# Ecological-Economic modelling and implications of land use change and wetlands extent on freshwater fisheries: The case of Lake Victoria (East Africa)

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## Abstract

Watershed degradation and the conversion of wetlands to alternative uses affect water quality, having profound implications for the freshwater ecosystem and fisheries of Lake Victoria. Through dynamic simulations using the Ecopath approach we show that eutrophication may explain the sudden upsurge of the introduced Nile perch during the 1980s. During these years an important trigger of change was an abrupt shift in primary productivity due to an external shock, probably related to an El Nino-ENSO event. The Nile perch population explosion during the 1980s caused a profound transformation of the fishing industry leading to a dramatic increase in fishing effort. Our equilibrium analysis from the base values of an Ecopath model for year 1989 depicts an overfishing situation. To quantify the combined effect of eutrophication and fishing effort on the aggregate fish stock and fishery we specify and estimate a biomass dynamics model, which embeds a measure of phytoplankton biomass as an environmental variable. We found that in a eutrophic state, which for the Kenyan waters of Lake Victoria was above an estimated chlorophyll-a concentration threshold of 17.11 mg $m^{-3}$, a further increase in phytoplankton biomass would cause a decrease in aggregate fish stock biomass and landings respectively. We then model the relation between land use change and water quality, considering the nutrient retention function of wetlands. First, we identify catchment-based indicators, in particular price indicators and wetland extent, which may anticipate changes in environmental variables driving stock assessment models. We then study both nutrient loading and wetland reclamation externalities to the fishery sector, focusing on the Yala swamp, one of the largest wetlands in the Kenyan basin of Lake Victoria. Finally we propose a spatially distributed land use tax as policy instrument for reaching the desired objective of changing farmers' behaviour and maximising the total economic benefit to society.

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## Author's Declaration

I declare that the thesis comprises only my original work except where indicated. I certify that due acknowledgement has been made in the text to all other material used. Part of Chapter 6 represents a joint publication (*ICES Journal of Marine Science, 62(3):* 483-492) with Prof. Charles Perrings under the title: "Indirect economic indicators in bioeconomic fishery models: agricultural price indicators and fish stocks in Lake Victoria".

# Chapter 1

# INTRODUCTION

## 1.1. The problem of lake eutrophication

Freshwater lakes represent only a small part of the total water on the planet. Inland waters cover less than 2% of the earth surface, which is about $2.5*10^6$ $km^2$. In terms of volume, freshwater accounts for even less relative to the total water in the hydrosphere. Lakes represent only the 0.009% of the volume of the total amount of water, while rivers and groundwater, other freshwater components, account for the 0.00009% and 0.3% respectively (Kalinin & Bykov, 1969; Luovitch, 1977).

Despite their relative small amount of water, freshwater lakes are essential to the maintenance of terrestrial life and in supporting human consumption and activities. Lakes and reservoirs are used in several ways: drinking water, industrial use, agricultural irrigation, aquaculture, transportation, tourism and recreation, fishery, wastewater and nutrients sinks. The efficiency of lakes in providing these services is strongly related to their water quality. The fact that human effects on lakes are growing worldwide, increases the concern that lake services are jeopardized by lake degradation (Naiman *et al.*, 1995).

Eutrophication is probably the best understood type of lake degradation (Hasler, 1947; Edmondson, 1969; Harper, 1992). The term eutrophication describes the condition of a water body in terms of its trophic state, classifying lakes in accordance with their primary productivity (Jørgensen, 1980). An eutrophic state is related to an excessive growth of aquatic plants (i.e. phytoplankton) to levels that are considered to interfere with desirable water uses (Scheren, 1995). However, as the amount of phytoplankton is linked to other structural components, the classification of trophic state may often be related to other variables or trophic state criteria. Water quality variables can be grouped generally into hydrological, morphological, physical, chemical, and biological types. This multidimensional trophic concept has often been based on a single criterion, such as nutrient loading, nutrient concentration, productivity, faunal and flora quantity and quality, etc. However, there is currently no consensus as to what single parameter can measure trophic status or what values define changes from one trophic level to the next (Table 1.1).

Trophic state classification defines a lake according to its degree of eutrophication, or lack thereof, relative to the oligotrophic state. In general, three main trophic levels can be identified: oligotrophic (i.e. low phytoplankton concentrations); mesotrophic (medium phytoplankton concentrations); and eutrophic (high phytoplankton concentrations). However, other intermediate levels, such as ultra-ologotrophic, oligo-mesotrophic, meso-eutrophic, hypereutrophic, and dystrophic can also be adopted (Likens, 1975; Jørgensen, 1980).

The state of eutrophic lakes can severely affect the provision of many of their services to society. Because of the reduced transparency of the water, for instance, eutrophication decreases the lake potential for tourism and recreational activities. The growing biomass of water weeds and floating macrophytes supported by high level of nutrients can reduce the working efficiency of hydroelectric turbines in reservoirs or create a severe obstacle to lake's navigation. The excess of nutrients prevents the use of water for human consumption or may require costly depuration processes. Water quality is also important for aquaculture and industrial use, both of which are affected by eutrophication.

Nevertheless, the most important consequence of eutrophication is probably its effect on fishery. A moderate increase in nutrient availability and phytoplankton productivity in lakes is not necessarily detrimental. Indeed, it can actually increase fish production (Stockner & Shortreed, 1988). However, high phytoplankton biomass and productivity, as in eutrophic lakes, have a detrimental effect on fish growth and reproduction rates through the depletion of oxygen concentration in superficial waters. Moreover, eutrophication changes phytoplankton species composition, along with other factors such as temperature (Tilman *et al.*, 1986; Varis *et al.*, 1989), light (Zevenboom, 1982) and pH (Shapiro, 1973).

Phytoplankton species composition influences phytoplankton productivity (Bootsma & Hecky, 1993). If phytoplankton can readily be utilized by zooplankton and fish, then nutrients and biomass will accumulate at higher trophic levels that are useful for human consumption (Hecky, 1984). However, if phytoplankton biomass cannot be utilized by higher trophic levels, phytoplankton biomass will increase (Bootsma & Hecky, 1993). This is the case of ecosystems in which phytoplankton species composition shifts from green algae and diatoms, typical of oligotrophic lakes, to a state dominated by blue-green (cyanobacteria) algae in eutrophic conditions. Such a shift in phytoplankton species composition might be expected to result in a lower efficiency of energy transfer to higher trophic levels, since cyanobacteria, which are difficult to digest for many fishes and zooplankton, are a poor food source (Lampert, 1981; Heerkloss *et al.*, 1984; Haney, 1987).

**Table 1.1:** Trophic state classification criteria.

| Variable | *Ultra-oligotrophic* | *Oligotrophic* | *Oligo-mesotrophic* | *Mesotrophic* | *Meso-eutrophic* | *Eutrophic* | *Hypertrophic* | *Dystrophic* | Source |
|---|---|---|---|---|---|---|---|---|---|
| **Phytoplankton biomass** (mg C m$^{-3}$) | <50 | 20-100 | | 100-300 | | >300 | | <50-200 | Jørgensen (1980); Likens (1975) |
| **Phytoplankton density** (cm$^{3}$ m$^{-3}$) | <1 | | 1-3 | | 3-5 | | >10 | | Jørgensen (1980); Likens (1975) |
| **Chla** ($\mu$g l$^{-1}$) | 0.01-0.5 | 0.3-3 | | 2-15 | | 10-500 | | 0.1-10 | Jørgensen (1980); Likens (1975) |
| | | < 1 | | | | > 6 | | | Carlson (1977) |
| | | < 2.8 | | 3.5-9 | | > 8.7 | | | Chapra & Tarapchak (1976) |
| | | < 3.5 | | | | 9-25 | > 25 | | Nürnberg (1996) |
| | | < 4 | | 4-10 | | > 10 | | | Thomann & Mueller (1987) |
| | | < 4.3 | | | | > 8.8 | | | Dobson *et al.* (1974) |
| **TP** ($\mu$g l$^{-1}$) | < 1-5 | | 5-10 | | 10-30 | | 30->5000 | <1-10 | Jørgensen (1980); Likens (1975) |
| | | < 10 | | 10-20 | | > 20 | | | Thomann & Mueller (1987) |
| | | < 10 | | 10-30 | | 30-100 | > 100 | | Nürnberg (1996) |
| | | < 14 | | | | > 25 | | | Porcella *et al.* (1980) |
| **TN** ($\mu$g l$^{-1}$) | <1-250 | | 250-600 | | 500-1100 | | 500->15000 | <1-500 | Jørgensen (1980); Likens (1975) |
| | | < 140 | | | | > 180 | | | Porcella *et al.* (1980) |
| | | < 350 | | 350-650 | | 650-1200 | > 1200 | | Nürnberg (1996) |
| **SD** (m) | | > 3.6 | | | | < 1.9 | | | Carlson (1977) |
| | | > 4 | | 2-4 | | 1-2 | < 1 | | Nürnberg (1996) |
| | | > 4 | | 2-4 | | < 2 | | | Thomann & Mueller (1987) |
| | | > 6 | | | | < 3 | | | Dobson *et al.* (1974) |
| **Net DO** (mg l$^{-1}$) | | < 4.5 | | | | > 5 | | | Porcella *et al.* (1980) |
| **Min DO** (mg l$^{-1}$) | | > 7.2 | | | | < 6.2 | | | Porcella *et al.* (1980) |
| **ODR** (mg m$^{-2}$ day$^{-1}$) | | < 250 | | | | > 550 | | | Mortimer (1941) |
| **PP in growing season** (mg C m$^{-2}$ day$^{-1}$) | <50 | 50-300 | | 250-1000 | | >1000 | | <50-500 | Jørgensen (1980); Likens (1975) |
| | | 30-100 | | | | 300-3000 | | | Rodhe (1969) |
| **Total Annual PP** (g C m$^{-2}$ year$^{-1}$) | | 7-25 | | | | 75-700 | | | Rodhe (1969) |
| **Total organic carbon** (mg l$^{-1}$) | | <1-3 | | <1-5 | | 5-30 | | 3-30 | Jørgensen (1980); Likens (1975) |
| **Total inorganic solids** (mg l$^{-1}$) | 2-15 | | 10-200 | | 100-500 | | 400-60000 | 5-200 | Jørgensen (1980); Likens (1975) |
| **Light extinction coefficient** ($\eta$ m$^{-1}$) | 0.03-0.8 | 0.05-1 | | 0.1-2 | | 0.5-4 | | 1-4 | Jørgensen (1980); Likens (1975) |

ODR= oxygen deficit rate; DO= dissolved oxygen; PP= primary productivity; SD= Secchi depth; TN= total nitrogen; TP= total phosphorus; Chla= chlorophyll-a

In addition, some cyanobacteria are responsible for the production of toxic substances that cause massive fish kills during algal blooms.

Higher phytoplankton biomass, accompanied by changes in species composition towards cyanobacteria, leads to higher unassimilated biomass. This results in more dead biomass of phytoplankton sinking to the sediment bed. The potentially detrimental effects of increased sedimentation include a rise of the anoxic boundary depth, due to increased oxygen demand from biomass decomposition and respiration by bottom-living bacteria (Cornett & Rigler, 1979), and more rapid light attenuation which reduces the depth of the euphotic zone (Bootsma & Hecky, 1993). Moreover, deoxygenated conditions in bottom waters increase the rate of phosphorus recycling from sediments, thereby further increasing primary production (Caraco, 1993).

Deoxygenated conditions in eutrophic lakes may also result in shrinkage of available fish habitat (Bootsma & Hecky, 1993). This problem is particularly important to fisheries that target high-priced fish – usually a top predator. These species usually require high oxygen concentration: a minimum 60-70% saturation (Jørgensen, 1980). As consequence, top predator fishes are rarely present in hypereutrophic lakes and lakes polluted with organic matter. Under eutrophic conditions high oxygen demanding fishes may not be the only species affected. An excess of nitrogen concentrations can increase the presence of ammonia ($NH_3$) within the lake. Ammonia, which is the product of a series of chemical processes from an inorganic form of nitrogen such as ammonium, is considered to be toxic to all fish. A concentration of 0.025 mg $l^{-1}$ of un-ionized ammonia affects fish growth, although the lethal concentration is considerably higher, 0.5 mg $l^{-1}$ (Jørgensen, 1980).

