks, the framework is particularly attractive in that it does allow for the "...systematic description in relationships and magnitudes as they exist at a point in time. This description in itself often provides much insight." (Isard 1972, p. 95). Ecologists have not applied input-output analysis extensively to the study of ecosystem structure and function (for explicit examples of ecological applications see Finn 1976; Hannon 1973).

Figure 4.1 provides a useful conceptual framework for this investigation. The lower right quadrant represents interactions within the ecological system and is analogous to what an ecologist envisions through the definition of food webs, with food webs depicting the flow of ecological commodities between trophic species or groupings. Positive cell entries in the upper right quadrant represent outputs of ecological processes provided as resources or products directly as economic commodities. Negative cell entries in the lower left quadrant represent the use of ecological commodities within economic processes. Whether the product of an ecosystem process is recorded as an output of an ecological process (economic commodity, upper right quadrant) or as an input to an economic process (ecological commodity, lower left quadrant) involves an arbitrary decision on behalf of the researcher (Isard 1969).

For the purposes of this investigation, we are interested in the biophysical value of products of natural ecosystems as ecological commodities, expressed as some measure of ecosystem function value per unit of economic production value within a given economic sector. This defines technical coefficients within the ecological-economic model:

$$a_{ij} = \frac{Z_{ij}}{X_j} \tag{4.1}$$

where a_{ij} = technical coefficient defining the production relationship between the ecological commodity input i (biophysical units) and the economic sector j (dollars);

Z_{ij} = flow of ecological commodity input i to economic sector j (biophysical units) and,

X_j = gross output value of economic sector j (dollars).

Firm-level economic production processes and use of ecological commodities will be examined, ignoring more macroeconomic intersector flows as represented by the larger input-output framework (Figure 4.1). In other words, we will be concerned with the direct capture of biophysical ecosystem values as represented by technical coefficients within each technologically distinct coral reef fishery of Jamaica.

What may be used as an index of ecosystem function value for use in equation 4.1? Certainly, it would be possible to rely on a simple measure of biomass, yet it is more advantageous to adopt a measure which more explicitly and consistently acts as a proxy for the ecosystem function which gave rise to the commodity. There is a growing body of ecological work which continues to examine the relationship between traditional measures of biodiversity and ecosystem function (e.g. Bengtsson *et al.* 1997; di Castri and Younés 1990; Ehrlich and Ehrlich 1981; Grassle *et al.* 1991; Lamont 1995; Schulze and Mooney 1994; Walker 1992). The evidence is often contradictory and, if anything, seems to point in the direction that the most often relied upon measures of biodiversity may not be adequate indices because the relationship is often functional process specific, ecological system specific (in space, time, and scale), and dependent on the definition of biodiversity employed. About all that can be generally said about the role of biodiversity in relation to function is that it provides the "...medium for energy and material flows, which in turn provide ecosystems with their functional properties..." (Myers 1996). Yet, as we are more narrowly concerned with a tropical coral reef ecosystem in this study (i.e. a specific system), it may be that a currently existing index of biodiversity is able to sufficiently act as a proxy measure of a specific, relevant functional process one may want to define.

Biodiversity Indices Developed for Ecosystems

Community or species composition indices, encompassing what is more usually referred to as biodiversity indices, are the most firmly established and heavily utilised family of indicators for simply describing the species components of ecosystems. Given space restrictions, it will not be possible to fully review all existing biodiversity indices here as they may apply to this project. The theoretical basis for major index "groups" are briefly outlined and comments made with respect to their potential application as proxy measures for ecosystem function.

There are numerous ecosystem composition indices available. The greatest drawback facing most indices is their inappropriateness for comparing "unlike" systems and the limited information they provide regarding the functioning of ecosystems. Following the method of Magurran (1988), biodiversity measures can be broadly categorised as (i) species richness indices (and species-area curves), (ii) indices derived from species abundance models (e.g. geometric series, logarithmic series, log-normal distribution, the broken stick model), and (iii) indices based on the proportional abundances of species. Taxonomic distinctiveness measures have recently emerged and are treated here as, in essence, a fourth category. The discussion concerning this latter group of indices is more fully expanded as presented here, due to its particularly relevant application to ecosystem conservation concerns and the considerable recent attention the measures have received.

Species Richness Indices

Species richness is the most simple way in which to summarise the components of a community. The most common measure of species richness is simply the total number of species (S). Other possible desirable measures of richness (Magurran 1988) include Margalef's index:

$$D = (S-1) / \ln N \tag{4.2}$$

where D = diversity;

S = number of species; and,

N = total number of individuals among all species.

Care must be taken, however, in describing species richness as there will be systematic variations in the index dependent on the size of the area sampled. Consequently, a simple measure of species richness (S) or equation 4.2 alone is inadequate.

Rosenzweig (1995; see also Gaston 1996 for an overview of species richness concepts), in exploring the patterns of diversity over space, identifies four basic types of species richness relationships as described by species-area curves: 1) among tiny pieces of single biotas; 2) among larger pieces of single biotas; 3) among islands of one archipelago; and, 4) among areas that have had separate evolutionary histories. Each separate "type" of species-area curve reflects a separate set of common processes operating at different scales (Rosenzweig 1995). Indicators of diversity may be derived from the described relationships. Upon reflection, it is evident that when comparing species richness based on species-area relationships, one must compare similar systems at similar scales, as well as account for any non-linearities in the relationship, for the metric to be valid (Rosenzweig 1995). It is not readily evident how species richness may systematically or in a predictable manner relate to ecosystem function.

Indices Derived from Species Abundance Models

Diversity indices have been derived from mathematical descriptions or through the comparison of curves describing the distribution of the numbers of individuals among species (e.g. log series α). As outlined by Magurran (1988), the four models of species abundance (i.e. geometric series, logarithmic series, log-normal distribution, and the broken stick model) can be viewed as representing a successional and/or habitat-type specific progression from a system with a few dominant species to a system with many species more or less equally abundant. In other words, a geometric series pattern is seen in harsh or early successional environments (dominated by a few species), whereas the broken stick model is applicable in stable or late successional habitats (with more or less equal numbers of many species).

The distinguishably different partitioning of resources may be reflected in the different distribution patterns as revealed in the models (particularly the geometric series and the broken stick) and thus may be biologically meaningful (Magurran 1988). The log series is the most widely used species abundance model and is a reasonable model for most situations; however, the model is statistically derived and thus does not necessarily lend itself to biological interpretation (i.e. resource partitioning or function attributes; Magurran 1988). Moreover, the validity of comparisons between regions represented by different model types becomes ambiguous as the statistics are not commensurable given that the weighting of species richness and proportional abundance differs as the index is dependent on the model from which it is derived. Only "like" distributions may be compared, but even in such circumstances the interpretation may be questionable (e.g. intersecting curves).

Indices Based on the Proportional Abundances of Species

Indices based on the proportional abundances of species take into account both the species richness and evenness into account. This includes indices derived from information theory, such as the Shannon index (Magurran 1988) which measures uncertainty or complexity:

$$H = -\sum p_i \ln p_i \tag{4.3}$$

where H = Shannon diversity; and,

p_i = proportion of individuals found in the ith species.

And indices which are considered dominance measures (weighted towards the more abundant species), such as Simpson's index (Magurran 1988):

$$D = \sum \left({n_i(n_i - 1)} \middle/ {N(N-1)} \right) \tag{4.4}$$

where D = Simpson's diversity

n_i = number of individuals in the ith species; and,

N = the total number of individuals in the community.

For all measures in this category, they differ in essentially how they are weighted - whether richness or evenness is emphasised. In any event, their behaviour is known to be highly dependent on the underlying species abundance pattern (see preceding section; Magurran 1988), and thus again are restricted to comparing "like" systems, although statistical jack-knifing may obviate this concern (Dixon 1993; Zahl 1977). How indices vary systematically with other variable characteristics of systems remains to be explored.

Taxonomic Distinctiveness Measures

Taxonomic distinctiveness measures as composition indices are beneficial for two primary reasons - the ability to compare disparate regions meaningfully by considering the species compositions on a common cladistic or taxonomic tree, and the widespread focus shown by researchers on the conservation of a diverse set of organismal features (assuming taxonomy as a proxy for phylogeny; e.g. Faith 1992, 1994). The concept of taxonomic distinctiveness has been most fully and appropriately developed by Solow *et al.* (1993) and Weitzman (1992, 1995). In essence, the index scheme places greater value on suites of species which are more different, in which differences may ultimately be measured based on morphological, biochemical, genetic, and/or behavioural characteristics. In this way, a link between biodiversity value and the possible functional characters of organisms is made (Faith 1994; Williams and Humphries 1996). Of the diversity indices developed and in use thus far, taxonomic distinctiveness measures (and their derivations) provide the clearest possible theoretical link with ecosystem function, although precise relationships are not possible to define.

Vane-Wright *et al.* (1991) presented a discussion of the calculation of diversity based on the summation of bifurcation nodes from a hierarchical classification or tree based on evolutionary relationships. This was modified by May (1990) to include the summation of all branches at all nodes of the hierarchical tree, thus taking into account the number of branches at each node (the difference can be significant with partially resolved hierarchies). Solow *et al.* (1993) criticised the method of Vane-Wright *et al.* (1991) in that it did not distinguish between the case in which there are closely related species from the case in which there are not. The diversity measure of Weitzman (1992, 1995; see also Solow *et al.* 1993) focuses on giving a greater value to a set of species that are not closely related by measuring the dissimilarity-distance between any pair of objects in the set:

$$D\{X\} = \max[D(X \setminus s_i) + d(s_i, X \setminus s_i)] \qquad (4.5)$$

where s_i = an element of the mixture of X.

But what can be used as a measure of dissimilarity-distance between any pair of species? It is often difficult to obtain a complete feature analysis (e.g. genetic or morphological) of the species involved or to examine the estimated evolutionary times. One can assume the ultrametric case (Weitzman 1995) in which the evolutionary distances are standardised according to the hierarchical branch with the largest number of nodes and given a value of one at each node. This ensures that the measure of diversity (in essence, taxonomic diversity) depends only on the branching pattern of the hierarchical tree (Faith 1992). The above solution is the same as that of Faith's (1992) minimum spanning path along a cladogram for his measure of phylogenetic diversity. Note that this measure of distance (equation 4.5) does not take into account the size of the set in question (i.e. species richness), which would involve an arbitrary weighting of $d(s_i,s_j)^c$ with $0<c<1$ (Solow *et al.* 1993). In any event, the measure of dissimilarity-distances requires selection of a surrogate set of characteristics and an assumed evolutionary process model for character changes though time (e.g. see Williams and Humphries 1996).

Directions for Indices of Ecosystem Function

As it relates to the assignment of economic values, how do biodiversity indices fare? Certain biodiversity measures, such as species compositional indices, may indeed be relevant and necessary descriptors of biological systems when examining option, existence, or bequest values - in other words, when one is concerned with potential future direct uses of genetic resources or non-use economic values. Such indices may also be relevant when examining present direct use values, such as is the case with bioprospecting (i.e. monetary values associated with genetic resources represented by a particular ecological community). However, as we are concerned with the production of an economically valuable and useful ecological commodity through the functioning of a natural ecosystem, the use of currently developed measures of biodiversity, as briefly outlined above, does not seem appropriate.

The research initiatives of ecologists who continue to examine the relationship between biodiversity and ecosystem function may be misguided in this respect. It seems that an implicit goal of the continuing research which examines the relationship between traditional or more established biodiversity indices and ecosystem function is to further justify the importance of biodiversity itself and to inform resource management decisions for the establishment of appropriate policy for maintaining the health and *functioning* of ecosystems. This is most likely based on an underlying belief by the scientific community that biodiversity *per se* is important, and that conservation efforts can be further appropriately justified if an explicit link with function can be demonstrated. Indeed, it is not uncommon for the link between biodiversity and ecosystem function to be explicitly assumed. Continuing to examine the relationship between biodiversity measures and ecosystem function may be an interesting academic question, but it is not the most useful approach if one's ultimate aim is to explore ecosystem health or function relationships and develop proxy measures for such relationships. One must let information needs direct research efforts.

Traditional diversity measures, such as composition indices, are often seen as indicators of the well-being of ecological systems. Although an increase in the diversity does generally reflect an increase in the complexity of a system, the theoretical connection between diversity and function remains to be revealed (if it at all exists). The indices based on the numbers of species, the numbers of individuals, and/or their distributions do not in a systematic manner sufficiently capture or describe the elements of ecosystem function necessary for use in this project. It is inherently likely that biodiversity is an important defining characteristic of an ecosystem, but one should not set out with the *a priori* notion that biodiversity itself as defined by a particular index should be the critical indicator for all use values, whatever the measure employed. There is a pressing need to more critically examine the use of such indices, particularly as these have become increasingly important to informing policy and conservation efforts. *Can traditional measures of biodiversity be used as indicators of ecosystem function?* - generally speaking, no.

A very general definition of function which also applies to ecosystems is that specifically used to describe a mathematical relation: "a variable quantity regarded in relation to another or others in terms of which it may be expressed or on which its value depends" (*The Oxford English Reference Dictionary* - 2nd edition, 1996). In other words, a formal description of processes defines function. Ecosystem function is often quite differently and often loosely defined, but in essence addresses the relationships between living entities and ecological processes involving materials, energy and information exchange (gene flows and communication; e.g. Lawton and Brown 1993; Martinez 1996; Noss 1990). By defining a function, one must describe both the process in question and the species or groups of individuals interacting with or through that process. However, in defining such function one must avoid the teleological problems associated with implicitly defining "proper function" - employing a general definition such as that which "...addresses the relationships between living entities and ecological processes" with a focus on the simple detection, description and measurement of the processing of materials and energy can help avoid such problems (Martinez 1996).

To begin from the conceptual roots of an ecosystem index, Williams and Humphries (1996) note that functional characters can be used to measure the differences between organisms, including generally the levels of functional anatomy and trophic relationships. In this way, taxonomic distinctiveness measures are the most suitable of the compositional indices discussed as species' character differences are emphasised, albeit in an indirect manner. Further, taxonomic distinctiveness measures often represent genetic or phenotypic suites which may aid in the adaptive ability of ecosystems. However, there are other theoretical avenues concerning indicators to explore before we settle on an index of ecosystem function.

Currently, there are no widely accepted indices of ecosystem function *per se*. Some preliminary work in that direction, however, is evident from the literature. Indeed, there appears to be a growing movement by many ecologists towards a focus on finding general, measurable attributes which characterise the functioning of ecosystems. For example, Martinez (1996) advocates a focus on the conceptualisation of biodiversity itself as "the spatial and temporal variability of the structure and function of living systems." This definition explicitly recognises the importance of establishing proxy measures for ecosystem function.