Eutrophication often results in the simplification of aquatic ecosystems. Species diversity decreases (Rapport, 1989; Schindler, 1990) and food chains become less complex. This ultimately reduces the ecosystem resiliency to external shocks and overfishing. The effect of decreasing stocks of phytoplanktivorous fishes due to phytoplanktivores-targeted overfishing or the introduction of new predator species, can lead to phytoplankton biomass accumulation even without the presence of cyanobacteria. This is the typical case of top-down cascading effect that propagates through the food web in aquatic ecosystems.

## 1.2. Understanding lakes in their watershed context

Scientific studies on lake ecosystems processes have increased in spatial and disciplinary scope during the past century (Carpenter & Cottingham, 1997). Although many insights have been derived from the view of lakes as bounded systems defined by the land-water interface, limnologists recognize that lakes must be understood in the landscape context of their catchments (Likens, 1984; Wetzel, 1990). This is the field of watershed management, whose aim is the protection of water resources through an understanding of the interactions between land, water, and people for the production of goods and services (Dixon, 1997).

Watersheds have been studied from various perspectives. Traditionally the field of watershed management had a strong hydrologic focus without considering economic factors. Economic factors were the foci in water-resource development studies undertaken by economists to analyse issues concerning economic efficiency of water use and investments. Among the earliest research in this field were economic studies by Eckstein (1958), Krutilla & Eckstein (1958), Hirshleifer (1965), Maas *et al.* (1962), and Dorfman (1965, 1972). However, many of these studies were particularly concerned with capital investment decisions, mostly ignoring socio-economic issues of watersheds and how inhabitants used land and water resources to produce income (Dixon, 1997). Taking such a limited single-issue approach to the catchment frequently resulted in resource developments that compromise people, the environment and wider resource-use options (Goldsmith & Hillyard, 1986; Adams, 1992). The modern concept of watershed management has grown out of the long tradition of both physical and economic analysis of water resources. Watersheds are approached as complex biophysical and socio-economic units that require management which combines both economic and hydrologic factors (Dixon, 1997).

This concept includes the study of lake ecosystems and dynamics. It is now widely recognized that the many services lakes provide to humans are impaired not only by exploitation of lakes but also by changes in land use, vegetative cover, and other actions that affect the hydrology of the watershed. Ecosystem ecology, which offers a rich understanding of the physical, chemical, and biotic processes of lakes, requires also an understanding of how changes in agriculture, riparian land use, forestry, fossil fuel consumption, and demand for ecosystem services link lakes to larger social and economic systems (Postel & Carpenter, 1997).

In the field of environmental economics there are some studies that aim to integrate people and lakes in considering the processes that control normal and degraded lakes at a catchment scale. These studies provide a conceptual framework to model the interaction between human activities and eutrophication of lakes by using simple integrated models. Examples of this kind are provided by Carpenter & Cottingham (1997), Carpenter *et al.* (1999a), Janssen & Carpenter (1999), Mäler *et al.* (2000), and Janssen (2001). They target in particular the case of shallow lakes combining dynamic models of an ecosystem and a rule-based model of human behaviour following a multi-agent approach. However, they do not attempt any valuation of the externalities produced by decreasing water quality due to eutrophication.

When looking, in particular, at the externalities that eutrophication may generate to fishing activities, there are no integrated models able to link lake fisheries with land use change in the lake's watershed, and suggesting at the same time both an optimal use of land and an optimal deploition of fishing effort. Nevertheless, around the world many great lakes or inland seas, such as Caspian Sea, Lake Victoria, Lake Baikal, and the Laurentinian Great Lakes among others, are facing a dramatic situation of declining fish production due to environmental degradation. In many of these cases, agricultural and fishery policies are jointly responsible for the current situation. Our aim is to explore the theoretical issues involved in the case of Lake Victoria (East Africa).

## 1.3. Defining the area and the context of the study

Lake Victoria is the largest tropical lake in the world. The three surrounding countries Tanzania, Uganda and Kenya control 49%, 45% and 6% of the lake respectively, using its resources for fishery, freshwater and transportation. The lake lies at 1,134 meters above sea level. It has a surface of 68,800 km$^2$ and a land drainage area of slightly over 181,000 km$^2$ across five countries (Table 1.2). Its shape is roughly square with greatest length and width being about 412 km from north to south, between latitudes 0°30'N and 3°12'S, and 355 km from west to east, between longitudes 31°37'E and 34°53'E respectively. Much of the lake is less than 40 m deep with a maximum depth recorded of

84 m and a volume of water estimated at about 2,760 km$^3$. The bottom is mainly covered by a thick layer of organic mud, but with patches of hard substrate, sand, shingle or rock. The coastline is indented with many bays and gulfs and is close to 3,500 km in length.

**Table 1.2:** Lake Victoria geographical characteristics.

| **Country** | **Lake surface area** | | **Shoreline** | | **Tributary basin** | |
|---|---|---|---|---|---|---|
| | km$^2$ | % | km | % | km$^2$ | % |
| Kenya | 4,113 | 6 | 550 | 17 | 38,913 | 21.5 |
| Tanzania | 33,756 | 49 | 1,159 | 33 | 79,570 | 44 |
| Uganda | 31,001 | 45 | 1,750 | 50 | 28,857 | 14.9 |
| Rwanda | 0 | 0 | 0 | 0 | 20,550 | 11.4 |
| Burundi | 0 | 0 | 0 | 0 | 13,060 | 7.2 |
| *TOTAL* | *68,870* | *100* | *3,450* | *100* | *180,950* | *100* |

***Source***: Bullock *et al.* (1995).

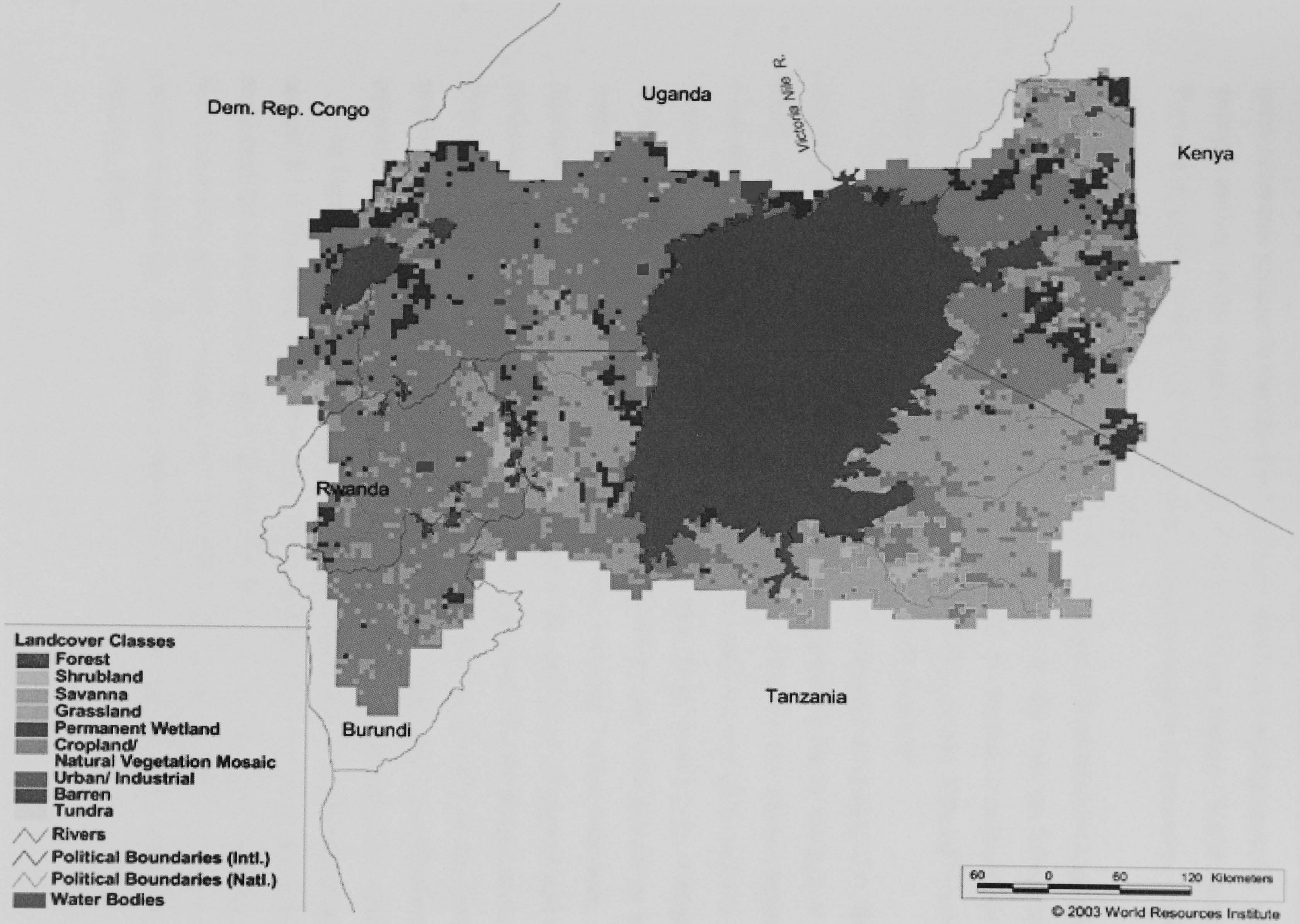
Lake Victoria became the focus of the attention in the 19th century, when it was identified as one of the sources of the Nile. Because it has a control structure for a large hydropower plant at its outlet at Owen Falls, the lake is classified as a very large reservoir. Hydrologically it behaves like a closed system with a water balance dominated by evaporation and precipitation (Table 1.3). The Kagera and Nzoia Rivers are the principal tributaries. Historically, an above average rainfall in the early 1960s resulted in a sudden rise in water level up to 2 m and extensive shore flooding. Since then, lake levels have receded to historic averages. Limnologically, Lake Victoria is monomictic and primary productivity is high.

**Table 1.3:** Lake Victoria annual water balance 1965-1990.

| **Water balance component** | **Volume (km$^3$)** | **Lake level equivalent (m)** |
|---|---|---|
| Lake rainfall | 125 | 1.81 |
| Lake evaporation | 110 | -1.6 |
| Lake net rainfall | 14.5 | 0.21 |
| Catchment rainfall | 293 | – |
| Catchment evaporation | 270 | – |
| Inflow from catchment | 23 | 0.34 |
| Net basin supply | 37.5 | 0.55 |
| Outflow (Owen Falls) | 38 | -0.57 |

***Source***: Bullock *et al.* (1995).

**Figure 1.1:** Lake Victoria and its watershed.



*Source*: IUCN Water Resources eAtlas (http://www.iucn.org/themes/wani/eatlas/html/af16.html).

Lake Victoria and its catchment support around 30 million people, as about one third of the population of Kenya, Tanzania and Uganda (estimated to be 90 million)[1]. The gross economic product in the lake basin is about US$ 3 to 4 million annually, yielding an average income in the range of US$ 90 to US$ 270 per annum (World Bank, 1996a). Population density in the basin is above the national average in all countries, with growing rates that are among the highest in the world.

Lake Victoria fisheries are a very important source of livelihood for local population. About 3 million people earn their living directly or indirectly from the fishing industry of Lake Victoria in the three countries[2]. Lake wide fish production is estimated at between 400,000-500,000 metric tons, with Tanzania landing 40%, Kenya 35% and Uganda 25%. Total value of the catch is between US$ 300-400 million annually[3].

In Kenya, in terms of contribution to the gross domestic product (GDP), the fishing industry is small but growing. While the industry accounted for an average of 0.2% of GDP during 1971-1981, the contribution rose to 4.4% in the 1990s (Ikiara, 1999). Considering that over 90% of the country total fish catch is obtained from Lake Victoria, Kenya's fishing industry strongly depends on the lake. In terms of source of employment, it has recently been estimated that a total of 798,000 people were directly or indirectly supported by the fishing industry. These figures included 34,000 fishermen, 238,000 dependants and 526,000 other people engaged in the provision of support and ancillary services (Bokea & Ikiara, 2000). The employment of these 560,000 people accounted for 25% of the country's employment in the informal sector and 14.5% of the country total employment (Bokea & Ikiara, 2000). In a country where the unemployment rate is estimated at between 25-30%, fishing is therefore a very important economic sector.