Holling *et al.* (1995) also advocate focusing on the generalisable functional attributes of ecosystems - species diversity should *mean more than just the number of species or the genetic composition of the assemblage*. Friend and Rapport (1991) noted that indicators which reflect the maintenance of biotic integrity (defined as an "identifiable suite of structural and functional relationships") are appropriate and would be sensitive to the complex nature of ecosystems.

Holling *et al.* (1995) further note that "...syntheses have become possible that suggest that the diversity and complexity of ecological systems can be traced to a small set of biotic and abiotic, or physical, processes each operating over different scale ranges." Furthermore, the resilience of a system (as defined by Holling 1973 as a system property which allows for an ecosystem to maintain the persistence of its elements as an identifiable entity or suite of relationships in the face of disturbance) is traceable to this small set of processes. Indeed, it is becoming evident that it is the ability of the ecosystem in question to maintain the key processes themselves which is important, the particular species involved being of less consequence (Holling 1992). This suggests that conservation efforts should be focused on keeping these functional processes intact and has led to efforts to define and measure resilience in ecosystems (e.g. DeAngelis 1980; Ives 1995; Ludwig *et al.* 1997; Neubert and Caswell 1997; there are various meanings and definitions of resilience employed throughout the ecological literature - Grimm and Wissel 1997 provide a useful review the various uses of ecological stability concepts; see also Grimm 1996; Grimm *et al.* 1992).

Costanza *et al.* (1993) go so far to suggest that "...the variety of different responses displayed by organisms [as a set] to a range of physical environmental changes..." could be used as one measure of functional diversity.

According to Holling (1992) and Holling *et al.* (1995), one simplification which can be made regarding the "multiscale complexity" evident in natural ecosystems is that all processes at all scales undergo a cycle of birth, growth, death and renewal; moreover, they suggest that it is the processes of death (release and disruption of the strong interconnectedness between species) and renewal (reorganisation and release of nutrients and space), each with distinct temporal and spatial attributes, which largely determine functional diversity. What causes the strong connectedness of an ecosystem to break down and how are nutrients and energy made available for renewed growth? It can be asserted that the loss of functional biodiversity is either through the alteration of the key physical processes themselves, or through the excessive removal or harm of the biota. In essence, it is the maintenance of the biogeochemical cycling and the trophic interactions that must be emphasised. Costanza *et al.* (1995) further postulate that the species diversity of a system is a function of the predictability of the environment at and below the time and space scale of the system of interest. Such disturbances at or below the scale in question likely serve to promote the building of resilience in a system (Costanza *et al.* 1995). It is this release on a smaller scale, or healthy *creative destruction*, which is key to the integrity of the system in question (Costanza *et al.* 1995; Holling *et al.* 1995). Note that species diversity as usually defined is not necessarily linked to resilience.

Ecological theory has yet to advance a general predictable or supportable theoretical relationship between properties of ecosystem resilience and ecosystem function, although much has been debated about the possible general relationships between stability and structure (e.g. Hall and Raffaelli 1993; MacAuthur 1955; May 1972, 1973, 1975; Pimm 1982, 1984). Indeed, a general and predictable relationship between properties of resiliency and how ecosystems function is not intuitively obvious and may not exist. Stability statements alone tend to be specific to the ecological situation under consideration and can not be assumed to apply to the larger, more general case (Grimm and Wissel 1997). As it is the intent of this dissertation to focus on the development and use of an index of ecosystem function, measures of resilience were not explicitly considered within the analytical framework. Yet, as properties of resilience are important to consider in conservation in terms of maintaining the system as one with distinct and identifiable characteristics, the measurement of ecosystem resiliency as a potentially fruitful direction for research is left for others.

Where else can one turn to define an index of ecosystem function? This chapter is intent at exploring a general proxy measure of the functional characteristics of species within a community which is supported by theory, calculable using a limited dataset and information base, and applicable to a wide array of ecosystems. An Index of Captured Ecosystem Value (ICEV) will be derived relying on relatively simple descriptions of the structure of the food web of interest and the nature of the trophic relationships between its elements. Information theory will provide the basis for the analysis.

Methods

The Index of Captured Ecosystem Value

It is reasonable to accept that ecological processes and ecosystem function are synonymous terms (Lawton and Brown 1993) or that ecological functions are more generally interactions with ecological processes (Martinez 1996). Functional diversity may be quantified from the type and number of functional groups represented in an ecosystem and the nature of their interactions. Food webs are simply depictions or models of how species in a community interact (e.g. Hall and Raffaelli 1993; Lawton 1989; Pimm 1982; Yodzis 1989), in which interactions with processes are as either consumers or resources, and thus can serve as the basis for the derivation of an index of ecosystem function. This dissertation will present an index of ecosystem function based on a description of the interactions between functional groups in a food web and an analysis of assumed flows within the food web network.

The development of the index will be approached initially from a basis in information theory (e.g. see Kullback 1968). Ulanowicz (1986) and Ulanowicz and Norden (1990) outline the theory behind measures of uncertainty and information content relevant to the analysis of network flows in ecosystems, building on the previous work of MacArthur (1955), Rutledge *et al.* (1976), Ulanowicz (1980), and Hirata and Ulanowicz (1984).

Following Ulanowicz (1986), the uncertainty regarding a specific outcome of an event can be described by

$$H_i = K \log(1/p_i) \tag{4.6}$$

or as

$$H_i = -K \log(p_i) \tag{4.7}$$

where H_i = the uncertainty associated with outcome *i*;

p_i = the probability of outcome *i*; and,

K = a constant (imparts the physical dimensions or "size" to the index; Ulanowicz 1986).

(logarithms are used in the function for convenience to ensure that the index possesses the desirable properties of being non-negative, taking on a value of zero in the absence of all uncertainty, and being additive for the co-occurrence of unrelated outcomes; Aczel and Daroczy 1975; Ulanowicz 1986)

Where a suite of outcomes are possible, one can estimate the *average* uncertainty as

$$H = -K \sum_i p_i \log p_i \tag{4.8}$$

Information concerning events represents a reduction in the uncertainty associated with predicting the outcomes. Thus, one can similarly define the *average gain in information* concerning a suite of possible outcomes as

$$I = K \sum_i p_i \log\left(p_i / p_i^{\bullet} \right) \tag{4.9}$$

where $p_i^{\bullet}$ = the probability of outcome *i* based on an initial assumed distribution; and,

p_i = the probability of outcome i based on new information.

The linking of events is an organisational characteristic of network flows. Applying information theory principles to network flows, one can describe the information concerning the outcome of an event provided by knowledge of a linked or related event as

$$I = K \log\left[p\langle b_j | a_i \rangle \Big/ p\langle b_j \rangle \right] \tag{4.10}$$

where $p\langle b_j | a_i \rangle$ = the probability of b_j given that a_i has occurred; and,

$p\langle b_j \rangle$ = the probability that b_j will occur.

One can then describe the *average mutual information* as presented by Ulanowicz (1986) as

$$A = K \sum_i \sum_j p(a_i, b_j) \log\left[p\langle b_j | a_i \rangle \Big/ p\langle b_j \rangle \right] \tag{4.11}$$

When applied to a network, equation 4.11 measures the information of the flow structure, or specifically, "...how well, on the average, the network articulates a flow event from any one node to affect any other specific locus." (Ulanowicz 1986, p.97). To apply this to an open ecosystem and ensure that the solution is symmetrical with respect to the consideration of either compartment inputs or outputs, one first defines the matrix describing the flows from the ith compartment to the jth compartment, $\mathbf{T}_{ij}$, where i and j range from 0 to n+1 (0 refers to exogenous inputs to the system, while n+1 refers to losses or exports from the system; Ulanowicz and Norden 1990). The average mutual information (Ulanowicz and Norden 1990) then becomes

$$I = \sum_{i,j=0}^{n+1} f_{ij} Q_i \log\left(\frac{f_{ij}}{\sum_k f_{kj} Q_k} \right) \tag{4.12}$$

where $Q_i = \frac{\sum_k T_{ik}}{\sum_{l,m} T_{lm}}$ = the estimated probability that flow passes through compartment i, and;

$f_{ij} = \frac{T_{ij}}{\sum_k T_{ik}}$ = the fraction of total flow through compartment i that also passes through compartment j (or the estimated conditional probability that flow will pass through j given that it has passed through i).

Ulanowicz (1980, 1986) and Ulanowicz and Norden (1990) scale the measure of the average mutual information (I) by the size of the total flow through the system:

$$A = TI \tag{4.13}$$

where A = ascendancy; and,

T = total system throughput, or $\sum_{i,j} T_{ij}$.

System ascendancy, as a measure of the magnitude of the information flow through the network structure of an ecosystem, is effectively an index of ecosystem function. Moreover, it has many useful advantages, such as the ability to be applied to open, far-from-equilibrium systems (Hirata and Ulanowicz 1984; Ulanowicz 1986). However, its use does demand a relatively complete description of the nature and magnitude of the interactions between all species. Furthermore, there are the additional complexities associated with providing realistic and appropriate allowances for temporal and spatial variations in such relationships. As it is the intention of this dissertation to derive an index of ecosystem function which can be applied with a relatively limited knowledge of the ecosystem in question, it is necessary to "back away" from information demands associated with using the average mutual information or system ascendancy concepts strictly as described above. There is also the danger of "misplaced concreteness" in any index value calculated using an unrealistic level of information assumed for the ecosystem.

Let us assume that one *does not* have sufficient information regarding either the size of the system throughput or the associated probabilities of interspecific interactions for a given food web, but *does* have knowledge regarding the species or trophic species involved and whether or not interspecific interactions may occur between any pair of species. In other words,

$$f_{ij} = \frac{1}{l_i} \tag{4.14}$$

$$Q_i = \frac{l_i}{L} \tag{4.15}$$

$$\sum_k f_{kj} Q_k = \frac{l_j}{L} \tag{4.16}$$

where l_i = the total number of species i forward links with other species (i.e. the total number of species or trophic species which use species i as a resource or as prey);

l_j = the total number of species j backwards links with other species (i.e. the total number of species or trophic species which species j uses as a resource or as prey); and,

L = the total number of links between all species in the set.

Because one is assuming that we do not have sufficient information regarding the probabilities associated with any pair of interactions, all interspecific interactions are given an equal probability of occurring *as long as a link is know to exist.* Thus, assuming maximum uncertainty regarding the *strength* of interspecific interactions, equation 4.12, which defines the average mutual information, becomes

$$I = \sum_{i,j=0}^{n+1} \left(\frac{1}{l_i}\right)\left(\frac{l_i}{L}\right) \log\left(\frac{\frac{1}{l_i}}{\frac{l_j}{L}}\right) \tag{4.17}$$

which reduces to

$$I = -\sum_{i,j=0}^{n+1} \left(\frac{1}{L}\right) \log\left(\frac{l_i l_j}{L}\right) \tag{4.18}$$

Equation 4.18 thus defines the average gain in information regarding the occurrence of interspecific interactions or flows through the given food web given that one has knowledge of the interactions which do occur. This can be contrasted to the case in which there is no knowledge of the nature of the interactions, and there is simply *uncertainty* or complexity associated with a given set of species (the uncertainty associated with a given set of species is defined by the Shannon index of diversity, in which $H = -\sum_{i=1}^{s} P_i \ln P_i$, where P_i is the proportional contribution of species i to the total number of individuals or biomass in a set of s species; e.g. see Begon *et al.* 1986; Ulanowicz 1986).

Recall that it is the intent of this dissertation to develop an index of ecosystem function which will indicate the extent of that ecosystem function value which is captured through extraction of a natural resource, specifically using the coral reef fisheries of Jamaica as a case study. Equation 4.18 can be adapted for such a purpose.

Let us consider predator compartment j and prey or resource compartment i. The extraction of a species from compartment j (e.g. through a fishery) was ultimately made possible through flow from the set of compartments consisting of R to which compartment j is backward linked. The mutual information concerning the flow to compartment j given knowledge of the existence of the link with any one compartment i is

$$-\log\left(\frac{l_i l_j}{L}\right) \tag{4.19}$$

One can then subsequently consider the mutual information for all pairs of compartments i and j linked, considering *only* the set of compartments consisting of P from which natural resources are extracted, with each compartment j within the set P having a unique set R of species which it utilises as a resource. A conceptual example of this model is shown in Figure 4.2.

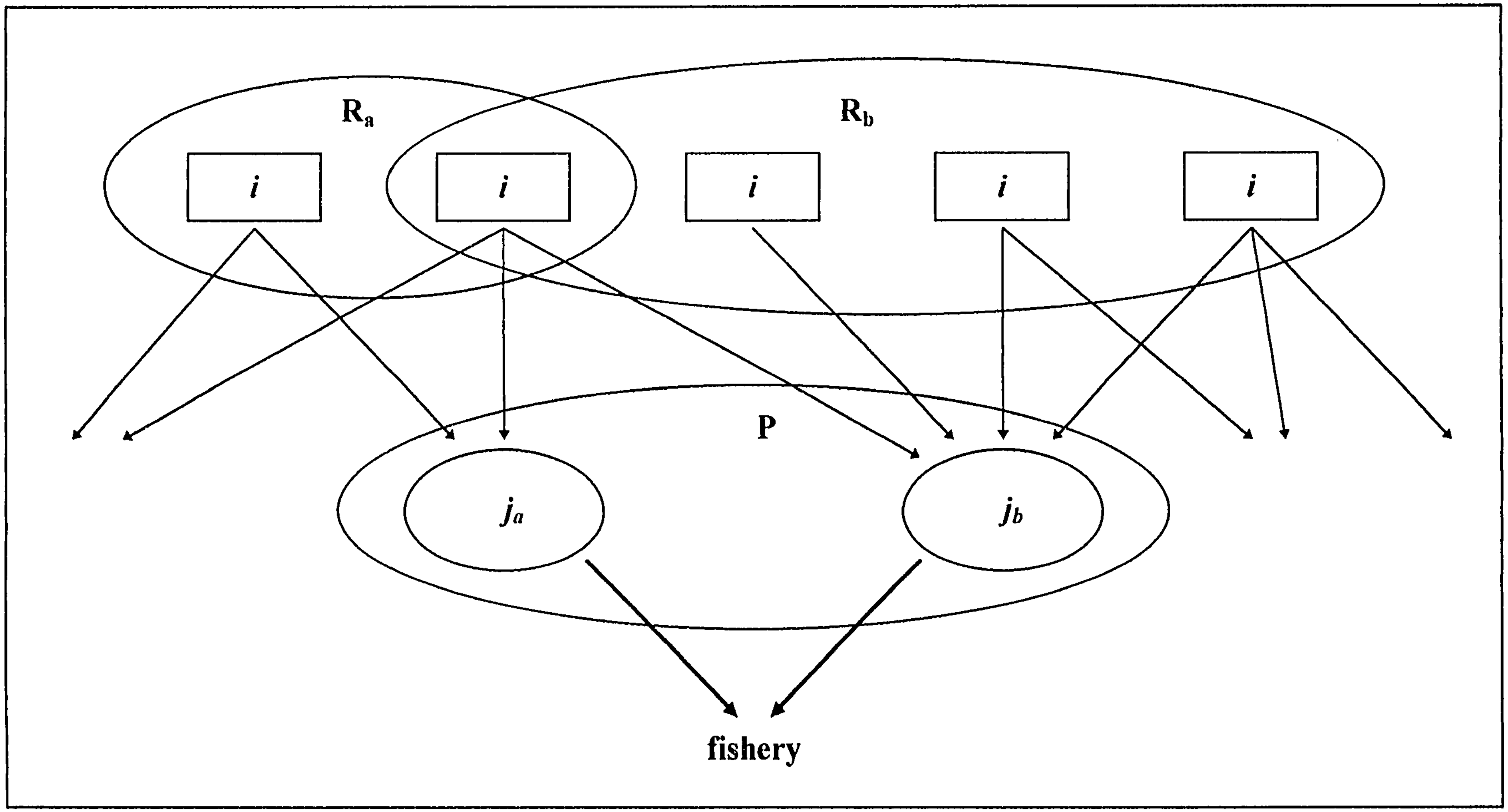

Figure 4.2. Conceptual trophic interactions model. The sum of the number of all forward links from prey species *i* is represented by l_i. The sum of the number of all backward links from predator species *j* is represented by l_j. The set of all *i* species utilised by species *j* defines set *R*. The set of all *j* species captured in a fishery defines the set *P*.