In Uganda, fisheries contribution to GDP has increased over the last decade, being around 2.1-2.2% during 1995-1998 period. Export earnings from fishing and fish products accounted for 4.9% of total exports in 1997 (NEMA, 1999). The value of fish exports has been increasing steadily between 1993 and 1996 from about US$ 8.9 million to US$ 45 million respectively. The fishing industry employs between 0.5 and 1 million Ugandans (NEMA, 1999).

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[1] Lake Victoria Environmental Management Project (http://www.lvemp.org).

[2] As for note 1.

[3] As for note 1.

Over 70% of the population of the three riparian countries is engaged in agricultural production, mostly as small-scale farmers cultivating sugar, tea, coffee, maize, cotton, or being engaged in livestock keeping or horticulture within the lake catchment[4].

In Kenya, more than 70% of the population lives in rural areas and depend on agriculture, whereas only 8% of country's area is cropland (World Bank, 1998). Kenya's population has increased rapidly: from 11 million in 1970 to 27 million in 1996, corresponding to an annual growth rate of around 3.5%. In contrast, agricultural annual production growth rate has decreased from 4.8% in the 1970s, through 3.3% during 1980-90, to a negative 1.5% in 1990-94. Consequently, food imports as percentage of total imports have increased from 8% in 1980 to 14% in 1993 (World Bank, 1996b). Moreover, arable land area per capita has decreased by 35%, from 0.23 ha in 1980 to 0.15 ha in 1995, and the population is projected to be 36 million by 2010 (World Bank, 1998).

In Uganda, agriculture is by far the most important sector in the national economy, producing domestic food requirements and generating raw materials for agro-based industries and commodities for export. It employs about 90% of the 20 million Uganda's population, contributing to 49% of GDP and 50% of total exports (Tugume 2001; IFPRI, 2002). About 56% of agricultural GDP is of subsistence nature (Tugume, 2001).

The importance of Lake Victoria has to be seen in the context of the widespread food security and its geographical location. In the mid-1990s it was estimated that 33% of the population in Africa, comprising of some 138 million people, was food insecure (IFPRI, 1995; USAID, 1994). The Lake Victoria basin, laying in the middle of a region which often has to face dramatic shortages of food resources and high rates of unemployment, is a crucial asset not only for the countries which share its water body, but also for their neighbouring countries. The land in the basin is one of the most fertile of the continent and the production of fish from Lake Victoria has represented, in the last decade, about 25% of the annual total catch from Africa's inland fisheries (FAO, 1995).

Recent trends, however, show that the Lake Victoria region is characterised by entrenched poverty and environmental degradation, arising from land degradation, soil erosion, desertification, biodiversity loss and overexploitation of natural resources (i.e. fisheries) among other factors. These problems have partly compounded difficulties in producing sufficient food, trapping people in a vicious downward cycle of food insecurity (Abila, 2000). In Kenya, for instance, average daily per capita calorie and protein supply

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[4] As for note 1.

has decreased by 7% and 10% respectively between 1981 and 1993 (World Resources Institute, 1998). In Uganda, taking into account that fish contributes over 50% of the animal protein supply, with the current high population growth, an annual fish catch close to the maximum sustainable yield, and an export commitment of around 65,000 tonnes per year, there is likely to be a considerable deficit in local fish supply (NEMA, 1999). Food insecurity in the Lake Victoria region is therefore a problem affecting many sectors, and goes beyond political or geographical boundaries.

### 1.3.1. Declining trends for fishery yields in Lake Victoria

Until the 1960s, Lake Victoria was traditionally regarded as a cichlid lake on account of the predominance of these species in the fish community (Greenwood, 1966). The cichlid fauna included two indigenous tilapiine species (Graham, 1929) and, through endemic speciation (Greenwood, 1965), more than 300 species of haplochromines, which alone comprised some 80% of the demersal fish biomass (Greenwood, 1966; Kudhongania & Cordone, 1974; Van Oijen *et al.*, 1981). An early lake-wide fish stock assessment survey by Kudhongania & Cordone (1974) during 1969-1971 provides some baseline estimates (Table 1.4) for the fish biomass in Lake Victoria.

Originally, the diverse fish fauna supported a mainly artisanal-multi species fishery (Witte & van Densen, 1994). Besides the tilapiines *Oreochromis esculentus* and *Oreochromis variabilis*, which formed the base of the most important commercial fishery (Graham, 1929; Fryer & Iles, 1972; Lowe-McConnel 1956, 1987; Marten, 1979), not less than 11 genera, including lungfish *Protopterus aethiopicus*, the catfishes *Bagrus docmak* and *Clarias gariepinus*, and the cyprinid *Labeo victorianus*, supported continuously or intermittently a significant fishery (Witte & van Densen, 1994). Haplochromines, due to their small size and body texture, had little commercial value and hence were among the least targeted (Scully, 1975). In this early multispecies fishery, the two indigenous tilapia were the most exploited and their inshore distribution in Lake Victoria contributed to this significance (Twongo, 1995).

By the 1960s, stocks of the native tilapiines and other large species had been reduced by overfishing. To replenish the declining commercial stocks of indigenous fishes in order to improve the poor state of fisheries in Lake Victoria, four exotic tilapiines, Nile tilapia (*Oreochromis niloticus*), *Oreochromis leucostictus*, *Tilapia zillii*, and *Tilapia melanopleura*, were introduced into the lake during the early 1950s (EAFFRO, 1964, Welcomme, 1967), followed by the introduction of Nile perch (*Lates niloticus*) in the early 1960s (Gee, 1965; Welcomme, 1988).

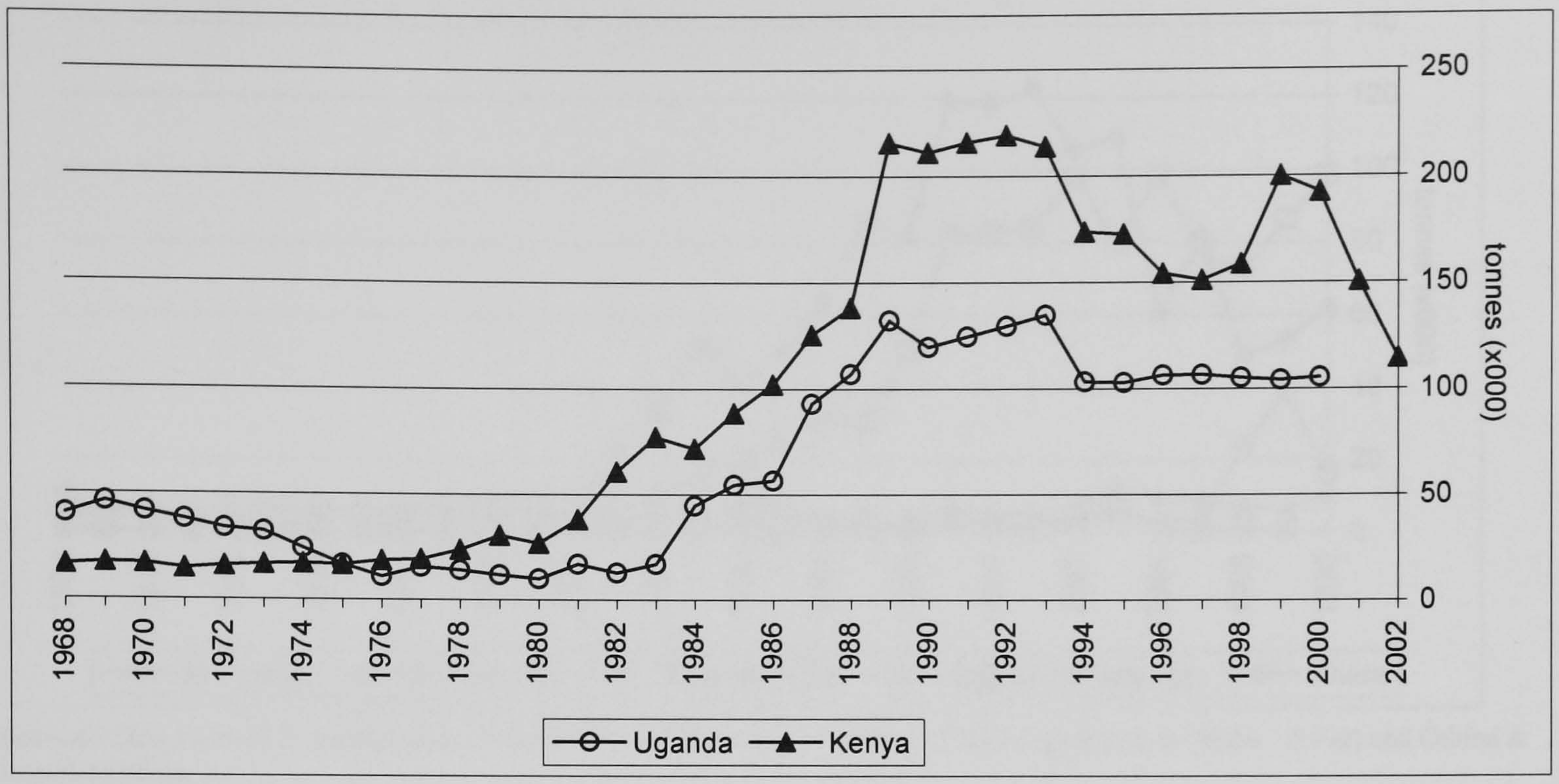
**Table 1.4:** Estimates of standing stocks (in metric tons) for Lake Victoria in 1969-1971.

| | Kenya | Tanzania | Uganda | All areas | |
|---|---|---|---|---|---|
| **Species:** | | | | | % |
| Haplochromine spp. (cichlids) | 36,694 | 321,282 | 205,592 | 563,568 | 83.00 |
| Lates niloticus (Nile perch) | 93 | 7 | 302 | 402 | 0.06 |
| Protopterus aethiopicus (lungfish) | 896 | 5,022 | 3,625 | 9,543 | 1.41 |
| Schilbe mystus (butterfish) | 45 | 547 | 54 | 646 | 0.10 |
| Mormyrus kannume (elephant-snout fish) | 28 | 119 | 181 | 328 | 0.05 |
| ***Catfishes:*** | | | | | |
| Bagrus docmak | 2,788 | 22,131 | 14,766 | 39,685 | 5.84 |
| Clarias mossambicus | 1,936 | 14,138 | 10,885 | 26,959 | 3.97 |
| ***Mochokidae spp.*** | | | | | |
| Synodontis victoriae | 309 | 13,256 | 9,644 | 23,209 | 3.42 |
| Synodontis afrofisheri | 42 | - | 9 | 51 | 0.01 |
| ***Tilapias:*** | | | | | |
| Oreochromis niloticus (Nile tilapia) | 48 | 912 | 535 | 1,495 | 0.22 |
| Oreochromis variabilis | 12 | 88 | 367 | 476 | 0.07 |
| Oreochromis leucostictus | - | - | 29 | 29 | 0.00 |
| Oreochromis esculentus | 636 | 9,569 | 1,624 | 11,829 | 1.74 |
| Tilapia zillii | - | 6 | 53 | 59 | 0.01 |
| ***Cyprinids:*** | | | | | |
| Rastrineobola argentea (sardine) | - | - | - | - | - |
| Labeo victorianus | 14 | 53 | 1 | 68 | 0.01 |
| Barbus altianalis (barbell) | 21 | 77 | 115 | 213 | 0.03 |
| *TOTAL* | *43,627* | *387,351* | *248,029* | *679,007* | *100.01* |
| Area (ha) | 416,730 | 3,767,490 | 3,139,890 | 7,324,110 | |
| Density (kg/ha) | 104.7 | 102.8 | 79.0 | 92.7 | |

***Source:*** Kudhongania & Cordone (1974).

Nile perch was officially introduced into Lake Victoria in May 1962 for the first time, when 35 individuals were introduced off Entebbe in Uganda, despite an unsettled controversy for and against the move (Graham, 1929; Beverton, 1959; Corbet, 1959; Fryer, 1960; Stoneman, 1960; Hamblyn, 1960a, 1960b, 1962; Anderson 1961). In September 1963, the lake was stocked with other 339 fishes. The purpose of the introduction of Nile perch was to "utilize" haplochromines, which were otherwise of little commercial importance (Acere, 1985). However, for over 20 years after its introduction into the lake, while traditional fish catches continued to decline (Figure 1.2), *Lates* catches were minimal.