In allowing the reduction of the average mutual information index to equation 4.18, however, we assumed no knowledge concerning the *size* of the network flows. The discussion so far similarly does not consider the magnitude of the economic extraction of the natural resource. In order to factor in the magnitude of this extraction, the total biomass (B) extracted in a given period is treated similar to Ulanowicz (1980, 1986) and Ulanowicz and Norden's (1990) consideration of total system throughput. Thus, using equation 4.18, we arrive at the Index of Captured Ecosystem Value (ICEV):

$$ICEV = -B\sum_{i,j}\left(\frac{1}{L_{ij}}\right)\log\left(\frac{l_i l_j}{L}\right) \tag{4.20}$$

where L_{ij} = the total number of links between all i,j pairs considered, in which all i,j pairs are those linked within the food web structure for which compartment j is utilised or extracted as a natural resource (i.e. within set P).

A General Food Web Model for Caribbean Coral Reefs and the Identification of Trophic Species

To apply the index, it is necessary to develop a typology of functional species groups based on the variety of interactions with ecological processes. At the most disaggregated level, individual taxonomic species or ontogenetic stages within any one species may be distinguished as functional groups. More relevant for food web studies, however, trophic species are identified. Trophic species are groups of taxonomic species or ontogenetic stages of species that seem to share both the same set of trophic consumers and resources, or predators and prey (Briand and Cohen 1984; Pimm *et al.* 1991; this is in contrast to the guild concept, in which taxa are grouped according to their use of resources alone - see Simberloff and Dayan 1991). It is noted that through the use of trophic species rather than individual taxonomic species or ontogenetic stages of species, biases introduced through the specification of functional groups may be reduced as more consistent criteria are utilised (e.g. taxonomic distinctions are often made based on variable morphological and phylogenetic bases; Martinez 1994). Differences in the necessarily subjective distinctions made by various researchers on what constitutes a significant specific predator-prey interaction can be reduced with the "lumping" into aggregate trophic species (Briand and Cohen 1984).

Information regarding the food web structure and species composition of Caribbean coral reefs is widely scattered. Information specific to Jamaica is limited - some community composition information exists, but is focused on resolving patterns for sessile benthic species (e.g. CARICOMP 1997; Goreau 1959; Goreau and Goreau 1973; Liddell and Ohlhorst 1981, 1987; Liddell *et al.* 1984). By far, most studies are isolative and species-specific, thus of little help in constructing a food web model. However, Opitz (1996) was able to construct a steady-state model for a generalised Caribbean coral reef ecosystem, in the process compiling a taxon interaction matrix based on assumed diet compositions derived from the numerous literature sources with an emphasis on fish species. Opitz (1996) used the comprehensive study of fish diets by Randall (1967) for the US Virgin Islands, supplemented with fish species listings provided in Fischer (1978), as the "scaffold" to develop the matrix which identified the fish taxa and key flora, invertebrate and vertebrate taxa on which they directly depend. The diet composition of non-fish taxa was developed from numerous other published sources relevant to the Caribbean. The taxa reported by Opitz (1996) are shown in Appendix A.

Trophic species were identified from the diet composition matrix presented in Opitz (1996). Before the analysis was applied, all non-zero interaction coefficients in the diet composition matrix were recoded to 1, thus recording predator-prey relationships as either "on" or "off". It was felt necessary to generalise the matrix in such a way because: (i) Opitz (1996) had to employ fairly rough estimation techniques to arrive at specific coefficient values, at times when only qualitative data were available regarding diet; and, (ii) the quantitative data for the matrix were based using information largely from Randall (1967) for the US Virgin Islands. It was not reasonable to maintain the interaction coefficients to three significant figures as presented in Opitz (1996) and apply the data to model present-day Jamaican coral reefs given the current lack of information regarding the food webs of Jamaican reefs or other regions of the Caribbean. This is not to say that use of this source for the development of a model applicable to Jamaica is invalid - one is simply relaxing the precision that is claimed to reflect the application. Use of the more generalised information regarding interspecific interactions for the purposes of this study is prudent.

There are caveats and limitations associated with food web statistics, particularly if comparisons are to be made between webs (e.g. Fonseca and John 1996; Paine 1988; see Discussion below for a more detailed presentation of the issues). One significant problem associated with using measures such as connectance, ascendancy, or average mutual information (or other statistics which are calculated based on a resolved food web structure or defined network flow) is their sensitivity to web resolution. This problem is lessened through use of an averaging function in the ICEV (as well as in the average mutual information index as previously formulated) to facilitate comparisons between food webs of different types. However, the index is still expected to be sensitive to alternative species grouping or clustering methodologies. Applying clustering methodologies to arrive at alternative trophic groupings other than the trophic species as identified here in order to explore the sensitivity of the ICEV to resolution is left to subsequent investigations.

For five coral reef-based artisanal fisheries (China net, trap, hand line, palanca, and speargun fishing), the capture of ecosystem function value as measured by the ICEV was explored using the data from the 1996 catch and effort survey conducted by the Fisheries Division (Ministry of Agriculture, Government of Jamaican; see Chapter 2 for data source description). Individual survey samples were eliminated from the set if they failed to record the composition of individual catches by genus (at a minimum) and by weight. Non-finfish species were ignored for the present analysis.

The technical coefficient associated with each fishery is represented as

$$a_{ij} = \frac{ICEV_{ij}}{V_j} \tag{4.21}$$

where a_{ij} = technical coefficient defining the production relationship between the ecological commodity input i (function value as measured by the ICEV) and the fishery j (Jamaican dollars);

$ICEV_{ij}$ = flow of ecological commodity input i to fishery j; and,

V_j = gross output value of fishery j (Jamaican dollars).

For each finfish taxon represented in the catch and effort surveys, the market price for the year 1996 and the ICEV value (equation 4.20) per kg, as well as the corresponding technical coefficient (equation 4.21) associated with each individual taxon, was calculated.

Similarly, the technical coefficient (equation 4.21) was calculated for all individual fishing efforts, considering the total ICEV and monetary value of each "bundle" of fishes. In order to explore whether or not fisheries using distinct technologies (i.e. China net, trap, hand line, palanca, and speargun fishing) lead to ecosystem function value as measured by the ICEV (equation 4.20) being *equally valued* by the market, a non-parametric Kruskal-Wallis (ANOVA by ranks) test was conducted to determine if the mean technical coefficient (equation 4.21) of individual fishing efforts within any one fishery was significantly different from any other fishery. All statistical procedures were conducted using SPSS® (version 7.5 for Windows) statistical software package.

Results

Identification of Trophic Species and the Information Provided through Knowledge of the Food Web Linkages

The identification of trophic species resulted in the reduction of the 250 original taxonomic groups as represented in the diet composition matrix of Opitz (1996; Appendix A) to 246 groups. Only four pairs of species of fishes shared both the same set of predators and prey as recorded in the matrix: i) *Diodon holocanthus* (long-spine porcupinefish) and *Diodon hystrix* (spot-fin porcupinefish) from the family Diodontidae, now designated as *Diodon* spp. trophic species; ii) *Gramma loreto* (royal gramma) and *Gramma melacara* (blackcap basslet) from the family Grammitidae, now designated as *Gramma* spp. trophic species; iii) *Kyphosus incisor* (yellow sea chub) and *Kyphosus sectatrix* (Bermuda sea chub) from the family Kyphosidae, now designated as *Kyphosis* spp. trophic species; and, iv) *Pomacanthus arcuatus* (grey angelfish) and *Pomacanthus paru* (French angelfish) from the family Pomacanthidae, now designated as *Pomacanthus* spp. trophic species. All other trophic species are as listed in Appendix A.

Table 4.1 records the total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species. For the complete trophic interaction matrix, the total number of links in the community (L) was 3317. Table 4.2 shows the market prices and ICEV value per kg for each finfish species recorded from Jamaican artisanal coral reef fisheries (specifically including speargun, palanca, hand line, trap, and China net fishing) from 1996 catch and effort surveys, as well as the corresponding technical coefficient (equation 4.21) associated with each individual taxon. In cases where the genus but not the species was recorded in the survey, or the species was not represented in the food web model, corresponding index and technical coefficient values were taken as the average for the genus as represented in the food web. Typical market prices are estimates of the price obtained for the year 1996 (unpublished data of the Fisheries Division, Ministry of Agriculture, Government of Jamaica). The values of the technical coefficients have a mean of 0.010 ($n = 61$), a maximum of 0.017 and a minimum of 0.004.

Table 4.1. Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
organic detritus	49	0
benthic algae/ spermatophytes	68	0
symbiotic algae	29	0
phytoplankton	12	0
decomposers/ microfauna	23	2
zooplankton	103	1
sponges	28	1
hydrozoans	16	4
sea fans	14	4
sea anemones	17	3
stony corals	16	4
bryozoans	8	5
sipunculid worms	22	3
priapuloids	1	1

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
chitons	30	6
gastropods	84	25
bivalves	64	3
scaphopods	17	1
squids	22	4
octopuses	28	9
polychaetes	82	17
echiuroids	3	3
pycnogonids	2	7
barnacles	9	3
stomatopods	45	14
amphipods	55	4
tanaids	19	2
isopods	45	2

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
shrimps	110	20
spiny lobsters	10	3
scyllarid lobsters	10	3
hermit crabs	40	5
crabs	110	18
hemichordates	7	3
asteroids	10	17
ophiuroids	39	4
echinoids	41	8
holothurians	14	3
tunicates	18	4
sea turtles	3	12
sea birds	1	2

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Acanthurus bahianus	23	2
Acanthurus chirurgus	19	4
Acanthurus coeruleus	18	4
Antennarius multiocellatus	8	3
Antennarius striatus	8	2
Phaeoptyx conklini	11	5
Apogon maculatus	10	6
Hypoatherina harringtonensis	26	1
Atherinomorus stipes	25	3
Aulostomus maculatus	15	29
Balistes vetula	4	18
Canthidermis sufflamen	4	4
Melichthys niger	12	9
Platybelone argalus argalus	13	2
Strongylura timucu	13	4
Tylosurus acus acus	7	12
Tylosurus crocodilus crocodilus	7	6
Entomacrodus nigricans	8	3
Ophioblennius atlanticus	16	3
Parablennius marmoreus	7	6
Scartella cristata	14	3

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Bothus lunatus	3	6
Bothus ocellatus	5	5
Caranx bartholomaei	8	8
Caranx latus	8	8
Caranx ruber	11	32
Decapterus punctatus	10	1
Oligoplites saurus	15	2
Selar crumenophthalmus	11	2
Seriola dumerili	8	9
Trachinotus falcatus	8	5
Trachinotus goodei	7	7
Carcharhinus acronotus	5	100
Carcharhinus falciformis	2	152
Carcharhinus leucas	1	16
Carcharhinus limbatus	4	20
Carcharhinus longimanus	2	16
Carcharhinus perezi	4	192
Galeocerdo cuvier	0	166
Negaprion brevirostris	5	105
Rhizoprionodon porosus	5	101
Chaetodon aculeatus	9	5

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Chaetodon capistratus	10	6
Chaetodon sedentarius	10	5
Chaetodon striatus	10	6
Amblycirrhitus pinos	9	7
Lambrisomus guppyi	3	4
Lambrisomus nuchipinnis	8	8
Harengula clupeola	21	2
Harengula humeralis	20	3
Jenkinsia lamprotaenia	33	2
Opisthonema oglinum	18	4
Heteroconger halis	13	1
Dactylopterus volitans	3	6
Dasyatis americana	7	16
Chilomycterus antennatus	4	5
Diodon spp.	8	5
Inermia vittata	12	2
Anchoa hepsetus	22	3
Anchoa lyolepis	21	3
Chaetodipterus faber	3	10
Fistularia tabacaria	5	2

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Eucinostomus argenteus	10	9
Gerres cinereus	10	10
Ginglymostoma cirratum	1	111
Coryphopterus glaucofraenum	7	5
Gnatholepis thompsoni	4	4
Gobiosoma evelynae	4	1
Gramma spp.	8	1
Rypticus saponaceus	10	6
Anisotremus surinamensis	3	12
Anisotremus virginicus	10	16
Haemulon album	10	19
Haemulon aurolineatum	24	11
Haemulon carbonarium	19	15
Haemulon chrysargyreum	18	14
Haemulon flavolineatum	23	16
Haemulon macrostoma	10	2
Haemulon parra	10	12
Haemulon plumieri	11	14
Haemulon sciurus	11	17
Hemiramphus balao	13	3

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Hemiramphus brasiliensis	14	2
Holocentrus ascensions	15	5
Holocentrus coruscus	14	2
Neoniphon marianus	14	3
Holocentrus rufus	15	10
Sargocentron vexillarium	14	10
Myripristis jacobus	16	6
Kyphosus spp.	3	1
Bodianus rufus	5	11
Clepticus parrae	13	1
Halichoeres bivittatus	13	12
Halichoeres garnoti	14	11
Halichoeres maculipinna	11	13
Halichoeres poeyi	6	13
Halichoeres radiatus	12	9
Xyrichtys novacula	11	7
Xyrichtys splendens	9	5
Lachnolaimus maximus	4	8
Thalassoma bifasciatum	12	11
Lutjanus analis	5	24

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Lutjanus apodus	5	46
Lutjanus cyanopterus	5	17
Lutjanus griseus	6	5
Lutjanus jocu	5	47
Lutjanus mahagoni	12	14
Lutjanus synagris	5	28
Ocyurus chrysurus	7	9
Malacanthus plumieri	4	15
Tarpon atlanticus	3	2
Aluterus schoepfii	4	3
Aluterus scriptus	4	9
Cantherhines macrocerus	4	6
Cantherines pullus	17	16
Monacanthus ciliatus	16	12
Mugil curema	12	2
Mulloidichthys martinicus	13	15
Pseudupeneus maculatus	16	14
Echidna catenata	12	2
Lycodontis moringa	14	2
Gymnothorax vicinus	5	7
Aetobatus narinari	4	107

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Ogcocephalus nasutus	5	7
Myrichthys breviceps	11	3
Myrichthys ocellatus	11	5
Ophichthus ophis	4	2
Opisthognatus aurifrons	9	1
Opisthognatus maxillosus	9	5
Opisthognatus whitehurstii	2	5
Acanthostracion polygonius	3	6
Acanthostracion quadricornis	3	12
Lactophrys trigonus	4	15
Lactophrys bicaudalis	3	9
Lactophrys triqueter	3	15
Pempheris schomburgki	9	3
Centropyge argi	1	2
Holacanthus ciliaris	4	6
Holacanthus tricolor	5	3
Pomacanthus spp.	3	9
Abudefduf saxatilis	12	7
Abudefduf taurus	10	6
Chromis cyanea	12	1

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Chromis multilineata	14	1
Microspathodon chrysurus	11	10
Stegastes fuscus	15	16
Stegastes leucostictus	11	15
Stegastes planifrons	13	16
Stegastes variabilis	11	14
Priacanthus arenatus	4	9
Heteropriacanthus cruentatus	10	10
Scarus coelestinus	15	14
Scarus iserti	20	1
Scarus guacamaia	15	1
Scarus taeniopterus	18	2
Scarus vetula	16	4
Sparisoma aurofrenatum	21	5
Sparisoma chrysopterum	16	1
Sparisoma radians	22	1
Sparisoma rubripinne	15	2
Sparisoma viride	15	3
Equetus lanceolatus	10	3
Equetus punctatus	10	10
Odontoscion dentex	11	5

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Pareques acuminatus	10	6
Euthynnus alletteratus	7	11
Scomberomorus cavalla	7	40
Scomberomorus regalis	7	30
Scorpaena brasiliensis	4	4
Scorpaena grandicornis	11	2
Scorpaena inermis	3	4
Scorpaena plumieri	6	9
Scorpaenodes caribbaeus	10	4
Alphestes afer	6	7
Cephalopholis cruentata	14	22
Cephalopholis fulva	15	22
Epinephelus adscensionis	5	10
Epinephelus guttatus	5	21
Epinephelus itajara	3	6
Epinephelus striatus	5	67
Hypoplectrus aberrans	10	8
Hypoplectrus chlorurus	10	6
Hypoplectrus nigricans	11	8
Hypoplectrus puella	11	9
Mycteroperca bonaci	4	2

Table 4.1 (continued). Total number of links with prey (l_j) and the total number of links with predators (l_i) for each trophic species.

trophic species	l_i	l_j
Mycteroperca tigris	5	30
Mycteroperca venenosa	5	54
Paranthias furcifer	10	3
Serranus tigrinus	13	4
Archosargus rhomboidalis	2	7
Calamus bajonado	4	6
Calamus calamus	4	9
Calamus pennatula	4	11
Diplodus argentus caudimacula	10	4
Sphyraena barracuda	4	47
Sphyraena picudilla	5	3
Sphyrna lewini	1	43
Sphyrna tiburo	5	104
Synodus foetens	11	2
Synodus intermedius	7	26
Synodus synodus	13	1
Canthigaster rostrata	4	17
Sphoeroides spengleri	11	17
Mustelus canis	5	109

Table 4.2. Market price and ICEV value per kg for each finfish species recorded from Jamaican artisanal fisheries (speargun, palanca, hand line, trap, and China net) from 1996 catch and effort surveys, as well as the defined technical coefficient.

trophic species	price (J$ kg^{-1})	ICEV (kg^{-1})	technical coefficient (J^{-1})
Acanthurus bahianus	88	1.46	0.017
Acanthurus chirurgus	88	1.08	0.012
Acanthurus coeruleus	88	1.33	0.015
Aulostomus maculatus	88	0.95	0.011
Balistes vetula	88	0.69	0.008
Scartella cristata	110	1.23	0.011
Bothus lunatus	132	1.36	0.010
Caranx bartholomaei	110	1.40	0.013
Caranx latus	110	1.16	0.011
Caranx ruber	110	0.67	0.006
other *Caranx* spp.	110	0.87	0.008
Selar crumenophthalmus	110	1.73	0.016
Seriola dumerili	110	1.78	0.016
Chaetodon capistratus	88	1.47	0.017
Chaetodon striatus	88	1.27	0.014

Table 4.2 (continued). Market price and ICEV value per kg for each finfish species recorded from Jamaican artisanal fisheries (speargun, palanca, hand line, trap, and China net) from 1996 catch and effort surveys, as well as the defined technical coefficient.

trophic species	price (J$ kg^{-1})	ICEV (kg^{-1})	technical coefficient (J^{-1})
Diodon spp.	88	1.02	0.012
Gerres cinereus	88	0.83	0.009
other *Gerres* spp.	88	0.80	0.010
Haemulon album	110	0.81	0.007
Haemulon carbonarium	110	0.87	0.008
Haemulon chrysargyreum	110	0.70	0.006
Haemulon flavolineatum	110	0.68	0.006
Haemulon macrostoma	110	1.40	0.013
Haemulon parra	110	0.78	0.007
Haemulon plumieri	110	0.78	0.007
Haemulon sciurus	110	0.69	0.006
other *Haemulon* spp.	110	1.05	0.010
Holocentrus ascensions	88	0.91	0.010
other *Holocentrus* spp.	88	0.82	0.009
Neoniphon marianus	88	1.23	0.014

Table 4.2 (continued). Market price and ICEV value per kg for each finfish species recorded from Jamaican artisanal fisheries (speargun, palanca, hand line, trap, and China net) from 1996 catch and effort surveys, as well as the defined technical coefficient.

trophic species	price (J$ kg^{-1})	ICEV (kg^{-1})	technical coefficient (J$ $^{-1}$)
Holocentrus rufus	88	0.79	0.009
Kyphosus spp.	132	1.69	0.013
Lutjanus analis	132	0.94	0.007
Lutjanus apodus	132	0.66	0.005
Lutjanus griseus	132	1.12	0.008
Lutjanus jocu	132	0.62	0.005
Lutjanus mahagoni	132	1.07	0.008
Lutjanus synagris	132	0.63	0.005
other *Lutjanus* spp.	132	0.76	0.006
Ocyurus chrysurus	132	0.88	0.007
Mugil curema	132	1.46	0.011
Mulloidichthys martinicus	132	0.70	0.005
Priacanthus arenatus	176	0.94	0.005
Heteropriacanthus cruentatus	176	0.82	0.005
Scarus coelestinus	110	0.70	0.005

Table 4.2 (continued). Market price and ICEV value per kg for each finfish species recorded from Jamaican artisanal fisheries (speargun, palanca, hand line, trap, and China net) from 1996 catch and effort surveys, as well as the defined technical coefficient.

trophic species	price (J\$ kg^{-1})	ICEV (kg^{-1})	technical coefficient (J\$$^{-1}$)
Scarus guacamaia	110	1.69	0.015
Scarus taeniopterus	110	1.58	0.014
Scarus vetula	110	1.45	0.013
other *Scarus* spp.	110	1.11	0.010
Sparisoma spp.	110	1.80	0.016
Scomberomorus cavalla	176	0.89	0.005
Scomberomorus regalis	176	0.90	0.005
Epinephelus itajara	132	1.70	0.013
Epinephelus striatus	132	0.50	0.004
Mycteroperca bonaci	132	2.19	0.017
other *Mycteroperca* spp.	132	0.77	0.006
Paranthias furcifer	132	1.33	0.010
Serranus tigrinus	132	1.16	0.009
Sphyraena barracuda	132	0.72	0.005
Sphyraena picudilla	132	1.58	0.012
Synodus intermedius	88	0.93	0.011

Fishery Landings and ICEV Values

For each of the five coral reef-based fisheries technologies examined in this investigation, summary statistics of the values of the technical coefficients (equation 4.21) calculated using the results of individual fishing efforts are shown in Table 4.3. The results of the non-parametric Kruskal-Wallis (ANOVA by ranks) test were highly significant ($\chi^2 = 106$; df = 4; $p < 0.001$), indicating that fisheries based in different technologies capture different levels of ecosystem function value per dollar of catch. In other words, "surplus value" is created. This surplus value appears to be rooted in the observation that certain fisheries themselves rely on species representing the capture of a greater or lesser amount of ecosystem information value - in other words, fishing species which are more or less "connected" within the food web than is the case with other fishing technologies, and these differences not being reflected in market prices. This interpretation is supported by an analysis of the average mutual information associated with individual species captured by each fishery (Kruskal-Wallis test; $\chi^2 = 80$; df = 4; $p < 0.001$), indicating that the captured ecosystem value is fundamentally different among fisheries utilising different technologies.

Table 4.3. Summary statistics of the values of the technical coefficients (J$^{-1}; equation 4.21) associated with China net, trap, hand line, palanca, and speargun fishing.

fishery	n	mean	SD	minimum value	maximum value
China net	47	0.00698	0.00104	0.00529	0.01088
trap	279	0.00887	0.00190	0.00407	0.01659
hand line	31	0.00683	0.00137	0.00507	0.01027
palanca	16	0.00632	0.000796	0.00464	0.00791
speargun	20	0.00771	0.00220	0.00543	0.01511
total	393	0.00832	0.00197	0.00407	0.01659

Discussion

It is evident that Jamaican artisanal coral reef fisheries based in the use of distinct technologies capture ecosystem information values which are not, to the same extent, proportionately reflected in the market values obtained for the catch. Certain fisheries appear to exploit species which are directly and indirectly more intimately connected within the coral reef food web as measured by the ICEV, and thus rely on the capture of greater ecosystem function value for each unit of monetary value as measured by the market. Further detail regarding these patterns will require subsequent analyses, likely requiring an expanded dataset with improved representation by certain technologies (see Table 4.3).

This study was primarily interested in deriving and exploring the application of an alternative theory of value which can be applied to the extraction of a renewable natural resource. This objective, I believe, was successfully accomplished. This analysis demonstrates that if one considers biophysical values (i.e. the biophysical *contribution* of the coral reef ecosystem to fisheries production), distortions between market values and "supply-side" values become evident.

What does this creation of "surplus value" mean? In effect, we are not valuing the ecosystem contributions to the creation of final products to the same extent within a given sector, or perhaps between sectors of the economy. We are not consistently considering ecosystem function values.

As has been the tone throughout the above derivation and demonstrated use of the ICEV, I by no means assert that it is the only correct index to use in the given context. The theoretical development and use of indicators and indices is still very much an art rather than a hard science, with much room left for future developments and improvements. The remainder of this discussion will be devoted to providing comment and direction with that in mind.

Perturbations and the Dynamics of Coral Reef Ecosystems

The food web model used for the derivation of the ICEV is static. It does not consider or accommodate dynamic changes to the structure or functioning of the ecosystem which may occur through either human-induced or natural influences. Coral reefs are not static entities, but dynamic and evolving. It may indeed be possible to modify the structure of the ICEV or the method of its calculation to accommodate food web dynamics, yet to do so would require a level of scientific knowledge concerning coral reef ecosystems which may not be currently sufficient. Information concerning ecosystem impacts and processes have begun to emerge, however, providing direction for possible future index developments.

The General Nature and Effect of Impacts on Coral Reef Functioning

On a world-wide basis, the most significant negative impacts on tropical coral reefs are believed to be due to overfishing (leading to fish population declines, loss of species diversity, and altered ecosystem states), sedimentation (leading to inhibition of coral growth, reduced larval settlement success and survivorship of benthic species, and loss of species diversity), and nutrient enrichment (leading to algal overgrowth, eutrophication, reduced larval settlement success and survivorship of coral species, and loss of species diversity; Roberts 1993). Problems of "overfishing" include biological overextraction of the resource, as well as destructive fishing practices through the use of explosives and poisons (McManus *et al.* 1997; Pauly *et al.* 1989). Sedimentation and nutrient enrichment stressors are most often linked to inappropriate land development, coastal infilling, dredging, agricultural runoff, and poor waste disposal practices (Pastorok and Bilyard 1985; Rogers 1990). In addition, coral bleaching events appear to have increased significantly in recent years, most likely attributable to a rise in sea temperatures, although other causal factors such as high exposures to UV radiation, disease, sedimentation, and nutrient enrichment may also be evident (Brown and Ogden 1993; Goreau 1992).

The coral reefs of Jamaica have been impacted by poor sewage and solid waste disposal practices, sedimentation from coastal construction, habitat loss from coastal infilling and mangrove destruction, intensive exploitation of the fisheries resources, and physical damage from boat anchors, as well as scuba and snorkelling activities. Further impacts have occurred by significant coral bleaching events, the massive die-off of the long spined black sea urchin (*Diadema antillarum*) in the early 1980s, Hurricane Allen in 1980, and Hurricane Gilbert in 1988 (Goreau 1992; Hughes 1994; Hughes *et al.* 1985; Kaufman 1983; Lapointe 1989; Liddell and Ohlhorst 1986; Woodley *et al.* 1981).

In terms of more broadly defined community compositional characteristics, the coral reefs of Jamaica have been documented to have undergone a substantial phase shift in recent years. Hughes (1994; see also Done 1992; Hughes 1989; Kaufman 1983; Liddell and Ohlhorst 1986; Woodley *et al.* 1981) outlined the processes that led to the changes that have occurred: i) chronic overfishing since the 1960's, dramatically reducing the presence of large predatory species and herbivores by preventing sufficient numbers of local fishes from reaching maturation; ii) extensive physical damage to coral species (primarily branching corals and soft corals, although the direct effect on the densities, distributions, and behaviours of fish species was also dramatic; Kaufman 1983; Woodley *et al.* 1981) from Hurricane Allen in 1980 (the negative effects to be later reinforced by Hurricane Gilbert in 1988); and, iii) mass mortality of the echinoid *Diadema antillarum* in 1983 due to the rapid spread of a species-specific pathogen (mortality rates close to 100% in some locales; Hughes *et al.* 1985). The destruction of the coral-dominated benthos by the hurricane and the resulting increase in available substrate, in concert with the severely reduced populations of herbivorous fishes and sea urchins, lead to the transition to an algal-dominated benthic community. Such multiple stable states may be possible through a variety of mechanisms, with three in particular likely being most relevant to the Jamaican case study (Knowlton 1992): i) the inverse density dependence of *D. antillarum* reproduction (depensation or the Allee effect) resulting in decreasing reproduction rates below a critical echinoid population threshold; ii) the inability of the large branching corals to return to their former high population levels following hurricane damage as predators can now maintain the population levels low; and, iii) the shift from palatable to unpalatable algae with the rapid growth and eventual domination of algae within the community, reducing any predatory response which it may illicit (i.e. an increase in the population level of herbivorous fishes).