**Figure 1.2:** Annual catches of total fish biomass (all species) from the Kenyan and Ugandan waters of Lake Victoria.

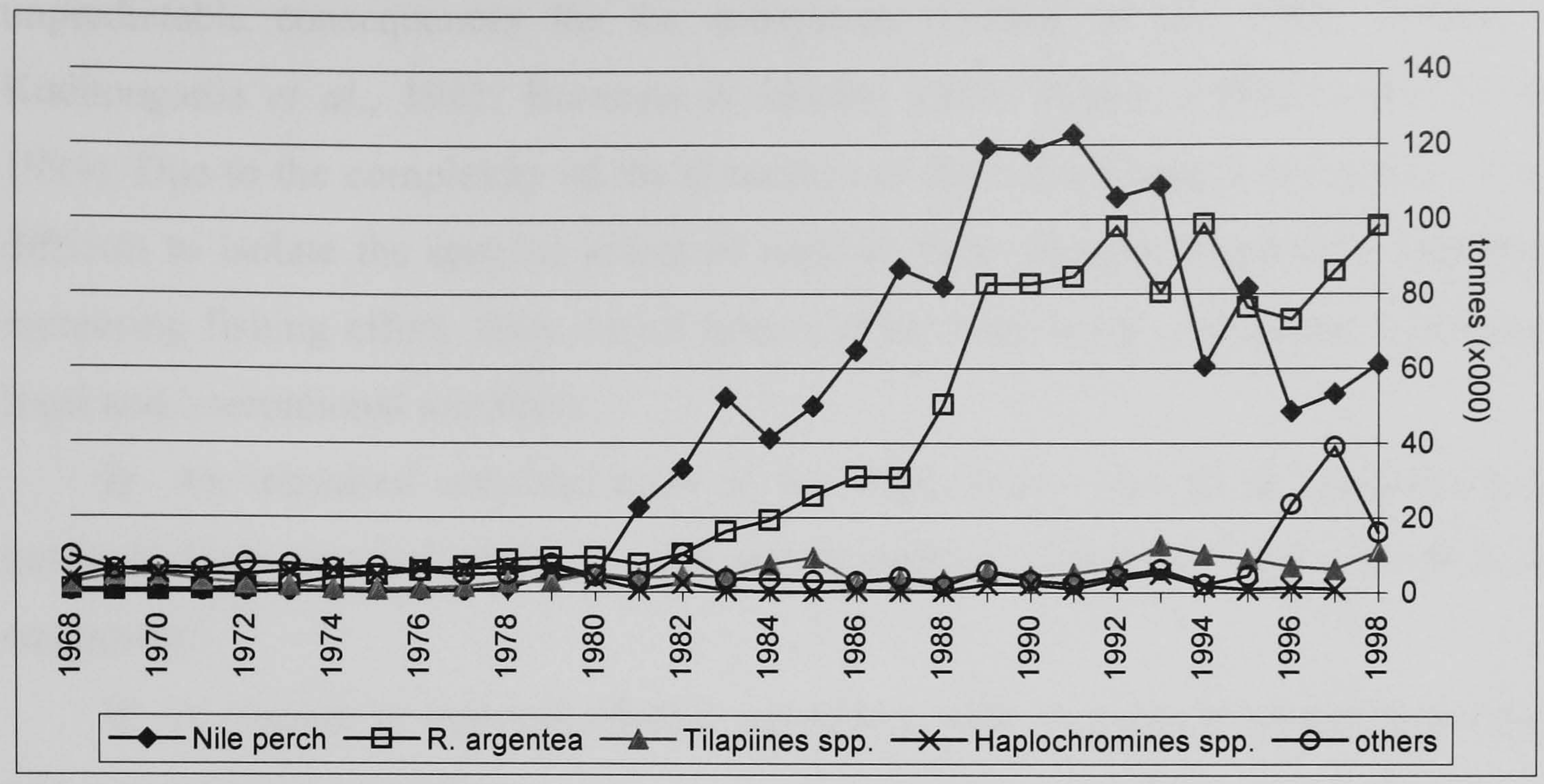


***Sources:*** Kenyan data 1968-1985 and 1997-2002 from Kenya Fishery Department; Kenyan data 1986-1996 from KMFRI, published in Othina (1999) and Othina & Tweddle (1999); Ugandan data from Uganda Fisheries Department, published in Namisi (2000).

*Lates* catches became significant only from 1975, 1977 and 1978, in the Ugandan, Kenyan (Figure 1.3) and Tanzanian sectors of the lake respectively (Ssentongo & Welcomme, 1985). In the 1980s, *Lates* catches increased exponentially rising in few years from virtually zero to almost 60% of total yield in Kenya (Okemwa, 1984; CIFA, 1988; Ogari & Asila, 1990; Ogutu-Ohwayo, 1990; Ssentongo & Welcomme, 1985), Uganda (Okaronon *et al.*, 1985; Kudhongania & Chitamwebwa, 1995), and Tanzania (Ligtvoet *et al.*, 1988; Bwathondi, 1990; Ogutu-Ohwayo, 1990).

Along with the rapid increase in the stocks of Nile perch, the composition of fish biomass in the lake changed dramatically following ecological changes which affected virtually all levels in the ecosystem (Witte *et al.*, 1992a). The result has been a dramatic reduction in species richness and commercial extinction of many indigenous fishes. Haplocromines, which once accounted for about 80% of the lake fish biomass, were the most affected. By 1987 their contribution to the demersal ichthyomass had decreased to less than 1% (Witte *et al.*, 1992a), although there is new evidence of a slow increase in biomass between 1991 and 1995 (Seehausen *et al.*, 1997; Witte *et al.*, 2000).

**Figure 1.3:** Annual catches from the Kenyan waters of Lake Victoria.



*Sources:* Data 1968-1975 adapted from Ochumba (1995); data 1976-1998 from KMFRI published in Othina (1999) and Othina & Tweddle (1999).

Following the dramatic decline in species diversity, the new fishery is now dominated by three species: the introduced predator Nile perch, the exotic herbivore Nile tilapia and an indigenous zooplanktivore *Rastrineobola argentea*. In fact, while all other species declined, only Nile tilapia and *Rastrineobola argentea* increased in the presence of increasing population of Nile perch. After its boom in the early 1980s, Nile perch fishery continued to grow in the following years. In Kenya, aggregate landings peaked at a maximum of more than 210,000 tonnes during the 1989-1993 period (Figure 1.2). However, since 1994 fish landings have been in sharp decline, mostly due to declining catches of Nile perch (Figure 1.3). By 1998, Nile perch landings were half of those at the beginning of the decade despite increased effort. A similar scenario has also been observed in Uganda and Tanzania.

These trends raise the question as to whether catch levels can be sustained on a long-term basis. There is an urgent need to understand the causes of fish stock decline in order to take adequate management measures to reverse the trend. Given the role of Lake Victoria in providing foreign currency and food sources to local populations, sustainable fishery management is a priority, particularly when the agricultural productivity of the lake basin is also suffering a long-term declining trend (§ 1.3.2).

It is now acknowledged that single factors are inadequate by themselves to account for the evolution of present fisheries and ecological status of the lake. It is instead believed that several biotic and abiotic factors have interacted synergistically with sometimes unpredictable consequences for the ecosystem (Coulter *et al.*, 1986; Bruton, 1990; Kudhongania *et al.*, 1992; Bootsma & Hecky, 1993; Hecky, 1993; Lowe-McConnell, 1994). Due to the complexity of the dynamics of the Lake Victoria ecosystem, it is very difficult to isolate the specific effect of each of these factors. In general, together with increasing fishing effort, three major causes of environmental change are highlighted by local and international scientists:

**i)** An increased eutrophication of the lake's water due to an increased input of nutrients from external sources, such as atmospheric deposition and run off from the catchment;

**ii)** A change in regional climate associated with changes in precipitation regimes and increasing temperatures;

**iii)** The long-term consequences of invasive species (Nile perch and exotic tilapiines) introduction which led to the disruption of energy flow patterns and may have exerted a certain influence on the eutrophication process.

#### *1.3.1.1. Overfishing*

Since the beginning of the last century, the demand for fish from Lake Victoria was stimulated by the development of urban centres along the lake's shore and the arrival of the railway at the Nyanza Gulf in 1908 (Graham, 1929). At this time the traditional Lake Victoria fishery was exploited by simple fishing gears, such as basket traps, hooks and seine nets of papyrus, each of which exerted little fishing pressure (Kudhongania & Chitamwebwa, 1995). However, the introduction of the more efficient flax gill net into the fishery in 1906 and its subsequent widespread use by 1916, together with the sustained use of beach seines (Mann, 1969), greatly increased fishing pressure on the larger and more commercially preferable *Orochromis esculentus*, such that by 1920 localized overfishing of the species had set in (Graham, 1929). According to data from early studies (Beverton, 1959; Garrod, 1961), initial catch rates of more than 25 fish per net-night declined to about 2 fish by 1940. Similar trends were also recorded for *Labeo* and other major commercial fish species (Whitehead, 1959). For this reason, in order to maintain the level of the catch, around the late 1960s the fishery shifted from gillnetting fishery, mostly targeting tilapias, to bottom trawling, exploiting smaller species such as haplochromines (Tumwebaze *et al.*, 2002). In Kenya commercial trawling was carried out as early as 1968 (Mbuga *et al.*, 1998), while in Tanzania it started only in 1976 in the Mwanza Gulf (Witte, 1981). Here, however, as elsewhere else in the lake, this fishery was not sustainable. As catch rates fell (Witte & Goudswaard, 1985), trawling became uneconomical and some boats switched to trawling for Nile perch when it became more abundant (Reynolds & Greboval, 1988).

These historic evidences show that overfishing has been a problem in Lake Victoria from as early as the 1920s. However, the profound transformation of the fishing industry following the Nile perch explosion during the 1980s leaded to a much more dramatic increase in fishing effort. In those years, the success of the Nile perch fishery was recognised not only in East Africa but also internationally, and a thriving overseas market for the species was established (Crean *et al.*, 2002). Inevitably this lucrative international trade in Nile perch attracted other users, and brought an unprecedented inflow of national and international capital, transforming the once locally-based fishery. In Kenya, for instance, within a few years the fish processing capacity grew from nil in the early 1980s to 15 registered factories with a processing capacity far in excess of the level commensurate with the resource productivity (Bokea & Ikiara, 2000). In Uganda the trend was similar and was further encouraged by the investment promotion policies introduced by the government in 1991. These policies have brought the number of fish processing firms from 3 in 1990 to 10 in 1998, and processing capacity from 90 to 295 tonnes per day (UNEP, 1999). Now, around the whole Lake Victoria shore, there are 35 fish factories (Abila & Jansen, 1998). Most have a processing capacity that far exceeds the amount of fish they are able to obtain (Asowa-Okwe, 1996; Jansen 1997). All these plants are processing Nile perch for export markets. In addition to that, during the 1990s a fishmeal-based animal feeds industry developed with enormous consequences for the stocks of the lake's sardine *Rastrineobola.* In Kenya, for instance, in less than a decade a total of 8 animal feeds factories have prospered, 6 using large amounts of *Rastrineobola* as input, while 2 use Nile perch skeletons to produce fishmeal (Bokea & Ikiara, 2000).

In order to meet the steady increase in demand by processing factories, more fishermen were recruited to work in the lake and many seasonal and part-time fishermen became full-time fishermen. Then, the number of fishermen in Lake Victoria increased progressively in all the three lacustrian countries (Table 1.5). In the Kenyan part of the lake it rose from about 11,000 in 1973 to 30,000 in 1995, reaching a peak of 33,000 in year 2000. Analogously, Tanzania passed from about 16,000 fishermen in 1973 to 35,000 and 56,000 in 1995 and 2000 respectively. Also Uganda recorded an increase from about

8,000 fishermen in 1972 to almost 35,000 in year 2000. The same increasing trend has been followed by the number of boats in all the three countries.

**Table 1.5:** Historical trends in fishing effort in Lake Victoria.

| Year | Tanzania | | Kenya | | Uganda | |
|---|---|---|---|---|---|---|
| | Boats | Fishers | Boats | Fishers | Boats | Fishers |
| **1970** | 3785 | 12091 | — | — | — | — |
| **1971** | 3302 | 14131 | — | — | 3264 | — |
| **1972** | 3494 | 14690 | — | — | 2700 | 8100 |
| **1973** | 3976 | 15917 | 4100 | 10700 | — | — |
| **1974** | 3332 | 13381 | — | — | — | — |
| **1975** | 3918 | 15037 | — | — | — | — |
| **1976** | 3794 | 14773 | — | — | — | — |
| **1977** | 3398 | 14327 | — | — | 3000 | — |
| **1978** | 3448 | 17654 | — | — | — | — |
| **1979** | 4457 | 20937 | 4600 | 18000 | — | — |
| **1980** | 3997 | 20587 | — | — | — | — |
| **1981** | 4199 | 20787 | — | — | — | — |
| **1982** | 4245 | 18263 | 5000 | 20000 | — | — |
| **1983** | 4141 | 15194 | 5000 | 20000 | — | — |
| **1984** | 4650 | 17827 | — | — | — | — |
| **1985** | 4408* | 17086* | 5500 | 21500 | — | — |
| **1986** | 7404* | 24241* | 5000 | 21000 | — | — |
| **1987** | 6667* | 22207* | — | — | 4000 | — |
| **1988** | 6546* | 22926* | 5000 | — | 3470 | — |
| **1989** | 7757* | 29816* | 5500 | 22000 | 8000 | — |
| **1990** | 7797* | 29095* | — | — | — | — |
| **1991** | 5948* | 25900* | 6229 | 24000** | — | 30000 |
| **1992** | 5041* | 20064* | 7279** | 25000** | — | — |
| **1993** | 4185* | — | 7279** | 25000** | — | — |
| **1994** | — | — | 7425** | 25000** | — | — |
| **1995** | 7953* | 34832* | 8000** | 30000** | — | — |
| **1996** | — | — | — | — | — | — |
| **1997** | — | — | — | — | — | — |
| **1998** | 7618* | 32403* | — | — | — | — |
| **1999** | — | — | — | — | — | — |
| **2000** | 15533*** | 56443*** | 10014*** | 33037*** | 15544*** | 34889*** |

*Sources:* *data from TAFIRI cited in Mkumbo & Cowx (1999); **data from several sources cited in Ikiara (1999); ***LVFRP (2001); otherwise, data from several sources cited in Reynlods *et al.* (1995).

The general increase in fishing pressure during the 1980s led to the already mentioned spectacular increase in the amount of fish landed. In Kenya, for instance, fish landings were supported by an increase in aggregate catch per unit effort (CPUE) from a value of 82 kg boat$^{-1}$ day$^{-1}$ in 1986 to 180 kg boat$^{-1}$ day$^{-1}$ in 1989 (Othina, 1999; Othina & Tweddle, 1999). Other studies (Reynolds *et al.*, 1995), restricted to the Nile perch fishery alone, reported values of 56.4 kg boat$^{-1}$ day$^{-1}$ in 1986 and 67.4 kg boat$^{-1}$ day$^{-1}$ in 1989 (Table 1.6). This underlines the role played by increasing *Rastrineobola* stocks in explaining the CPUE trend for the aggregate catch.

However, after its peak in 1989 aggregate CPUE decreased with increasing effort. A CPUE of 86 kg boat$^{-1}$ day$^{-1}$ in 1998 (Table 1.7) was almost the same as the value reported in 1986. The situation in the Kenyan gillnet fishery, which targeted Nile perch and tilapias, was worse than the data suggest, as effort is expressed in boat days and does not consider the number of gears per boat (Othina & Tweddle, 1999). Rabuor & Polovina (1995) also noticed that mesh sizes were getting smaller, nets were getting longer and soak times were increasing. These changes in fishing strategies are responses to declining catch rates and these trends have continued.

**Table 1.6:** Trends (1975-1989) in CPUE (kg boat$^{-1}$ day$^{-1}$) for Nile perch in Lake Victoria.

| Year | Tanzania | Kenya | Uganda |
|---|---|---|---|
| **1975** | 39.7 | — | — |
| **1976** | 40.0 | — | — |
| **1977** | 56.7 | — | 14.2 |
| **1978** | 42.2 | — | — |
| **1979** | 38.9 | 18.4 | — |
| **1980** | 50.1 | — | — |
| **1981** | 51.0 | — | — |
| **1982** | 44.1 | 33.2 | — |
| **1983** | 52.6 | 42.5 | — |
| **1984** | 60.0 | — | — |
| **1985** | 75.9 | 44.1 | — |
| **1986** | 87.4 | 56.4 | — |
| **1987** | 69.0 | — | 63.8 |
| **1988** | 102.5 | 68.5 | 84.7 |
| **1989** | 79.4 | 67.4 | — |

***Source:*** Reynolds *et al.* (1995).

**Table 1.7:** Trends (1989-1998) of total biomass CPUE (kg boat$^{-1}$ day$^{-1}$) for Kenyan commercial fisheries of Lake Victoria for different type of fishing gear.

| Year | Gillnets | Long lines | Beach seines | Mosquito seines | ALL GEARS |
|---|---|---|---|---|---|
| **1989** | 145.2 | 64.7 | 387.8 | 256.6 | **180** |
| **1990** | 92.3 | 63.9 | 449.8 | 240.8 | **152** |
| **1991** | 66.9 | 59.8 | 339.2 | 221.4 | **145** |
| **1992** | 64.6 | 44.3 | 301.7 | 220.2 | **137** |
| **1993** | 45.5 | 43.1 | 222.3 | 203.8 | **115** |
| **1994** | 36.4 | 43.3 | 52.3 | 210.0 | **93** |
| **1995** | 41.0 | 30.9 | 118.7 | 147.4 | **86** |
| **1996** | 67.3 | 42.5 | 108.7 | 113.8 | **78** |
| **1997** | 58.9 | 45.5 | 120.6 | 133.7 | **93** |
| **1998** | 47.3 | 53.8 | 164.7 | 164.4 | **86** |

**Source:** Othina & Tweddle (1999).

Uganda and Tanzania showed a similar pattern in both total aggregate landings and CPUE, supporting the evidence of overfishing. Indeed, the declining trend of CPUE over the last decade in Lake Victoria is a classic indicator of overexploitation. Other fishery indicators, such as size at first capture, length at first maturity, and paucity of larger fish caught, may also be used to support the evidence of overfishing (Caddy & Mahon, 1995).

There is evidence of decreasing length at first capture, in particular for the Nile perch. Experimental trawling over several years reported that in 1988 the length at first capture for Nile perch was 70-80 cm TL (Ligtvoet & Mkumbo, 1991), which decreased to 50-60 cm TL in 1992 and to 40-50 cm TL in 1994, remaining around the same level until the end of the 1990s (Mkumbo *et al.*, 2002). Much of this decline in size at first capture has been linked to increased overall effort and reduction in mesh size, which now seems to have stabilized around 12.5 cm (Mkumbo *et al.*, 2002). In this respect, it has been suggested that the Nile perch gillnet fishery of Lake Victoria should be based on mesh sizes of 12.7 cm or over (Schindler *et al.*, 1998).

Reduction in size at first maturity, due to the ability of fishes to adjust their biological characteristics in order to compensate for a reduction in their stock, may also be an indicator of overfishing (Caddy & Mahon, 1995). Length at first maturity is defined as the minimum length at which gonadal development has advanced to a reproductive stage in 50% of individuals. Early maturing populations tend to grow quickly to reproductive size and then divert energy away from somatic growth to reproduction. This tactic has been adopted by many animals living in unstable environments where adverse biotic (i.e. competition, predation or overfishing) and abiotic (i.e. eutrophication) factors are prevalent (Charlesworth & Leon, 1976; Stearns, 1976). However, in Lake Victoria it is difficult to distinguish whether this is due to lower growth rates caused by natural biotic elements (i.e. predation, competition, invasive species) or to heavy fishing pressure selecting in favour of earlier maturing individuals (Getabu & Nyaundi, 1999).

In the case of Nile perch, for instance, surveys in Tanzanian waters showed that length at first maturity decreased from 60 cm TL in males and 95-100 cm TL in females in 1988 (Ligtvoet & Mkumbo, 1991), to 53 cm TL and 55 cm TL in males and females respectively during 1997-1999 (Mkumbo & Ezekiel, 1999).

For Nile tilapia, since the early study on size at first maturity by Wanjala & Marten (1976), there has been no significant downwards trend. However, a catch survey in the Kenyan waters of the Nyanza Gulf during 1998-1999 reported length at first maturity for Nile tilapia at 28-30 cm TL for females and 32-34 cm for males (Ojuok, 1999). This earlier maturation of females relatively to males is in contrast with the findings by Balirwa (1998) in the Ugandan sector of the lake, where males first spawn at about 28 cm TL against the 32 cm TL of females. The early maturation of females in the Kenyan sector might be again a tactic to maximise reproductive success (Stearns, 1976; Charlesworth & Leon, 1976) possibly linked to a population response to overfishing, which is thought to be relatively heavier in Kenya than in Uganda (Ojuok, 1999).

For *Rastrineobola*, early studies during 1970-1974 in the Nyanza Gulf reported length at first maturity of 52 mm SL for males and 44 mm SL for females (Okedy, 1974). Recently, Wandera (1999) found that male *Rastrineobola* matured at 41 mm SL, while females at 42 mm SL. From other samples collected during 1999-2000, it was estimated that the average length at first capture for the whole lake population was 38.7 mm (Marshall, 2001). However, it varied between 39.4 mm in Tanzania and 36.8 mm in Kenya (Marshall, 2001). Again, the Kenyan status may be due to a higher degree of fishing pressure. Moreover, comparison between the results of these late studies and those during the early 1970s shows that *Rastrineobola* exhibits a declining trend in length at first maturity over the last decades.

The paucity of larger fish is another sign of heavy fishing pressure on stocks. A relative small number of large mature fish is of concern as there is potentially a lack of spawning stock. During the last few years there have been evidences of this kind for both Nile perch and Nile tilapia. Bottom trawl surveys in the Kenyan waters of Lake Victoria during 1997-1998 caught few Nile perch greater than 80 cm TL and *Oreochromis niloticus* greater than 50 cm TL (Getabu & Nyaundi, 1999), even though these species attain maximum sizes of 205 cm TL and 65 cm TL respectively (Asila & Ogari, 1988; Getabu, 1992).

In conclusion, data from literature and field studies provide evidence which supports the idea that Lake Victoria fisheries are overexploited. However, the declining stocks of commercial fishes in Lake Victoria over the last decades may also be related to other elements, such as perturbations within the food web and eutrophication of the lake's water. Nevertheless, there is an urgent need for the three riparian countries to act rapidly in order to implement a common strategy for regulating the fishery within the lake.