Such "simple tales" of community dynamics must be interpreted with caution. The dominance of algae within Jamaican shelf benthic communities is believed to persist today (e.g. CARICOMP 1997; Sullivan and Chiappone 1994), although the research is less than sufficient to state with any degree of certainty whether or not any "recovery" to a state of greater coral dominance is or is not occurring. The combined effects of physical disturbance and interspecific competition may even ultimately enhance species diversity, thus perhaps having an overall positive effect (Hughes 1989). The scientific community does not yet sufficiently understand the mechanisms through which such phase shifts operate or their significance to the functioning of coral reef ecosystems. As noted by Done (1992), it is through an understanding of the ecological processes that we will begin to understand the fate of coral reefs within a human-dominated or human-influenced ecosystem.

Much of the research concerning the ecology of Jamaican coral reefs has occurred only over the last 40 years. Interpreting the significance of the ecological changes which have occurred over that relatively limited time frame is difficult (Jackson 1997; Roberts 1993). We have likely seen dramatic changes in the composition of coral reef communities (and more generally within the tropical marine environment) in the Caribbean over the last 500 years, particularly as evidenced by the decline in the larger vertebrate marine species (Jackson 1997). The scientific definition of what constitutes a "normal" coral reef community, and the subsequent extrapolation of the composition and magnitude of potential fisheries harvests under "optimal" or "normal" ecological conditions, constitutes a problem of "shifting baselines" (Pauly 1995). Coral reef ecosystem models may be highly specific to a moment in time and irrelevant concerning questions of sustainable use over longer time frames.

How does one distinguish short-term, perhaps anomalous phenomena from longer-term trends in ecological structures and processes which may be of greater significance? This is certainly not a question which the relatively young field of coral reef ecology is able to address, but one need not necessarily worry about addressing such a question. Given the ever-increasing economic dependence of humans on coral reefs, the argument could be made that it is the current state of reefs, the services that they provide, and the current dynamics measured on short-term ecological scales which are of concern (J. Woodley, as quoted in Roberts 1993). Historical trends may be of less relevance.

The Impacts of Fishing on Coral Reef Ecosystems

The food web model employed here, and thus the ICEV, is not able to even more narrowly take into account the direct and indirect dynamic effects of fishing on ecosystem structure and functioning. Considering it is the capture of ecosystem function value through fishing which is the immediate concern for this study, this warrants the greatest attention in future related research. As a more obvious outcome, fishing will result in a decline in the biomass of the fished species, often targeting a specific size category, and affect the fished species' behaviours, distributions, and reproductive abilities (see reviews by Jennings and Lock 1996; Russ 1991; Jamaican study by Koslow *et al.* 1988). Of course, there are also notable physical damages to coral reef habitats which may compromise the productivity or integrity of the coral reef community as a direct result of the chosen method of fishing. This includes damage from traps, nets, and divers, as well as through the use of explosives and poisons (Jennings and Lock 1996; Jennings and Polunin 1996). But how will the components of the ecosystem indirectly respond to fishing pressures? What of the dynamic responses to the removal of individuals from the community?

Documentation is biased in favour of heavily overfished systems, with little consideration for less catastrophic ecosystem effects and often under less than sufficiently experimentally controlled conditions, making mechanistic interpretation of the results difficult (Jennings and Lock 1996; Russ 1991). The effects of fishing on ecosystems include shifts in the composition of the community. Changes in the relative numbers or biomass of species may occur, acting through trophic interactions (either an increase or a decrease in the size or importance of particular interactions) directly or indirectly linked to the species targeted in fisheries (Jennings and Lock 1996). This may in turn lead to changes in the nature and strengths of the predator-prey relationships, in the structure of the ecosystem, and in the functional roles of individual species (Jennings and Polunin 1996; Roberts 1995).

Figure 4.3 depicts theoretical shifts in the importance of the relationships between tropical coral reef community trophic groups as indirect responses to fishing pressures. Most notable is the effects of reductions from fishing in the populations of particular trophic species (piscivorous fishes, herbivorous fishes, and invertebrate-feeding fishes) key to maintaining the overall composition of the system. For example, the intensive removal of herbivorous and invertebrate-feeding fishes may allow invertebrate populations (in particular, sea urchins) to increase, subsequently decreasing the populations of algae but increasing the rate of bioerosion of coral substrate. This in turn may result in a significant loss of species diversity, structural complexity, and overall loss of function and reductions in the fishable biomass (Jennings and Polunin 1996; Roberts 1995). The resulting dominant species of invertebrate grazers are vulnerable to removal (e.g. the species-specific pathogenic attack on *D. antillarum* in Jamaica and the wider Caribbean; see above), potentially leading to the dominance of filamentous and fleshy algae within coral reef communities. Thus, there may be three community equilibrium points possible, depending on the intensity of fishing pressures and the potential removal of invertebrate grazers: i) fish and coral dominance (no or very low fishing pressures); ii) urchin and turf algae dominance (intermediate fishing pressures); and, iii) filamentous and fleshy algae dominance (intensive fishing pressures; McClanahan 1995).

Much is not known concerning either the direct of indirect effects of fishing on coral reef communities. Simulation modelling approaches may aid in our understanding of the processes (e.g. McClanahan 1995). Further field research and experimental manipulations may allow us to refine models concerning the effects (Figure 4.3), or even dismiss the significance of fishing pressures on community structures and functions (e.g. Jennings and Polunin 1997). Indeed, as noted by Jennings and Polunin (1997), in certain instances both physical and biological processes operating on a broad range of scales may more significantly affect communities than fishing activities alone.

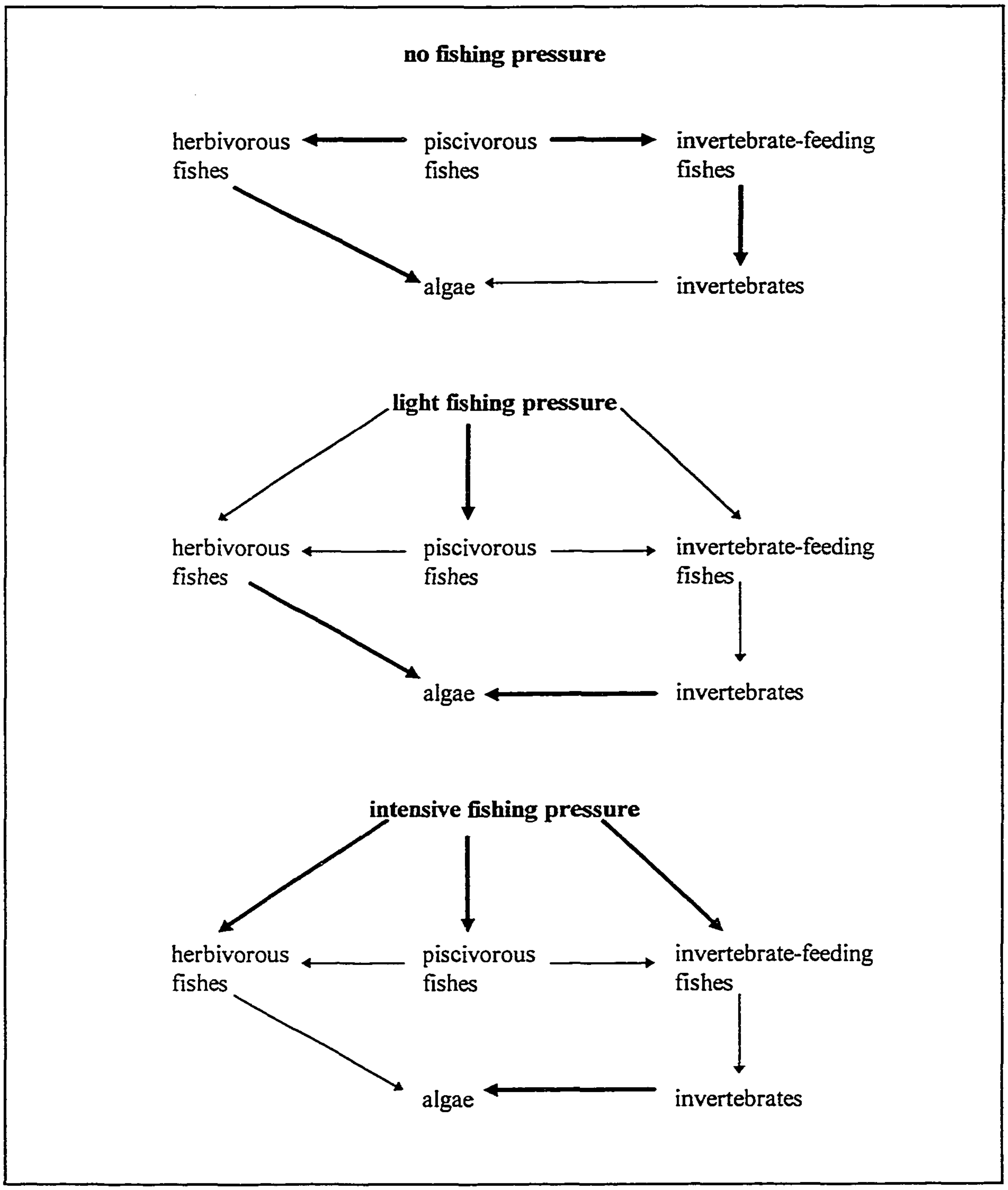

Figure 4.3. Theoretical shifts in the importance of tropical coral reef community trophic group relationships in response to fishing pressures. Darker arrows indicate a greater importance to the relationship (adapted from Jennings and Polunin 1996; see also Roberts 1995).

Future Directions for Index Development

In addition to not accounting for coral reef ecosystem dynamics, the methodology applied in this analysis did not make any explicit distinctions between the ways in which the functioning of ecosystems of different "types" may systematically vary. In other words, it would be useful to explore the application and behaviour of the ICEV as calculated for the extraction of a renewable natural resource from different ecosystems as represented by distinct food webs.

As an example of some of the principles which may come into play, Holling *et al.* (1995) noted that as it relates to the role of the biotic and abiotic actors, a distinction must be made between ecosystems in which the biota can exert some influence over the nature of the affecting physical processes (in effect, controlling the variability in the system over the shorter temporal scales), and ecosystems for which the biota are forced to adapt continuously to changing conditions. Many of the processes which promote resilient, functionally diverse ecosystems in the marine environment (especially pelagic systems) are likely physical in nature as the component species are generally unable to develop complex physical structures (Costanza *et al.* 1995). Coral reefs are the obvious exception in the marine environment, as they are able to develop complex structures and biologically regulated processes.

Costanza *et al.* (1993) distinguish between the dominance of the "red noise" characteristic of physical influences in a truly marine environment, and the dominance of the "white noise" characteristic in a terrestrial or terrestrially influenced (e.g. estuarine) environment. Adaptability of organisms in a marine environment can be quite pronounced as organisms encounter this relatively more predictable physical variation (red noise). However, although perhaps representing highly adaptive systems, many marine systems are unable to develop many complex environment influencing structures and the species can be highly mobile. The high mobility of many marine organisms, including migratory and transient behaviour, can make it difficult to characterise an ecosystem. By focusing on regions with a high number of endemic species with narrow and well defined spatial distributions, as is the case for a tropical coral reef, this issue is more amenable to control.

Perhaps one of the most pronounced shortcomings of the present analysis is the inability to take into account thresholds or criticality. Questions concerning the appropriate level of exploitation for sustainable use have dominated natural resource economics and biological studies of exploited populations; however, as argued previously (see Chapter 1) the question before us was not the level of exploitation possible or the fate of the exploited populations, but measuring the *contribution* to the value of the economic product. However, it may indeed be fruitful to explore the incorporation of bioeconomic principles directly into the construction and use of an index of ecosystem function. Such quantitative modelling was not possible here.

Meredith *et al.* (1994) discuss thresholds of environmental criticality as both an objective construct, as is the case concerning biophysically and economically defined standards, and a subjective construct, which may often be the case concerning socially defined standards. They emphasise the possibility of "multiple anthropocentric assessment perspectives" as determined by the particular set of personal values. The ICEV was more objectively defined based on ecological and economic principles. However, the selection and definition of the indicator necessarily involved a value judgement concerning what was to be measured (see below). Broadening the indicator to consider criticalities will increase the subjective content inherent in the indicator construction and use simply because the "dimensionality" of the indicator will be increased, requiring further value judgements to be made, often where there is a lack of scientific evidence to which one can appeal. This will not compromise the validity of the indicator *per se*, but will require prudent documentation and justification for the values selected.

It would also be potentially fruitful to explore the junctions between an ecosystem-based application of information theory, as done in this study, and economic information theory models. Inadequate information and risk as introduced into economic decision-making models is through consideration of either endogenously-modelled market uncertainties or exogenously-modelled event uncertainties (e.g. see reviews by Hirsheifer and Riley 1979, 1992). The economics of information, as relevant here, involves the process of individual economic agents attempting to gain information regarding exogenous event uncertainties in order to assist in the decision-making process. The ICEV represents a gain in information regarding the contribution of a natural ecosystem to productive value and, in essence, reduces uncertainty regarding the role of ecosystems in the economic production process (a role that is normally exogenous to neo-classical economic production models). As a more specific example of how information economics may potentially be applied, there is a significant body of work examining the effect of asymmetric information between principles and agents in the formation of contractual relationships (e.g. see Feltham *et al.* 1988; Macho-Stadler and Pérez-Castrillo 1997). As a specific issue of moral hazard (a gain in information by one party after the establishment of the "contract" between fishers and management authorities) or adverse selection (private information held by one of the contracting bodies), the ICEV or other aspects of ecosystem information theory may be able to be applied to economic asymmetric information models as represented by conflicting objectives through knowledge gained concerning the value of ecosystem function. It would be interesting to formally pursue this "information adding" angle as it relates to existing economic information theory and decision-making.

General Issues Concerning Food Web Statistics - the Case of Connectance

The Index of Captured Ecosystem Value (ICEV) requires information regarding the structure of the food web of interest, and as such is effectively a food web statistic. There are caveats and limitations associated with food web statistics, primarily stemming from the limitations of the food webs descriptions themselves. As stated by Lawton (1989), "In Pimm's words (1982), most published webs are therefore 'caricatures of nature'. My dictionary defines caricature as a representation exaggerated for comic effect, or a ludicrously inadequate or inaccurate imitation! This is a fair assessment of most of the published information on food webs."