#### *1.3.1.2. Eutrophication*

Eutrophication is another important factor in explaining the recent decline of fish stocks in Lake Victoria. Many water quality analyses over the last few decades show that Lake Victoria has progressively shifted from a mesotrophic to an eutrophic state (Table 1.8). In 1960-1961, for instance, Talling (1965) reported a chlorophyll-a concentration,

Table 1.8: Water quality parameters for Lake Victoria (Kenyan and Ugandan waters).

| Year | Chl-a (µg/l) | PO$_4$ (µg/l) | TP (mg/l) | NO$_3$ (µg/l) | TN (mg/l) | Secchi depth (m) | Place | Source |
|---|---|---|---|---|---|---|---|---|
| | | | | | | | **Offshore (Kenya)** | |
| 1930 | — | — | — | — | — | 7.3-7.9 | Offshore (Kenya) | Worthington (1930) |
| 1986 (Feb-Aug) | 8-77.6 | — | — | — | — | 0.2-1.2 | Offshore (Kenya) | Ochumba & Kibaara (1989) |
| 1990 | — | — | 0.07-0.103 | 10-30 | 0.44-1.16 | 1.8 | Offshore (Kenya) | Gophen *et al.* (1995) |
| 1994-1995 | 2-21 (dry season) / 2-13 (rainy season) | — | — | 100 | — | 0.4-1.35 | Offshore (Kenya) | Lung'ayia *et al.* (2000) |
| 1998 (June) | — | 87 | — | 117 | — | 0.9-2.4 | Offshore (Kenya) | Lung'ayia *et al.* (2001) |
| 1997-1999 | 6.3-14.1 | 22-90 | — | 20-180 | — | 0.75-2.6 | Offshore (Kenya) | Kenyanya (1999) |
| | | | | | | | **Inshore (Kenya)** | |
| 1930 | — | — | — | — | — | 1.3-1.45 | Nyanza gulf (Kenya) | Worthington (1930) |
| 1960-1961 | 21 | — | — | — | — | — | Nyanza gulf (Kenya) | Talling (1965, 1966) |
| 1976 | 17 | — | — | 8 | — | 0.75-1.4 | Nyanza Gulf (Kenya) | Melack (1976b) |
| 1986 (Feb-Aug) | 31.4 | — | — | — | — | — | Nyanza Gulf (Kenya) | Ochumba & Kibaara (1989) |
| 1994-1995 | 9.3-71.5 (dry seas.) / 8.8-17.2 (rainy seas.) | — | — | — | — | 0.58 - 0.68 | Nyanza Gulf (Kenya) | Lung'ayia *et al.* (2000) |
| 1998 (June) | 61 (dry season) | — | — | 128 | — | 0.5-1 | Nyanza Gulf (Kenya) | Lung'ayia *et al.* (2001) |
| 1997-1999 | 10.7-38.1 | 41-77 | — | 30-330 | — | 0.5-1 | Nyanza Gulf (Kenya) | Kenyanya (1999) |
| | | | | | | | **Offshore (Uganda)** | |
| 1960-1961 | 1.2-5.5 | 10 | — | 10-112 | — | 6.4-8.4 | Offshore (Uganda) | Talling (1965) |
| 1989-1991 | 8.4-40 | — | — | — | — | 1.3-3 | Offshore (Uganda) | Mugidde (1993) |
| 1990 | 11-23 | — | 0.062-0.124 | 11-84 | 0.336-0.448 | 1.2 | Offshore (Uganda) | Hecky (1993) |
| 1991 (Apr) | 13.1 (rainy season) | — | — | — | — | — | Offshore (Uganda) | Lehman & Branstrator (1993) |
| 1992 (Apr) | 11.2 (rainy season) | — | — | — | — | — | Offshore (Uganda) | Lehman & Branstrator (1993) |
| 1994-1998 | 13.5 | | | | | | Offshore (Uganda) | Mugidde (2001) |
| | | | | | | | **Inshore (Uganda)** | |
| 1960-1961 | 10-15 | — | — | — | — | — | Pilkington Bay (Uganda) | Talling (1965) |
| 1989-1991 | 22.2-67.1 | — | — | — | — | — | Pilkington Bay (Uganda) | Mugidde (1993) |
| 1992 (Apr) | 24.3 (rainy season) | — | — | — | — | — | Pilkington Bay (Uganda) | Lehman & Branstrator (1993) |
| 1994-1998 | 71 | | | | | | Napoleon Gulf (Uganda) | Mugidde (2001) |

which is a measure of algal/phytoplankton biomass, of 1.2-5.5 $\mu g\ l^{-1}$ in Ugandan offshore waters and 10-15 $\mu g\ l^{-1}$ in the inshore waters of Pilkington Bay. After 30 years, in the same waters, Mugidde (1993) measured a dramatic increase to 8.4-40 $\mu g\ l^{-1}$ and 22.2-67.1 $\mu g\ l^{-1}$ for offshore and inshore concentrations values respectively. These findings were also confirmed by other studies (Lehman & Branstrator, 1993; Hecky, 1993). Similar increasing trends on chlorophyll-a concentration have been reported for the Kenyan part of Lake Victoria (Ochumba & Kibaara, 1989; Gophen *et al.*, 1995; Kenyanya, 1999; Lung'ayia *et al.*, 2000; Lung'ayia *et al.*, 2001) against baseline values provided by Talling (1965, 1966) and Melack (1976b). Other water quality variables, such as mean water transparency (Secchi depth), nitrates, and phosphates concentrations show a similar path.

Even though recent water quality studies for Lake Victoria showed a clear pattern towards eutrophication, it is believed this process had already began in the 1920s, and became apparent in the early 1960s (Hecky, 1993; Bootsma & Hecky, 1993). Indeed, after at least 140 years of adequate year-round bottom oxygenation (Verschuren *et al.*, 2002), the seasonal deep-water anoxia first observed in 1960-1961 by Talling (1966) represented the earliest stage of eutrophication-induced deep water oxygen loss in Lake Victoria.

Changes in water quality were accompanied by a change in phytoplankton species composition, shifting from a state dominated by green algae and diatoms to the present state dominated by cyanobacteria blue-green algae. Analyses of fossil diatom data from permanently buried assemblage show that the pelagic diatom community of Lake Victoria was stable between 1820 and 1940 before increasing in abundance between about 1940 and the early 1960s (Verschuren *et al.*, 2002). However, in the mid-1980s fossil-diatom records reveal that there was a restructuring of the algal community to cyanobacteria dominance (Verschuren *et al.*, 2002). This evidence is also confirmed by extensive cyanobacteria algal blooms observed after 1987 by Ochumba & Kibaara (1989). Now, the phytoplankton community, which once showed clear seasonal successions of diatoms, cyanobacteria and green algae and several other taxa (Talling, 1965, 1987), is persistently dominated by cyanobacteria (Lung'ayia *et al.*, 2000). Diatoms, in particular, were out-competed because of their high *Si* needs and high sinking rate (Talling, 1966; Kilham, 1990). As consequence, algal biomass is now five times higher than in the 1960s and phytoplankton primary productivity has doubled (Mugidde, 1993), causing a four-fold decrease in water transparency.

Because cyanobacteria are generally considered a poor food source (Lampert, 1981; Heerkloss *et al.*, 1984; Haney, 1987), the recent shift in phytoplankton species composition in Lake Victoria may be expected to result in a lower efficiency of energy transfer to higher trophic levels (Bootsma & Hecky, 1993). However, the ability of some tilapiines to digest cyanobacteria (Moriarty, 1973; McDonald, 1987) may have created a better environment for some herbivorous fishes. On the contrary, such an environment would not favour zooplanktivorous fish species, since cyanobacteria are a poor food for zooplankton (Bootsma & Hecky, 1993). Indeed, in Lake Victoria changes in phytoplankton have been accompanied by changes in the composition of zooplankton community (Mavuti & Litterick, 1991; Gophen *et al.*, 1992). Also the composition of the benthic macrofauna (insects and mollusca) has been reported to have changed over the last decade (Muli & Mavuti, 2001).

A direct consequence of eutrophication in Lake Victoria is the oxygen depletion of its waters. The higher rate of photosynthesis from the more abundant phytoplankton biomass causes both higher mean oxygen concentrations in surface waters compared to the past, and deoxygenation of deep waters (Hecky, 1993; Hecky *et al.*, 1994; Mugidde, 1993). Gophen *et al.* (1995), comparing their measurements with earlier studies (Talling & Talling, 1965; Talling, 1966; Ochumba & Kibaara, 1989; Ochumba, 1987), suggested that during the last two decades the volume of hypoxic water in Lake Victoria has increased, and stratification is more long-lasting. Indeed, algae primary production in the epilimnion is now much greater than grazing capacity and, upon death, excess organics flux to the hypolimnion. The large algal biomass sediment to the bottom of the lake and the ensuing decomposition processes of dead algae contribute to the anoxic conditions now observed. This may explain the shift of the anoxia isoplet (oxycline) at a shallower level in Lake Victoria. In fact, in 1960-1961 deoxygenation to less than 1 mg $l^{-1}$ DO lasted from January to June, but anoxia was not shallower than 54 m (Talling, 1966). However, a recent study by Hecky (1993) reported an oxycline around 45 m. This evidence is also supported by Gophen *et al.* (1995) who recorded the DO isopleths of 1 mg $l^{-1}$ between 18-48 m, with most of the deep water in Lake Victoria below 34 m being consistently anoxic.

One consequence of low oxygen in deep water (persisting throughout the stratified season) is the enhancement of denitrification rates (Hecky, 1993). Enhanced denitrification in the water column results in low ratios of dissolved nitrogen to phosphate in deep waters. Mixing of deep and surface water, either across the thermocline or during annual mixing, results in a low N:P loading ratio for surface water. This favours blue-green cyanobacteria, which are able to produce nitrogen on their own through atmospheric fixation.

Fish stocks in Lake Victoria may have been severely affected by the deoxygenated conditions caused by the eutrophication process. As large cyanobacteria are observed to become more abundant in the open lake (Ochumba & Kibaara, 1989), frequent localized fish kills have been recorded in nearshore shallow waters in association with temporary anoxic conditions (Ochumba, 1987, 1990). Massive fish mortality has been observed as more than 2,400 tonnes of fish died in 1984 in the Nyanza Gulf due to anoxia (Ochumba, 1990). Even though these events may have occurred occasionally in the past under sporadic climatic conditions (Newell, 1957), the consequences of the phenomena are worsened by the effects of eutrophication. Moreover, the general shallower level at which the oxycline is now encountered is limiting the habitat extent of many species that are unable to survive in anoxic waters. During the 1960s deoxygenated waters covered a bottom area 2.5 times smaller than at the present (Gophen *et al.*, 1995). This progressive reduction in habitat area may have initially favoured high catches of Nile perch but could have led to overexploitation in the long run.

Eutrophication may also explain the commercial disappearance of haplocromine species in Lake Victoria during the last decades. It is now hypothesized that the increasing turbidity and deoxygenation, caused by eutrophication, broke down the food web through a combination of demersal haplocromines displacement and increased invertebrate abundances (Hecky, 1993). Increased stocks of insect larvae and especially the lake shrimp, *Cardinia nilotica* (Witte *et al.*, 1992b), favoured survival of juvenile Nile perch, which feed largely on invertebrates (Ogutu-Ohwayo, 1990a). Increased survivorship of juvenile perch led to explosive growth of the adult perch population which decimates haplocromines and drastically reduced the biodiversity within the lake.

The increased external input of nutrients through superficial loading and rainfall is believed to be the main cause of eutrophication in Lake Victoria. Bottom sediment analysis by Hecky (1993) showed a substantial increase in sediment concentrations for both nitrogen and phosphorus during the last fifty years. This trend is much more significant for phosphorus, even though it began earlier for nitrogen (around the 1920s) than it did for phosphorus (around 1950s). Sedimentary phosphorus increased by 31% during 1950-1990, while sedimentary nitrogen increased by only 17% during 1920-1990. According to Hecky (1993), the increased deposition rates of nitrogen beginning in the 1920s and phosphorus beginning in the 1950s were likely the result of watershed and airshed disturbance. Clearing and burning of forest and savannah, for planting and grazing to sustain the rapidly increasing human population, first released nitrogen as the natural vegetation was removed (Likens *et al.*, 1970; Bayley *et al.*, 1992). This trend intensified until exposed soil and burning to stimulate soil fertility were widespread and led to mobilization of terrestrial phosphorus by water and wind erosion (Andreae, 1991). Many authors (Hecky & Bugenyi 1992; Scheren *et al.*, 2000) now believe that the observed changes in the phytoplankton community are the logical consequence of excess nutrient loading from the drainage basin of Lake Victoria. However, lack of long term monitoring data and the complexity of ongoing ecosystem changes hamper full appreciation of the impact of catchment disturbance on the lake's present condition (Verschuren *et al.*, 2002).

#### *1.3.1.3. New species introduction*

New species introduction within the fish community is not a new phenomenon. Since the middle of the 19th century, 1,354 introductions of 237 fish species into 140 countries have been documented (Welcomme, 1988). The introduction of non indigenous fishes into new aquatic habitats is often used to maintain or increase fisheries harvest, or for biological control of weeds or other pests (Schofield & Chapman, 1999). However, several studies around the world showed that the delicate balance of many aquatic ecosystems might be severely disrupted by such transfers (Zaret & Paine, 1973; Kempinger & Carline, 1977; Zaret, 1982). Negative impacts include predation on existing fish stocks, hybridisation, introduction of parasites, competition for food and other resources with indigenous fishes, and destruction or alteration of the existing food webs (Welcomme, 1984).