In order to illustrate and discuss some of the relevant issues, the debate surrounding the calculation and use of one of the more prominently explored food web statistics - connectance - is briefly outlined. The calculation of the ICEV and food web connectance both rely on a description of the links between trophic species, and as such are inherently related (note also that connectance can also be formally related to indices of diversity; Margalef and Gutiérrez 1983). The critical debate concerning the calculation and use of information theory indices to describe network flows as it relates to many of the issues discussed here has yet to develop in the literature. Some preliminary comments are made with respect to the ICEV, but a thorough exploration of the issues is left for future work.

Food web *connectance* (C) effectively seeks to compare the functional diversity of webs with different numbers of species. It is traditionally defined as the total number of realised trophic interactions between species in a community as a proportion of the total number of possible interactions (Pimm *et al.* 1991; Warren 1994). In a common operational form

$$C = \frac{2L}{S(S-1)} \tag{4.22}$$

Alternatively, the expression is also written as

$$C = \frac{L}{\left[S(S-1)/2 \right]} \tag{4.23}$$

where L = the total number of realised predator-prey relationships between trophic species; and,

S = the total number of trophic species.

For each predator-prey interaction, equation 4.22 considers the dynamic effect of predator on prey *and* of prey on predator (hence the numerator as $2L$ not simply L; Pimm *et al.* 1991; Warren 1994). Equation 4.22 also excludes cannibalism within trophic species. In keeping with the theory of connectance, intraspecific interactions should be excluded from the analysis; however, interactions within trophic species groupings may be significant. Alternatively, *directed connectance* (DC) may be used as a measure of ecological complexity (Martinez 1991):

$$DC = \frac{L}{S^2} \tag{4.24}$$

Thus, directed connectance is the ratio of all realised interactions to all possible interactions, including cannibalism, but excluding the dynamic effect of the prey on the predator; alternatively, it can be thought of as "the average fraction of species which is consumed by any one trophic species" (Martinez 1996).

Caveats Associated with the Use of Food Webs and Food Web Statistics

There has been a significant amount of discussion in the literature concerning the validity of ecological research which has relied on the construction of generalised or aggregated food webs and the exploration of the properties of food web statistics such as connectance (e.g. Hall and Raffaelli 1993; Lawton 1989; Paine 1988; Peters 1988; Pimm *et al.* 1991). Certainly, many of the criticisms of the methodologies of past studies are valid, yet the most extreme criticisms which denounce the "nonoperational constructs" of research in ecology (Peters 1988), as illustrated by food web theoretical developments, are overly harsh. Ecology in general, including food web research, has advanced and benefited through often "subjective" interpretations as derived from relatively narrow study contexts and natural history-based descriptions. It is indeed dangerous to place too high a degree of confidence on statistics derived from highly subjective constructs, yet it is also equally dangerous to be naive of the nonoperational components inherent within all scientific studies, as all theoretical developments necessarily involve human interpretation and constructed models of reality of varying degrees. Particularly in cases which involve the marriage of economic and ecological theory as a potential contributing tool for social policy development (as I hope this study represents), a purist Popperian methodological standard for the ecological component may unnecessarily restrict the exploration of theoretical connections and of principles which may in turn guide future research.

That being said, following Peters' (1988) critique of the issues, it is necessary to be forthcoming concerning the caveats and limitation of the use of food web statistics associated with: i) operational simplifications of system complexities; ii) relativisms and subjectivity; iii) scale effects; and, vi) existing tautologies. This transparency assists in the presentation of the current research results with the necessary perspective. Although it may concern many physical scientists that the calculation and use of food web statistics necessarily also involves the use of qualitative components in the analyses, as well as having the specified goals of the research possibly affect the structure and tools of analysis, this is not a concern in the context for which the ICEV is developed

- as a transdisciplinary index which is useful beyond being simply a tool for biophysical scientific inquiry (see below).

Note that there is an inherent bias in any index based on an ultimately subjective set of resolved relationships between species. The level of resolution in the food web description, or the degree to which the data are subdivided for the selected depiction of the trophic relationships (i.e. the clumping or aggregation of trophic level species), may affect the index value (e.g. Paine 1988; Martinez 1991, 1993b) although there certainly is no clear answer on this matter (e.g. Sugihara *et al.* 1989). Results may thus in part reflect artefacts of the sampling procedure and methods of analysis. Perhaps the most extreme example of this potential problem is what Paine (1988) calls "artistic convenience" - the larger the ecological community, the more difficult it is for the researcher to detect and measure realised interactions, resulting in higher structural resolution of smaller communities. By not distinguishing between species or ontogenetic stages of species as separate actors within the food web, there is the risk that much of the locally or dynamically important interactions are missed (Paine 1988; see also comments regarding scale below). However, in conscious recognition of this tendency, researchers may also be apt to report every linkage, no matter how minor - rare, relatively unimportant interactions are thus overemphasized (Pimm *et al.* 1991). Many of these concerns have led to calls for the standardisation of food web descriptions, including the adoption of explicit study methodologies and more exhaustive descriptions (ideally the systematic recording of all species and all links at all trophic levels; Cohen *et al.* 1993).

As an additional issue, in resolving the interactions between trophic species one may define specific interaction coefficients or simply define the links as either present or absent as done in this study. Such a simple binary designation may seem overly simplistic, but by specifying the magnitude of trophic exchanges one encounters problems associated with temporal and spatial changes in the strengths of the specific relationships (Martinez 1994). In other words, declared specific measures of species interactions may lead to "misplaced concreteness" in the values as variability is ignored or implicitly assumed unimportant. Without a sufficiently high degree of resolution, however, there are potential difficulties associated with: i) ontogenetic shifts in the types of resources used and the consumption patterns; ii) the observation of transient species; iii) the residence time of species within the community under study; and, iv) the observability of species which, depending on how these factors are dealt with in the analysis, will affect the structure of the food web (Paine 1988).

There are no easy answers to this potentially significant issue. One must simply diligently document the source and method of collection of the primary data, all data manipulation procedures, and the specific clustering techniques employed. Variation in methodology as a source of error can be controlled by developing specific criteria that allow for consistent determination of relevant trophic species and the depiction of linkages. This will allow results to be potentially compared with other studies or revisited at a later date (e.g. future theoretical advancements or advancements regarding statistical techniques allowing for more objective treatments of the data to determine food web resolution). It is also advisable, as with the construction of any ecological model, to employ sensitivity analyses concerning the selected methodologies. This can pose a difficulty if one is relying on published sources. However, as noted by Pimm and Kitching (1988), biases can be controlled by relying on secondary data sources for which descriptions are taxonomically disaggregated and quantitative in nature.

It has been the history of much food web research to focus on certain species and particular relationships, with the result that food webs are often constructed with an emphasis on the taxonomic or functional group of interest to the detriment of resolution and specification of the relationships between other members of the community (May 1983; Paine 1988; Peters 1988). This can be attributed to a general lack of construction guidelines or established sampling methodologies. Thus, using secondary data of varying origins can result in statistical analyses which reflect artefacts of such differences. Such errors associated with food web construction may be further propagated as species or trophic species are defined in relation to each other - errors associated with one grouping may result in errors in other groupings (Peters 1988).

Again, there is no "quick fix" to these valid concerns. One can only appeal to the transparency of researchers' methodologies and hope that the development of more standardised community sampling techniques become a reality in the near future. It is difficult to remove personal biases in research, no matter what the field of inquiry. Indeed, researchers selecting certain aspects of a system on which to focus at the expense of other system components is common practice in ecology. Use of a community-wide statistic such as connectance under these circumstances, however, may make comparisons between systems or studies difficult.

The extent or scale of a food web is defined as the spatial, temporal, or statistical domain of the data (Martinez 1994). The research on which food webs are based may differ with respect to: i) the duration of the field research; ii) the geographic extent of the study site; iii) the range if organisms examined; and, iv) the frequency and methods of sampling (Goldwasser and Roughgarden 1997; Peters 1988).

Investigations into the structures of food webs have explored changes or consistencies in properties across a range of scales as reflected by the number of species or trophic species involved. For example, variations in the proportion of basal species (autotrophs and detritovores), intermediate species (species with both predator and prey relationships with other species), and top species (species without predators), variations in the proportional distribution of links between basal, intermediate, and top species, and variations in the connectance, the total number of links or the linkage density with changes in species richness (*S*) have been studied and reviewed (e.g. Cohen and Briand 1984; Hall and Raffaelli 1993; Havens 1992; Lawton 1989; Martinez 1991, 1992, 1993a, 1994; Martinez and Lawton 1995; May 1983; Pimm 1982; Pimm *et al.* 1991; Schoener 1989; Warren 1990, 1994).

There has been a significant amount of controversy and relatively recent developments with regard to the effect of scale on the characteristics of food webs. Criticisms of particular studies have been made with regard to the commensurability of data of different origins, the criteria used to identify significant linkages, the level of aggregation (resolution - see comments above) to distinguish between trophic species within food webs, and the selection of the statistic and methodology for analysis (e.g. Fonseca and John 1996; Havens 1992; Martinez 1994). It is currently evident that for certain food web properties (e.g. the proportion of basal, intermediate and top species in a food web) there may be little variance with changes in the species richness of a described web, while for others (e.g. connectance) there may or may not be significant variation. Questions concerning the range over which properties are scale variant or invariant further confound the issues (Martinez and Lawton 1995).

The food web connectance statistic has also been criticised because the range of all possible values is restricted through the nature of the measure itself (i.e. the number of possible interactions defines the minimum value the ratio can assume, and this value will vary by the number of compartments in the web). This constraint, inherent in the measure's design, has been labelled a tautology (Peters 1988). This is certainly a significant drawback associated with the measure of connectance and certainly violates a condition of a "good" indicator (see below). However, this is not a criticism that can be levied at the ICEV.

There is an additional possible tautology associated with use of any measure as an indicator of a specific food web property, such as ecosystem function. As discussed below, the development and use of an index necessarily involves defined purposes or research goals, and assertion of a value base. This in turn *should* affect the definition of the tools designed for or used in the analyses. Nevertheless, an indicator designed for a specific purpose should not be so narrowly defined as to simply measure the system state with respect to any specified goal. An indicator should capture a wider set of system characteristics to avoid such tautologies. There are certainly many questions remaining and room for future research concerning the full meaning of the system descriptions provided by the ICEV or other indices based in information theory (e.g. see Ulanowicz 1986).

Food web theory is not yet sufficiently developed to explain variations in food web properties among webs (e.g. Pimm 1982; Pimm *et al.* 1991). The specific patterns evident with changes in scale and resolution, the influence of habitat or environmental disturbance, or the role of community structure and type have yet to be examined to the degree necessary to provide meaningful results (Martinez 1992; Martinez and Lawton 1995; Schoener 1989; Warren 1994). At this stage, it is perhaps best to presume that there is a range of statistic values possible within bounds as determined by the properties of the index itself and influenced by biological principles (e.g. stability restrictions, morphology restrictions, or compartmentation of webs; Pimm 1982; Pimm *et al.* 1991; Warren 1990, 1994). This recognises not only that our current level of understanding of how statistics vary among webs is insufficient, but that ecosystems are dynamic structures in both space and time - this latter issue being often ignored in analyses (Pimm *et al.* 1991; Warren 1990, 1994).

A Note on Connectance Index Construction

Standardisation to control for sample size through the use of ratios, as is usually the case in a connectance index, assumes a specific relationship between the variables - they must have a linear relationship passing through the origin (Atchley and Anderson 1978; Atchley *et al.* 1976; Fonseca and John 1996; Packard and Boardman 1988). This defines an isometric relationship. Specifically for the calculation of connectance as in equations 4.22 or 4.24, this relationship must hold between the number of realised links (L) and the total number of possible links (S^2 or $S(S-1)$). This may be an indefensible assumption leading to spurious conclusions when comparisons are made between webs with significantly different species richness (S) as an allometric relationship between the variables can be expected in many instances (Fonseca and John 1996).

As noted above, a further caveat of the traditionally defined index of connectance is that the range of possible values varies with scale. As each species must interact with at least one other species to be defined as being part of that food web, the minimum connectance (C_{min}) is the reciprocal of the species richness (S^{-1}) and, by definition, falls hyperbolically with an increase in the number of species (Fonseca and John 1996). The change in C_{min} with a change in S can be significant and again makes the validity of comparisons of connectance between food webs differing in species richness (S) using either equations 4.22 or 4.24 highly questionable. There may similarly be a theoretical upper limit to connectance values below that associated with a maximally connected web as defined by either S^2 or $S(S-1)$.

To avoid the problems associated with the use of ratios, and to generally make the analysis more amenable to an expanded topology of information, Fonseca and John (1996) advocate a "community allometry" approach to exploring connectance which relies on the more appropriate direct use of linear regression methods and analysis of covariance (ANCOVA; for a general analysis of the statistical properties associated with use of ratios and issues with correlation and regression, see also Atchley and Anderson 1978; Atchley *et al.* 1976; Kuh and Meyer 1955; Madansky 1964). Through the use of a power function,

$$I_r = a(I_{\max})^b \tag{4.25}$$

where I_r = the total number of realised interactions;

I_{max} = the total maximum number of possible interactions;

a = a constant; and,

b = the power function coefficient, which measures the rate of change of I_r with I_{max}.

The regression will remove the explained dependency of variations in I_r due to variations in I_{max}; consequently, the residuals of the regression in effect become a scale independent index of connectance (Fonseca and John 1996). This methodology can only be applied, however, when a suite of food webs of varying scale are available for study.

In exploring possible indices which could be used as an index of ecosystem function for this study, an index directly derived from the connectance statistic was considered but subsequently discarded due to the additional caveats and limitations associated with the measure. The ICEV, as an information theory-based index, has fewer such design drawbacks. Further, it was found that use of a connectance-based index required more theoretical assumptions concerning the ways in which ecosystems operate, over a more simple accounting of the information content inherent in a resolved structure of flows. Nonetheless, it may be fruitful to explore alternate indices to the ICEV as proxy measures of captured ecosystem function value.

General Considerations for Index Development for Use in Decision-Making

Biodiversity indices have been developed with the primary purpose of being used as tools for scientific description and inquiry. However, use has been extended within the arenas of government and management to help fulfil the need for ecosystem information relevant for broader matters of ecosystem management, policy and regulation. What may often be underappreciated, however, and as illustrated through the above review of biodiversity indices (see Introduction), such use may not be appropriate given the theoretical underpinnings and design of the specific indicator or index employed.