The impact of introductions in Lake Victoria has received a high public profile on account of the concern about the effects of Nile perch on endemic fishes in the lake due to the economic importance of its fisheries and its extremely high endemism (Pitcher & Hart, 1995). After the first fishery survey of Lake Victoria during 1927-1928, Nile perch introduction was suggested to feed on haplocromines that were abundant in the lake so as to convert them into a larger fish of greater commercial and recreational value (Graham, 1929). It is now recognised that the most dramatic and catastrophic fish introduction anywhere has been the transfer of Nile perch into Lake Victoria during the 1960s (Ogutu-Ohwayo, 1990a, 1990b, 1993). Over 50% of the more than 600 endemic fish species from the original community is known to have disappeared from Lake Victoria, including several hundreds haplocromine species (Hughes, 1986; Kaufman, 1992; Witte *et al.*, 1992a, 1992b; Kaufman & Ochumba, 1993; Kaufman *et al.*, 1997).

The introduced Nile perch is considered as the major contributor to the mass extinction since the decline and eventual collapse of haplocromine fishery during 1977-1983 was almost reciprocal with the increase in Nile perch abundance (Ogari, 1985; Ogutu-Ohwayo, 1990a, 1990b; Witte *et al.*, 1992a, 1992b). *Lates* is a generalist predator which selects the most abundant prey not greater than one-third of its length (Kudhongania & Chitamwebwa, 1995). This suggests that predation by Nile perch had considerable impact on the haplocromine stock. Okedi (1970) observed that haplocromines constituted 80% by weight of the food of *Lates*. Recent studies have revealed that the diet of adult *Lates* is now almost entirely comprised of *Cardinia nilotica*, a small microphagous prawn, and juvenile *Lates* (Hughes, 1986; Ogari & Dadzie, 1988). This shift in the feeding biology of Nile perch reflects the actual change in the composition of the native fish community. However, since *Lates* has become heavily dependent on cannibalism, an imminent crash of the fishery was recently predicted by many authors (Beadle, 1981; Ogutu-Ohwayo, 1990b).

The introduction of Nile perch did not act alone in explaining the changes within the freshwater ecosystem of Lake Victoria. The establishment of four exotic tilapiines in the lake since the early 1950s introduced interspecific competition with the two indigenous species, *Oreochromis esculentus* and *Oreochromis variabilis*. The introduced tilapias replaced the two endemic tilapia species through a variety of mechanisms (Lowe-McConnel, 1956; Fryer, 1961; Welcomme, 1964, 1966; Reinthal & Kling, 1994): *Tilapia zillii* out-competed *Oreochromis variabilis* for nursery areas; *Oreochromis leucostictus* thrived in suboptimal habitat in inshore areas where dissolved oxygen was lowest; and *Oreochromis niloticus* replaced *Oreochromis esculentus* offshore and also competed with *Oreochromis variabilis* for breeding grounds. The new species also enhanced the likelihood of genetic dilution through hybridisation (Lowe-McConnel, 1958; Welcomme, 1967).

The introduction of new species in lakes may also have serious limnological impacts which accelerate undergoing processes such as cultural eutrophication. In Lake Victoria a top-down process has been hypothesized in which exotic fish introductions, such as those that have occurred in the 1950s and 1960s, may have caused changes in nutrient dynamics and plankton community structure (Carpenter *et al.*, 1986; Witte *et al.*, 1992a; Ochumba, 1995). As the introduction of Nile perch eliminated the major part of haplocromine population, this resulted in the decline of grazing pressure on algae (Ochumba, 1995). In addition, the introduced Nile tilapia, out-competing the endemic tilapiine species, contributed to switch grazing pressure from phytoplankton to benthic organisms (Gophen *et al.*, 1992), and so further altered the water quality and the food chain connections in the lake.

### *1.3.1.4. Other environmental factors*

The dramatic changes in the ecosystem during the 1960s were coincident with both the introduction of Nile perch and exotic tilapiines and record high water levels on Lake Victoria (Welcomme, 1970). Between 1961 and 1964 exceptional rains fell in East Africa. The most remarkable effect on Lake Victoria was to raise its level by about 2.5 m during those years, doubling the output of the Nile outlet over its mean annual discharge (Kite, 1981). Surface area also increased from 68,800 $km^2$ to about 75,000 $km^2$ (Beadle, 1981). Due to a flushing time in excess of 100 years (Hecky & Bugenyi, 1992), after 30 years the level of Lake Victoria has yet to return to its pre-1960 level (Hecky *et al.*, 1994). Moreover, years of heavy rains, often associated with "El Niño" events, increased the lake level by 1.5 m during 1977-1980 (Kite, 1981) and resulted in unprecedented flooding of rivers and apparent rise in lake level during 1997-1998 (Lung'ayia *et al.*, 2001). During these events almost all the rooted vegetation was permanently submerged and eventually rotted away. Floating vegetation, such as papyrus swamps, were either washed ashore or disintegrated through wave action. Land vegetation in newly submerged areas also died. As result of the inundation of land which occurred during such periods, large loads of sediments and nutrients where pumped into the lake. Indeed, in another African lake, Lake Kariba, it has been shown that lake level fluctuations and the resultant nutrient input are important in promoting fish production (Karenge & Kolding, 1995). Also Fryer & Iles, (1972) consider that changes in water level have important implications for fish speciation and possibly fish production. Moreover, in Lake Victoria these water level perturbations could have modified near-shore habitats and changed breeding area available to many fish species (Lung'ayia *et al.*, 2001).

Another recent factor to be considered is the slight change in mean water temperature of the lake. Hypolimnion water temperature as low as 23.5 °C, which had been observed in the early limnological studies (Worthington, 1930; Fish, 1957, Talling, 1966), has not been seen during the recent studies in Lake Victoria (Ochumba, 1995). These data suggest a possible warming trend of climate in East Africa (Hastenrath & Kruss, 1992). As consequence, the thermal structure changed as indicated by shorter turnover and shallower depth of the thermocline (Ochumba *et al.*, 1994; Gophen *et al.*, 1993). These recent changes in climate have altered water column structure and mixing patterns, and may have favoured the development of blue-green algae and the loss of oxygen in the bottom waters of Lake Victoria (Kling, 1992; Hecky *et al.*, 1994).

In addition, during the last decades, Lake Victoria waters have been severely affected by the invasion and proliferation of water hyacinth *Eichhornia crassipes*. This waterweed, originary from South America, recently invaded the lake starting in 1989 (Twongo, 1992). It spread quickly in many parts of the lake, particularly in bays and inshore shallow waters, smothering nursery and foraging grounds for most fish species, particularly for those which are high oxygen-demanding. Its negative impact also results in the reduction of fish yields by the physical obstruction of access to fish landing and fishing sites along the shore (Twongo, 1991). Moreover, physical cover by water hyacinth mats reduces penetration of light going into the water and reduces mixing of the water column, thereby affecting nutrient dynamics (Lung'ayia *et al.*, 2001). Nutrients dynamics may also be altered by the fact that its extremely high growth rate makes water hyacinth a very important nutrient sink, thereby influencing lake and fishery productivity (Thompson, 1991; Taylor, 1991). On the other hand, its roots may provide cover from the predators for juvenile fish (Ogutu-Ohwayo, 1995) and support a large and diverse community of invertebrates (Willoughby *et al.*, 1993).

### 1.3.2. Declining trends for agricultural productivity in Lake Victoria basin

The point has already been made that in the Lake Victoria basin and in sub-Saharan Africa as a whole there is growing concern over the declining agricultural food production (Nandwa & Bekunda, 1998). Generally, nitrogen (N) and phosphorus (P) are the most limiting nutrients in tropical soils. In sub-Saharan Africa, farm households, developers and policy makers increasingly recognise soil nutrient depletion as one of the major constraints to sustainable agricultural and rural development (Smaling, 1993; Smaling *et al.*, 1996). A quantitative nutrient balance model for calculating inputs and outputs of nutrients in African land use systems was developed and applied on supra-national scale (Stoorvogel *et al.*, 1993). Calculating inputs and outputs led to the conclusion that there are considerable net fertility losses in each growing period. During 1982-84, average nutrient losses for arable lands in sub-Saharan Africa estimated by Stoorvogel *et al.* (1993) were 22 kg N and 2.5 kg P per hectare per year. These figures were predicted to become even worse (Table 1.9).

**Table 1.9:** Average nutrient balances of nitrogen (N) and phosphorus (P) for the arable land of some sub-Saharan countries.

| **Country** | **N (kg ha$^{-1}$ yr$^{-1}$)** | | **P (kg ha$^{-1}$ yr$^{-1}$)** | |
|---|---|---|---|---|
| | **1982/84** | **2000** | **1982/84** | **2000** |
| Benin | -14 | -16 | -1 | -2 |
| Botswana | 0 | -2 | 1 | 0 |
| Cameroon | -20 | -21 | -2 | -2 |
| Ethiopia | -41 | -47 | -6 | -7 |
| Ghana | -30 | -35 | -3 | -4 |
| Kenya | -42 | -46 | -3 | -1 |
| Malawi | -68 | -67 | -10 | -10 |
| Mali | -8 | -11 | -1 | -2 |
| Nigeria | -34 | -37 | -4 | -4 |
| Rwanda | -54 | -60 | -9 | -11 |
| Senegal | -12 | -16 | -2 | -2 |
| Tanzania | -27 | -32 | -4 | -5 |
| Zimbabwe | -31 | -27 | -2 | 2 |

***Source***: Stoorvogel *et al.* (1993).

In Kenya, national agricultural production is characterized by a negative nutrient balance, with estimated average annual losses of 40-42 kg N ha$^{-1}$ and 3 kg P ha$^{-1}$ respectively (Stoorvogel & Smaling, 1990; Stoorvogel *et al.*, 1993), which are predicted to remain negative at 46 kg N ha$^{-1}$ and 1 kg P ha$^{-1}$ in year 2000 (Stoorvogel *et al.*, 1993). These negative values are confirmed by Shepherd *et al.* (1995, 1996), who estimated a annual loss of 76 kg N ha$^{-1}$ and 3.8 kg P ha$^{-1}$ for crop land in western Kenya. Similar trends are also reported by Onyango (1998). These estimates, coupled with a downward trend in food production per capita over the last few years (FAO, 1996), can be classified as unsustainable. At district scale, aggregated nutrient balances for the major agricultural systems in the Kisii District showed more alarming predictions with negative annual balances of 112 kg N ha$^{-1}$ and 3 kg P ha$^{-1}$ (Smaling *et al.*, 1993). This indicates a fast rate of soil nutrient mining in agricultural systems in a well-endowed and densely populated part of the Kenyan highlands. At farm level, a recent study by Van den Bosch *et al.* (1998), analysing nutrient balances for 26 farms in three different districts in Kenya, estimated a negative mean balance of 71 kg ha$^{-1}$ yr$^{-1}$ for nitrogen and a positive balance of 3 kg ha$^{-1}$ yr$^{-1}$ for phosphorus. Farmer's perception is also confirming the estimations by scientists. In participatory rural appraisals carried out in the western Kenyan districts of Kakamega (Ogaro *et al.*, 1994), Kisii (Maobe *et al.*, 1994) and Kilifi (Nandwa *et al.*, 1994) farmers indicated that soil fertility decline was a major concern to them.

Similarly to the current situation in Kenya, negative nutrient balances may be expected also for the case of Uganda, where fertilizers use is negligible (see Table 1.11).