Friedland (1977), in discussing the development of ecosystem models, makes the distinction between the use of models intended primarily for scientific inquiry and those intended primarily to inform decision-making and policy. Concerning the use of models (the development of indicators relies on, at a minimum, the use of a conceptual model) intended for use in decision-making, Friedland (1977) notes that "The basic objective is not the discovery of previously unknown truths but the collection and integration of existing knowledge and its presentation in a form useful in the policy-making process." This has direct ramifications for what type of indicator or index may be most appropriate.

An index must be developed with a specific purpose in mind. Such purpose is reflected in the range of indicator characteristics and properties, and the derived index implicitly restricts the set of appropriate applications. But generally what are the properties of a good indicator or index? What are the potential uses one should consider and how should these impact the indicator design? The relevant body of theory relating to the derivation of sustainable development indicators is briefly reviewed here to provide some insight on these matters. The sustainable development literature has distilled some guiding principles as they relate to indicators of multi-dimensional, complex systems designed to explicitly inform government and management agencies (multi-dimensional as specifically used in this context refers to inter-related facets of human distribution and entitlement potentially concerning environmental, economic, social, and institutional issues; e.g. see the framework of Gustavson *et al.* 1999).

One might begin with a relatively simple question - what is an indicator? Specific definitions and uses found in the literature vary widely. Broadly, an indicator can be thought of as "...something that points out, or stands for, something else..." (Gallopín 1997). They are proxy variables for attributes which themselves are difficult, if not impossible, to measure. In other words, indicators provide information with respect to a phenomenon of interest, yet do not *directly* measure that phenomenon. Indicators have a significance which extends beyond the properties directly associated with any one particular indicator variable (Braat 1991; Hammond *et al.* 1995). They are useful wherever obtaining the necessary information requires simplification or amalgamation of an overwhelming amount of data, or when collection of the necessary data to directly measure the phenomenon is not possible. Kuik and Verbruggen (1991) state that the primary function of indicators "...lies in simplification: indicators are a compromise between scientific accuracy and the demand for concise information."

Indicators and composite indicators, or indices (generally defined as a function of two or more indicators), have been employed for a variety of purposes:

- assessing existing states (providing a picture of baseline system conditions);
- revealing trends and oscillations (examination of data through time and across space allowing for spatial and temporal comparisons and for the identification of "hot spots");
- determining linkages (modelling of the indicators to provide insight into the behaviour of the system and the identification of those factors which determine specific behaviours);
- predicting future states and trends (using modelling to predict future temporal or spatial characteristics or changes in system conditions);
- improving decision-making and planning (making the existing conditions explicit to decision-makers, and helping to make the alternatives and results of options more apparent);
- assessing the efforts of management and the effectiveness of governance (monitoring the direct and indirect effects of intervention, as well as the progress towards or movement away from a desired state);
- contributing to social awareness and involvement (helping to facilitate social education, interaction, and public participation in decision-making and the establishment of policy); and,
- increasing accountability of decision-makers to the public (making the results of policies and programs more explicit).

Indicators can also be broadly categorised as either predictive, in that they provide some information concerning a future state, or retrospective, in that they provide a description of historical conditions (e.g. Braat 1991). Descriptions of historical conditions are ultimately limited in that they only serve to outline historical states or assess decision-making and policy efforts "after the fact". Such retrospective indicators can be used to guide future policy, yet doing so relies on an underlying base of assumptions or beliefs on how the system will behave based on the past. Predictive "trajectories" of indicator values will prove more useful - indeed, some consider the diagnostic and predictive function of indicators to be of up-most importance, making linkages and uncertainties associated with system behaviour explicit (e.g. Braat 1991; Ruitenbeek 1991; Winograd 1997). Such trajectories require use of a specified model of the system in question and of the relationships between its components, and can include simple extrapolation of values, correlation and regression modelling, and/or complex simulation modelling.

It has been argued in the literature on sustainable development indicators that the values and ethics behind the selection of a particular indicator set should be made explicit. In other words, because indicators consist of information selected for the purpose of achieving a specific goal with specific underlying theories and assumptions, the motivations involve a value judgement and the assertion of an ethical base. Ethics and values can come into play at two stages - during the selection of the indicators, and during the interpretation of the indicators (Ruitenbeek 1991). With regard to the selection of indicators, Gallopín (1997) notes that an internalisation of the values and ethics within the indicator results in decision-making based often on a quantitative comparison of indicator measures with no further interpretation or manipulation of the numbers required (e.g. economic cost-benefit analysis). If this is to be done, then the ethical and value base of the indicator must be made clear at the onset. Preferably, an indicator should be structured such that one can make explicit judgements from the information provided by the indicator without allowing the results themselves to necessarily dictate the desired policy or management direction; that is, values and ethics should come into play during the interpretation of the indicators.

Social values and ethics can be expected to vary greatly geographically. Furthermore, social values and ethics are not fixed in space or time - they are constantly changing and evolving. For these reasons, it could be further argued that it is important to allow for some "flexibility" in selected indicator sets to accommodate changing values and ethics, or the indicators themselves and the research programme in which they are based risk becoming quickly "obsolete". A rigid and "value laden" set of indicators which will not accommodate such changes may serve a necessary and valuable purpose in a particular place and time, but is not generally desirable for an indicator set which will be applied over a broad geographical area or be expected to have "staying power" over time. Further, as noted by Ruitenbeek (1991), indicators are often chosen based on the demand of the users, not on a developed *a priori* theoretical or ethical base, despite the best intentions of the researcher. Over-prescription of a strong ethical base which pretends to constrict or guide the selection process is often just not realistic or forthcoming to indicator use in practice.

Selecting a flexible indicator set by avoiding value laden indicators does not limit one to objective over subjective indicators, or quantitative over qualitative indicators. Qualitative indicators, or indicators based within human perceptions, may still fulfil the conditions of "good" indicators, and in fact such indicators are sorely lacking in indicator research. Any indicator is not inherently value laden - some are more closely linked to a specific ethical and value base than others. For example, some indicators implicitly define what is "good" or "bad" (i.e. they have an implied directionality, what Anderson 1991, p.48, calls indicators with "automatic evaluation"), while others simply define a variable as something that is important to measure. Some indicators are tautological in that they narrowly define what they measure and the measure itself in turn reflects a specific policy objective - such indicators should be avoided as they are highly restrictive and likely say little of the characteristics of the overall system. There is also the danger that the selected indicator set will be dependent on a relatively narrowly defined body of theory and scientific knowledge, and thus may become obsolete if the theoretical base is refuted. Also, indicators may only be applicable to a specific spatial scale of the system in question; with hierarchical systems, different indicators may not effectively characterise behaviours across spatial scales as cause-effect relationships which define the behaviour of the system will tend to change from the local to the global scale (Gallopín 1997; Winograd 1997).

In order to adequately address the plurality of issues which the particular indicator set is designed to address, and to organise the development and measurement of the indicators or index, it is important to select indicators based within an adequate conceptual framework. The number of indicators selected should be minimal yet sufficient to describe and characterise the system components of interest - in fact, it could be said that a measure in itself does not become an indicator until it is related to a larger phenomenon through a conceptual framework. There have been numerous attempts to develop indicator and index systems over the past decade to describe or predict human conditions within the social sciences, and numerous frameworks have been developed for indicator selection within the realm of sustainable development research (e.g. see Hodge 1997; Kuik and Verbruggen 1991; Molden and Billharz 1997).

From this brief review of indicator uses and characteristics, the following statements concerning the properties of a "good" indicator for use in policy development and decision-making regarding the environmental management of multi-dimensional, complex systems are evident:

- the index design corresponds to the selected purpose and application;
- the value base behind the indicator design is explicit;
- the index provides a sufficient simplification or abstraction of the targeted system characteristics;
- the theory behind the design of the indicator is relatively robust;
- the sensitivity of the index to system parameter changes has been sufficiently explored and defined, and the index is sufficiently sensitive to meet the design purpose; and,
- the information provided by the index can be understood and applied by the user.

Most indicators or indices can not be expected to meet all of the above criteria; however, it remains the goal of indicator development to satisfy as many as is possible given shortcomings of the level of available scientific knowledge, logistical restrictions, and the demanding needs of the users. It is asserted that the ICEV meets all of the criteria of a "good" indicator, with the one exception that the sensitivity and behaviour of the index to parameter changes remains to be explored further.

As an example of how one may apply some of the above criteria in the evaluation of indicator sets for use in decision-making and policy development, Ruitenbeek and Cartier (1998) reviewed economic indicators in use or being developed for the assessment of the sustainability of forestry practices. The evaluation was based primarily on: i) an indicator in question being able to say something of the goals of economic efficiency, equity, and sustainability; ii) relevance of the indicator to different points of intervention (stand management, institutional, and policy levels);

and, iii) the indicator being administratively applicable to the indicator developer at the national, regional, or local level. The classes of indicators which most notably were found to be insufficient or undesirable included indicators which were economic valuations of global functions or regional ecosystem functions (as these are often too complex to accurately or regularly apply at the stand level, or there are much simpler indicators available which can capture equivalent information), and complex economic distribution indices and coefficients (as accurate estimation at the stand level is not feasible and, again, there are much simpler indicators which better apply at the local level and may even provide more useful information). What is the overall lesson to be gleaned for the study of Ruitenbeek and Cartier (1998), as well as evident from other indicator work? - *keep it simple.*

This present study was confronted with severe data restrictions, given our overall limited knowledge of the functioning, structure, and dynamics of most of the world's ecosystems. This can be expected to be the norm for some time to come, yet management decisions and government policy is significantly affecting how and in what ways humans are currently interacting with the environment. Science needs to begin to move closer to operational means to evaluate impacts and analyse the trade-offs involved. The ICEV is presented as one possible tool.

Chapter 5: Summary

This dissertation focused on the extraction of ecosystem values as represented by the flow from a renewable natural resource to the human economy. Two key premises were accepted at the onset: 1) the production of goods and services by the human economy relies on the drawing of services provided by natural capital and thus on the productivity of natural biotic systems; and, 2) the drawing of services provided by natural capital effectively captures or embodies function value of the originating ecosystem. The artisanal fisheries of Jamaica was used as a case study in the examination of the characteristics of economic production processes and the development of a biophysically-based index to account for captured ecosystem values. The development of a meaningful model was the ultimate goal.

In the examination of the Jamaican coral reef fisheries, the application of a bioeconomic model was deemed inappropriate. The coral reef fisheries addressed in this study can be characterised as highly diverse mixed species fisheries. As soon as one considers the quantitative modelling of multi-species or mixed species fisheries, and their associated complex ecological relationships, the problem becomes extremely complex and generally unmanageable. Moreover, in pursuit of sufficient bioeconomic models many of the elementary fundamentals of microeconomic production function construction have been neglected. As demonstrated by this project, there is validity in a return to an analysis of the fishery production process as an exertion of fishing effort at the level of the firm.

Cobb-Douglas and translog models of fishing effort were derived from catch and effort data for the years 1996 and 1997 to describe the relationships between catch and firm-level inputs as they vary by fishery within Jamaica. Data on the total catch, crew size, gear soak time, and quantity of gear were used to explore separate functions of effort for the use of China net, trap, hand line, palanca, speargun, and troll fishing technologies. By further accounting for the month and fishing location (i.e. north coast shelf versus south coast shelf and banks), the seasonal and regional influences on catch rates were similarly explored. Although in some instances the catch and effort data yielded less then satisfactory model fits, some patterns emerged. As a notable example, in the extended translog models for trap fishing and speargun fishing, a decrease in catch with a given level of effort was associated with fishing the north coast shelf waters. Positive seasonal influences were also revealed in catches during the late summer or fall (August through November) for five out of six fisheries examined, although the specific months in which the peak influence occurs and the relative size of the influence varies by fishery.

Additional questions remain regarding the structure of fishing activities within Jamaica and the effect of various qualities and quantities which may be characteristic of fishing efforts. In all but a few of the production function models, a large amount of the variance in catch levels remains unexplained. It was not possible to model many factors which may have a significant influence due to the limitations of the available data (e.g. questions remain concerning the possible effect of unaccounted variations in vessel fishing power and fishing skills, and regional and temporal variations in the availability of fishes on a finer scale than that captured by the divisions employed here).

Jamaican fisheries are largely overexploited. With the exception of the industrial conch fishery, the management regime is open access with no effective common property management structures evident. Confronted with the problems associated with the poor economic conditions of the country, limited government resources, and the numerous technical complexities associated with multi-species fisheries management, the successful economic management of the coral reef fisheries of Jamaica is not a task for the timid. Specific fisheries management options that have been suggested for Jamaica include gear limitations and area closures, accompanied by resource enhancement, fisher education and publicity campaigns, and improved institutional co-ordination and involvement. All of these suggested measures, however, do very little to directly address either the economic conditions of the fishers or the greater institutional and economic factors contributing to the overexploitation of the marine resources. Understanding the economic incentives involved is particularly important. In addition to the descriptive information concerning the nature of fisheries production processes as provided here, information concerning the characteristics of resource rents and the distribution of economic costs and benefits is necessary. The development of an economic data collection and analysis programme for Jamaica (also more widely applicable to countries of the developing tropics), which will allow for more informed decisions concerning the management of coral reef fisheries, is outlined based on an expansion of the existing Jamaican Catch and Effort Data Collection Programme.

The provided socio-economic profile of the fishers utilising the waters of the Montego Bay Marine Park serves as an example of a detailed characterisation of artisanal fisheries in Jamaica. In addition, the direct local use values as reported illustrate a more traditional economic approach to resource valuation. The monetary value of the contribution of natural resources to the final value of the goods whose production utilises those resources is ultimately represented by the resource rent earned. It is through the consideration of the local use values, in conjunction with subsequent analyses regarding the ecological condition of the marine environment and the sustainable level of use, which will allow management authorities to obtain more complete information on the extent of the local economic use benefits at risk of being lost if conservation efforts prove inadequate. However, resource rents do not measure nor necessarily adequately reflect ecosystem *functional contributions* to the economic production process.

Allowing information needs to direct research efforts, a non-monetary index is presented which measures the ecosystem function value captured by the economic process as embodied in the fishes. Ecosystem function addresses the relationships between living entities and ecological processes involving materials, energy and information exchange. Food webs are simply depictions or models of how species in a community interact, in which interactions with processes are either as consumers or resources. This served as the basis for the derivation of an index. The Index of Captured Ecosystem Value (ICEV) is developed from a basis in information theory relevant to an analysis of network flows in ecosystems.

Technical coefficients, describing the production relationship between ICEV values and market values of catches associated with individual fishing efforts, revealed that captured ecosystem function associated with fisheries using distinct technologies (i.e. China net, trap, hand line, palanca, and speargun) were valued differently by the market. The creation of "surplus value" appears to be rooted in the observation that certain fisheries target species which are more connected within the coral reef food web than those species typically captured by other fisheries. Consideration of the biophysical contribution of coral reef ecosystems to fisheries production reveals distortions between market and supply-side values. As stated in the previous chapter, in effect we are not valuing the ecosystem contributions to the creation of the final economic products to the same extent within the fisheries sector of the economy.

The development and use of indicators is still very much an art. All indicators or indices have associated caveats and limitations, and none are perfectly suited to their full range of desired uses, or often to their uses in practice. This is most simply a direct result of what an indicator is intended to be - a simplified characterisation of a complex reality. One must be ever mindful of the theoretical underpinning and design of the specific indicator or index employed, and thus of the appropriate context for its use. Questions concerning the indicator should be explored. What values does it reflect? Is the conceptual framework sound? Is the index or indicator set minimal yet sufficient? Is the theory behind the design robust? Is it sufficiently sensitive to system changes to provide the necessary information?

Following from the discussion in Chapter 4, there are notable problems with the ICEV. For example, it is dependent on a resolved food web structure. In future work, it would be desirable to explore the behaviour of the ICEV as dependent on resolution and scale. In addition, the index does not account for dynamic changes in the interactions between trophic species, and thus for changes in the individual functional roles of species over time. However, I believe that the objectives as set out on the onset of this project have been achieved. The ICEV, as presented, is a tool to account for the biophysical contribution of a natural ecosystem to an economic production process as value embodied in the ecological commodity. By examining how and to what extent ecological commodities are connected within their ecosystem, a fairly simple method of obtaining a "snapshot" of the dependency of particular economic production processes on ecosystem processes is provided.

It is this latter point which leads us to what I see as a key additional contribution of this dissertation. The examination of the economic production processes and resource values associated with the Jamaican artisanal fisheries, along with the subsequent analysis of the corresponding captured ecosystem values, asks us to think more explicitly regarding the economy's dependency on the natural environment. As stated early in this investigation, we are not asking

"How much can we take out?" but "What is the supporting ecosystem's role?" This study has revealed that some Jamaican fisheries are more dependent than others on ecosystem function in the provision of the ecological commodity. These values are not proportionately reflected in the market. This revelation begs us to look deeper at the question of how the harvesting of different species may be more or less dependent on ecological processes and interactions with other members of their community, and how fishing as an economic activity might be better managed.

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Appendix A:

Taxa Included in the Diet Composition Matrix

Table A.1 list the non-fish taxa and Table A.2 lists the fish taxa included in the diet composition matrix as reported in Opitz (1996) and used in this study for the derivation of trophic species groupings. An additional category, not shown here, consisting of "unidentified fish" was employed by Opitz (1996) in construction of the diet composition matrix. Vernacular names are as reported in Fischer (1978), Opitz (1996), and Froese and Pauly (1997). Where there were inconsistencies between sources for taxonomic (i.e. family, genus, and species) or vernacular names as shown in Table A.2, Froese and Pauly (1997) took precedent.

Table A.1. The 42 non-fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

organic detritus	priapuloids	isopods
benthic algae	chitons	shrimps
spermatophytes	gastropods	spiny lobsters
symbiotic algae	bivalves	scyllarid lobsters
phytoplankton	scaphopods	hermit crabs
decomposers/ microfauna	squids	crabs
zooplankton	octopuses	hemichordates
sponges	polychaetes	asteroids
hydrozoans	echiuroids	ophiuroids
sea fans	pycnogonids	echinoids
sea anemones	barnacles	holothurians
stony corals	stomatopods	tunicates
bryozoans	amphipods	sea turtles
sipunculid worms	tanaids	sea birds

Table A.2. The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Acanthuridae	*Acanthurus bahianus*	ocean surgeon
	Acanthurus chirurgus	doctorfish
	Acanthurus coeruleus	blue tang surgeonfish
Antennariidae	*Antennarius multiocellatus*	longlure frogfish
	Antennarius striatus	striated frogfish
Apogonidae	*Phaeoptyx conklini*	freckled cardinalfish
	Apogon maculatus	flamefish
Atherinidae	*Hypoatherina harringtonensis*	reef silverside
	Atherinomorus stipes	hardhead silverside
Aulostomidae	*Aulostomus maculatus*	trumpetfish
Balistidae	*Balistes vetula*	queen triggerfish
	Canthidermis sufflamen	ocean triggerfish
	Melichthys niger	black triggerfish
Belonidae	*Platybelone argalus argalus*	keeltail needlefish
	Strongylura timucu	timucu needlefish
	Tylosurus acus acus	agujon needlefish
	Tylosurus crocodilus crocodilus	hound needlefish
Blenniidae	*Entomacrodus nigricans*	pearl blenny
	Ophioblennius atlanticus	redlip blenny
	Parablennius marmoreus	seaweed blenny
	Scartella cristata	molly blenny

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Bothidae	*Bothus lunatus*	peacock flounder
	Bothus ocellatus	eyed flounder
Carangidae	*Caranx bartholomaei*	yellow jack
	Caranx latus	horse-eye jack
	Caranx ruber	bar jack
	Decapterus punctatus	round scad
	Oligoplites saurus	leatherjack
	Selar crumenophthalmus	bigeye scad
	Seriola dumerili	greater amberjack
	Trachinotus falcatus	permit
	Trachinotus goodei	palameto
Carcharhinidae	*Carcharhinus acronotus*	blacknose shark
	Carcharhinus falciformis	silky shark
	Carcharhinus leucas	bull shark
	Carcharhinus limbatus	blacktip shark
	Carcharhinus longimanus	oceanic whitetip shark
	Carcharhinus perezi	Caribbean reef shark
	Galeocerdo cuvier	tiger shark
	Negaprion brevirostris	lemon shark
	Rhizoprionodon porosus	Caribbean sharpnose shark
Chaetodontidae	*Chaetodon aculeatus*	longsnout butterflyfish

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Chaetodontidae	*Chaetodon capistratus*	foureye butterflyfish
	Chaetodon sedentarius	reef butterflyfish
	Chaetodon striatus	banded butterflyfish
Cirrhitidae	*Amblycirrhitus pinos*	redspotted hawkfish
Labrisomidae	*Lambrisomus guppyi*	mimic blenny
	Lambrisomus nuchipinnis	hairy blenny
Clupeidae	*Harengula clupeola*	false herring
	Harengula humeralis	red-ear herring
	Jenkinsia lamprotaenia	dwarf round herring
	Opisthonema oglinum	Atlantic thread herring
Congridae	*Heteroconger halis*	brown garden eel
Dactylopteridae	*Dactylopterus volitans*	flying gurnard
Dasyatidae	*Dasyatis americana*	southern stingray
Diodontidae	*Chilomycterus antennatus*	web burrfish
	Diodon holocanthus	long-spine porcupinefish
	Diodon hystrix	spot-fin porcupinefish
Inermiidae	*Inermia vittata*	boga
Engraulidae	*Anchoa hepsetus*	broad-striped anchovy
	Anchoa lyolepis	shortfinger anchovy
Ephippidae	*Chaetodipterus faber*	Atlantic spadefish
Fistulariidae	*Fistularia tabacaria*	cornetfish

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Gerreidae	*Eucinostomus argenteus*	silver mojarra
	Gerres cinereus	yellowfin mojarra
Ginglymostomatidae	*Ginglymostoma cirratum*	nurse shark
Gobiidae	*Coryphopterus glaucofraenum*	bridled goby
	Gnatholepis thompsoni	goldspot goby
	Gobiosoma evelynae	sharknose goby
Grammitidae	*Gramma loreto*	royal gramma
	Gramma melacara	blackcap basslet
Serranidae	*Rypticus saponaceus*	greater sopafish
Haemulidae	*Anisotremus surinamensis*	black margate
	Anisotremus virginicus	porkfish
	Haemulon album	white margate
	Haemulon aurolineatum	tomtate grunt
	Haemulon carbonarium	caesar grunt
	Haemulon chrysargyreum	smallmouth grunt
	Haemulon flavolineatum	French grunt
	Haemulon macrostoma	Spanish grunt
	Haemulon parra	Sailor's grunt
	Haemulon plumieri	white grunt
	Haemulon sciurus	bluestriped grunt
Hemiramphidae	*Hemiramphus balao*	balao halfbeak

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Hemiramphidae	*Hemiramphus brasiliensis*	ballyhoo halfbeak
Holocentridae	*Holocentrus ascensions*	squirrelfish
	Holocentrus coruscus	reef squirrelfish
	Neoniphon marianus	longjaw squirrelfish
	Holocentrus rufus	longspine squirrelfish
	Sargocentron vexillarium	dusky squirrelfish
	Myripristis jacobus	blackbar soldierfish
Kyphosidae	*Kyphosus incisor*	yellow sea chub
	Kyphosus sectatrix	Bermuda sea chub
Labridae	*Bodianus rufus*	Spanish hogfish
	Clepticus parrae	creole wrasse
	Halichoeres bivittatus	slippery dick
	Halichoeres garnoti	yellowhead wrasse
	Halichoeres maculipinna	clown wrasse
	Halichoeres poeyi	blackear wrasse
	Halichoeres radiatus	puddingwife wrasse
	Xyrichtys novacula	pearly razorfish
	Xyrichtys splendens	green razorfish
	Lachnolaimus maximus	hogfish
	Thalassoma bifasciatum	bluehead
Lutjanidae	*Lutjanus analis*	mutton snapper

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Lutjanidae	*Lutjanus apodus*	schoolmaster snapper
	Lutjanus cyanopterus	cubera snapper
	Lutjanus griseus	grey snapper
	Lutjanus jocu	dog snapper
	Lutjanus mahagoni	mahogany snapper
	Lutjanus synagris	lane snapper
	Ocyurus chrysurus	yellowtail snapper
Malacanthidae	*Malacanthus plumieri*	sand tilefish
Megalopidae	*Tarpon atlanticus*	tarpon
Monacanthidae	*Aluterus schoepfii*	orange filefish
	Aluterus scriptus	scrawled filefish
	Cantherhines macrocerus	whitespotted filefish
	Cantherines pullus	orangespotted filefish
	Monacanthus ciliatus	fringed filefish
Mugilidae	*Mugil curema*	white mullet
Mullidae	*Mulloidichthys martinicus*	yellow goatfish
	Pseudupeneus maculatus	spotted goatfish
Muraenidae	*Echidna catenata*	chain moray
	Lycodontis moringa	spotted moray
	Gymnothorax vicinus	purplemouth moray
Myliobatidae	*Aetobatus narinari*	spotted eagle ray

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Ogcocephalidae	*Ogcocephalus nasutus*	shortnose batfish
Ophichthidae	*Myrichthys breviceps*	sharptail eel
	Myrichthys ocellatus	goldspotted eel
	Ophichthus ophis	spotted snake eel
Opisthognathidae	*Opisthognatus aurifrons*	yellowhead jawfish
	Opisthognatus maxillosus	mottled jawfish
	Opisthognatus whitehurstii	dusky jawfish
Ostraciidae	*Acanthostracion polygonius*	honeycomb cowfish
	Acanthostracion quadricornis	scrawled cowfish
	Lactophrys trigonus	buffalo trunkfish
	Lactophrys bicaudalis	spotted trunkfish
	Lactophrys triqueter	smooth trunkfish
Pempheridae	*Pempheris schomburgki*	glassy sweeper
Pomacanthidae	*Centropyge argi*	cherubfish
	Holacanthus ciliaris	queen angelfish
	Holacanthus tricolor	rock beauty
	Pomacanthus arcuatus	grey angelfish
	Pomacanthus paru	French angelfish
Pomacentridae	*Abudefduf saxatilis*	sergeant major
	Abudefduf taurus	night sergeant
	Chromis cyanea	blue chromis

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Pomacentridae	*Chromis multilineata*	brown chromis
	Microspathodon chrysurus	yellowtail damselfish
	Stegastes fuscus	Brazilian damsel
	Stegastes leucostictus	beaugregory
	Stegastes planifrons	threespot damselfish
	Stegastes variabilis	cocoa damselfish
Priacanthidae	*Priacanthus arenatus*	Atlantic bigeye
	Heteropriacanthus cruentatus	glasseye
Scaridae	*Scarus coelestinus*	midnight parrotfish
	Scarus iserti	striped parrotfish
	Scarus guacamaia	rainbow parrotfish
	Scarus taeniopterus	princess parrotfish
	Scarus vetula	queen parrotfish
	Sparisoma aurofrenatum	redband parrotfish
	Sparisoma chrysopterum	redtail parrotfish
	Sparisoma radians	bucktooth parrotfish
	Sparisoma rubripinne	redfin parrotfish
	Sparisoma viride	spotlight parrotfish
Sciaenidae	*Equetus lanceolatus*	jack-knifefish
	Equetus punctatus	spotted drum
	Odontoscion dentex	reef croaker

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Sciaenidae	*Pareques acuminatus*	high-hat
Scombridae	*Euthynnus alletteratus*	little tunny
	Scomberomorus cavalla	king mackerel
	Scomberomorus regalis	cero
Scorpaenidae	*Scorpaena brasiliensis*	barbfish
	Scorpaena grandicornis	plumed scorpionfish
	Scorpaena inermis	mushroom scorpionfish
	Scorpaena plumieri	spotted scorpionfish
	Scorpaenodes caribbaeus	reef scorpionfish
Serranidae	*Alphestes afer*	mutton hamlet
	Cephalopholis cruentata	graysby
	Cephalopholis fulva	coney seabass
	Epinephelus adscensionis	rock hind
	Epinephelus guttatus	red hind
	Epinephelus itajara	esonue grouper
	Epinephelus striatus	Nassau grouper
	Hypoplectrus aberrans	butter hamlet
	Hypoplectrus chlorurus	yellowtail hamlet
	Hypoplectrus nigricans	black hamlet
	Hypoplectrus puella	barred hamlet
	Mycteroperca bonaci	black grouper

Table A.2 (continued). The 209 fish taxonomic groupings included in the diet composition matrix from Opitz (1996).

family	species	vernacular name
Serranidae	*Mycteroperca tigris*	tiger grouper
	Mycteroperca venenosa	yellowfin grouper
	Paranthias furcifer	creole fish
	Serranus tigrinus	harlequin bass
Sparidae	*Archosargus rhomboidalis*	western Atlantic seabream
	Calamus bajonado	jolthead porgy
	Calamus calamus	saucereye porgy
	Calamus pennatula	pluma porgy
	Diplodus argentus caudimacula	silver porgy
Sphyraenidae	*Sphyraena barracuda*	great barracuda
	Sphyraena picudilla	southern sennet
Sphyrnidae	*Sphyrna lewini*	scalloped hammerhead
	Sphyrna tiburo	bonnethead
Synodontidae	*Synodus foetens*	inshore lizardfish
	Synodus intermedius	sand diver
	Synodus synodus	diamond lizardfish
Tetraodontidae	*Canthigaster rostrata*	sharpnose puffer
	Sphoeroides spengleri	bandtail puffer
Triakidae	*Mustelus canis*	dusky smooth-hound