In fact, soil fertility mining is a major problem in Uganda as only 2% of farmers apply inorganic fertilizer and the gross average rate of fertilizer application is about 1 kg ha$^{-1}$, which is among the lowest in the world (NARO & FAO, 1999). Consequently, it is estimated that nutrients in Uganda are depleted at an annual average rate of 38 kg N ha$^{-1}$ and 17 kg P ha$^{-1}$ (IFDC, 1999). These negative estimates are higher than those of 32 kg N ha$^{-1}$ and 5 kg P ha$^{-1}$ presented in an earlier study for Uganda by Stoorvogel & Smaling (1990). This reflects a further decrease in soil fertility during the last few years. Indeed, according to a recent study by IFPRI (2002), agricultural development in Uganda has been slow or even negative in some periods since 1970, causing widespread rural poverty and environmental degradation. Increases in crop production have occurred mainly as a result of area expansion, in particular in the 1980s and 1990s. Crop yields have been almost stagnant or declining (Table 1.10). In value terms, land and labour productivity have grown slowly or even decreased in the same period. The slow growth in agricultural productivity since around 1980 is especially disturbing as it follows a decade of negative growth in the 1970s, while in the 1960s growth rates were positive and relatively high for all productivity indicators (IFPRI, 2002). Declining agricultural productivity is an evidence of soil fertility depletion which is supported by findings from researches on soil nutrient balances in several sites in eastern Uganda (Kaizzi *et al.*, 2002; Nkonya & Kaizzi, 2002) as well as soil analysis conducted in sites that were studied previously in the 1960s (Ssali, 2002). Evidences of declining soil fertility are also cited by farmers throughout Uganda (Pender *et al.*, 2002; Nkonya *et al.*, 2002; Sserunkuuma, 2002).

**Table 1.10:** Indicators of agricultural development in Uganda 1961-1998.

| | *Annual average growth (%)* | | | |
|---|---|---|---|---|
| | 1961-70 | 1971-80 | 1981-90 | 1991-98 |
| Population | - | 3.2 | 2.4 | 2.9 |
| Harvested area | 2.9 | -3.7 | 2.2 | 2.0 |
| Average crop yields (weight/area) | 3.2 | -3.1 | 1.0 | -0.8 |
| Economic productivity of land (value/area) | 3.8 | -3.1 | 1.4 | 1.4 |
| Labour productivity (value/agric. worker) | 1.9 | -4.4 | 1.0 | -0.3 |

***Source***: IFPRI (2002).

#### *1.3.2.1. The problem of soil erosion*

The concept of quantitative nutrient balance used in sub-Saharian Africa for assessing soil nutrient depletion (Stoorvogel & Smaling, 1990; Stoorvogel *et al.*, 1993) is based on five inputs (mineral fertilizer, organic manure, atmospheric deposition, biological nitrogen fixation, and sedimentation) and five outputs (harvested products, crop residues, leaching, gaseous losses, and soil erosion). As an element in explaining the negative trend of nutrient balances and agricultural productivity in the region, soil erosion ranks second after losses through harvested material, particularly for phosphorus and potassium (Schnier, 1993). This is in line with findings by other authors (Lal, 1985; Kilewe, 1987), who reported that organic matter nutrient element loss due to erosion is one of the major causes of fertility depletion in tropical soils.

Soil is a form of capital that is an essential input to agricultural production (Ekbom, 1995). However, agricultural land use often results in degradation of natural soil fertility through nutrient depletion and erosion. Around Lake Victoria soil erosion is particularly high. A study by Lufafa *et al.* (2002) applied to a micro catchment of Lake Victoria in Uganda, predicted an annual soil loss of 93 t $ha^{-1}$ in areas under annual cropland use, 52 t $ha^{-1}$ in rangelands, 47 t $ha^{-1}$ in banana-coffee intercrops, 32 t $ha^{-1}$ in banana sole crop systems, and almost no erosion at all in areas covered by forests or wetlands. Other authors report soil losses up to 90 t $ha^{-1}$ $yr^{-1}$ in Kenya's semi-arid areas (More *et al.*, 1979; Thomas *et al.*, 1981). In the central highlands of Kenya, Gachene (1995) reports soil loss up to 247 t $ha^{-1}$ $yr^{-1}$ for non-conserved maize plots on steep slopes, while Schneider (1993) estimates 93 t $ha^{-1}$ $yr^{-1}$ for inter-cropped maize and beans on 60% slope.

Soil erosion not only degrades the land which loses the soil – on-site productivity losses, but also causes problems far away from the source – off-site environmental problems. The soil, chemicals and pathogens it contains, are transported into waterways and onto other lands, inflicting off-site effects through the processes of sedimentation, eutrophication and leaching. The Ministry of Water Development (1992) reports, for instance, up to 19.9 million tonnes per year of suspended sediment loads to the major rivers of the Kenya's Lake Victoria basin. Soil erosion is therefore an economic problem in Lake Victoria because farmers and other land users ignore all or some of the user costs of the soil and off-site costs of erosion (Odendo & Wasike, 1999).

### 1.3.3. The importance of wetlands in Lake Victoria

As land-water interface, wetlands are a crucial element in studying a lake in the context of its watershed, especially for their buffering function between land and lake systems. This is particularly true in tropical and subtropical climates as in the African continent. Although much of Africa is arid, it also supports some of the worlds largest areas of freshwater wetlands (Beadle, 1981; Huges & Huges, 1992). The total area of these swamps is difficult to assess because of their seasonality. Nevertheless, the total extent of permanent swamps in Africa has been estimated at approximately 85,000 km$^2$, much of which is dominated by papyrus vegetation (Beadle & Lind, 1960; Beadle, 1981; Thompson & Hamilton, 1984). The most extensive papyrus swamps are found around the perimeter of Lake Victoria and in Uganda and Sudan associated with the River Nile basin (Beadle, 1981). According to Beadle & Lind (1960), in Uganda alone there were 6,475 km$^2$ of permanent swamp, covering approximately 2.7% of the country. However, these estimates have recently been revised thanks to new studies and the application of remote sensing. The area under permanent wetlands is currently estimated at 7,296 km$^2$ and seasonal wetlands at 22,809 km$^2$, for a total wetland area of 30,105 km$^2$ (UBOS, 1999). These data, together with new estimates of the original wetland extension, show a reduction of 7.32% from the total original wetland area of the country (UBOS, 1999). In Kenya and Tanzania, lake, river and valley swamps are also common although the total area is not known (Huges & Huges, 1992).

Wetlands are a prominent feature of the marginal zones of Lake Victoria (Carter, 1955). These wetlands often contain vegetation including not only papyrus (*Cyperus papyrus*), but also reeds (*Phragmites*) bulrush (*Typha*), hippo grass (*Vossia*) and *Miscanthidium* (Balirwa, 1995). Although their consumption levels are currently low, there are 22 species of edible wetland plants and 35 species of medicinal value (NEMA, 1999). Low impact exploitation of wetland vegetation was traditionally an important source of raw material and income to the local people. Papyrus is still harvested for thatching houses and making mats, baskets, furniture, fishing floats, rafts, etc. (Kairu, 2001; Gichuki *et al.*, 2001; NWCMP, 1996). Shrubs and papyrus growing in wetlands are used as wood fuel, particularly by the artisanal fish-smoking industry (NWCMP, 1995). In fact, in recent years, rapid disappearance of fuel-woods in many developing countries and particularly around Lake Victoria has stimulated investigation into the use of papyrus as an alternative source of fuel (Jones, 1983).

Wetlands are also particularly important in supporting fisheries, both in freshwater and marine ecosystems. Welcomme (1972) estimated that wetland and floodplain related catches contribute to about 40% of the total freshwater catch of the African continent. In Lake Victoria it is recognized that large scale clearance and modification of lake shore wetlands is partly responsible for the reduction and disappearance of major fish species such as *Labeo victorianus*, *Clarias gariepinus*, *Barbus*, and several *Mormyrids* (NEMA, 1999). Indeed, wetlands are an important habitat for many indigenous fishes, in particular for spawning and breeding activities. Some air-breathing fishes such as *Clarias liocephalus*, *Ctenopoma muriei*, and *Nothobranchius sp.*, are found principally in papyrus swamps and other wetlands (Chapman *et al.*, 1998). Other clariid catfishes as well as the lungfish *Protopterus aethiopicus*, which makes its breeding nest within the mud in papyrus wetlands, can exploit the anoxic waters of the dense interior of many swamps with their air-breathing capabilities. Also the small cyprinid *Barbus neumayeri* inhabits the dense interior of papyrus swamps (Chapman & Liem, 1995). The inshore waters of Lake Victoria and other East African lakes have also been described as the typical habitat for tilapia (Graham, 1929; Lowe-McConnel, 1958, 1975; Welcomme, 1965; Kudhongania & Cordone, 1974) and in particular for tilapia larvae (Mnaya & Wolanski, 2002). Indeed, in Lake Victoria tilapia larvae were found in abundance only in wetlands and in adjoining waters and no fish larvae were found at more than 150 m from the shore (Mnaya & Wolanski, 2002). This suggests that shoreline wetlands are used as nursery grounds as shown for Lake Victoria's shrimps *Cardinia nilotica* (Branstrator & Mwebaza-Ndawula, 1998) and for East African chiclids (Chapman & Liem, 1995).

In addition to their habitat functions, several lines of evidence now suggest that wetlands may protect some fishes from Nile perch predation by providing low oxygen refugia and serving as barriers to the dispersal of Nile perch. Wetlands, especially if dominated by papyrus, may protect fishes from predation because they provide structural complexity, reducing hunting efficiency. This particular function exerted by wetlands was explored by Chapman *et al.* (1996), who found a positive relationship between species richness among wetland ecotones and structural complexity in Lake Nabugabo, a small satellite lake of Lake Victoria.

Other studies have shown how swamps, providing low-oxygen refugia, may inhibit dispersal and serve as isolating mechanism. The waters of heavily vegetated swamps are often characterized by reduced oxygen levels due to low levels of incident light, minimal mixing of the water beneath the vegetation, and high rates of organic decomposition

(Carter, 1955; Chapman & Liem, 1995). Carter (1955) reported oxygen values averaging less than 0.1 mg $l^{-1}$ for the nearshore areas of littoral papyrus swamps in Lake Victoria, and average values of 2.5 mg $l^{-1}$ for the interface, the wetland ecotone, between papyrus and pelagic waters. Chapman *et al.* (1998) found mean oxygen levels around 1.2 mg $l^{-1}$ in a papyrus swamp in Uganda. In general, most wetland ecotones exhibit some degree of hypoxia, with mean dissolved oxygen around 3.8 mg $l^{-1}$ or below (Chapman *et al.*, 1996).

The low-oxygen refugia function provided by wetlands is possible because fishes vary in their tolerance to deoxygenated water. Chapman et al. (1995), for instance, found that some of the cichlids from Lake Victoria can tolerate extremely low levels of oxygen and that lacustrine cichlids endemic to Lake Victoria are more tolerant to hypoxia than ecologically similar species from Lake Tanganyika. It is also well known that some species, such as lungfish and catfishes, can live in extreme anoxic conditions thanks to their air-breathing capabilities. Chapman *et al.* (1996) estimate a mean tolerance level of water deoxygenation for some Lake Victoria fish species that varies from 2.51-2.68 mg $l^{-1}$ for clariid catfishes, to 2.77-2.99 mg $l^{-1}$ for some haplocromines, and 3.84 mg $l^{-1}$ for juvenile Nile perch. This would confirm Fish's (1956) findings that extensive wetlands may serve as barriers to the dispersal of Nile perch because *Lates* require water with relatively high dissolved oxygen. The relative unsuitability of low-oxygenated wetlands ecotones to serve as habitat to *Lates* is also confirmed by Schofield & Chapman (1999), who found that Nile perch was 3.7 times more abundant in offshore areas than in inshore areas near wetland ecotones. Similar results were also presented by Chapman *et al.* (1996) who found i) a low abundance of Nile perch in wetland ecotones relative to exposed inshore areas; ii) a positive relationship between Nile perch abundance and dissolved oxygen; and iii) a negative relationship between species richness among ecotones and dissolved oxygen.

Another important service provided by wetlands is their support in improving water quality. This contributes not only to the provision of drinking water to local population, but indirectly it also supports the fishing industry. The function of wetlands in reducing inflow of nutrients, namely nitrogen and phosphorus, from the basin into the Lake Victoria waters has recently been investigated by local scientists. For the Nakivubo swamp in Uganda, Kansiime & Nalubega (1999) estimated nutrient removal within the range of 55-67% and 33-65% for the loadings of total nitrogen and total phosphorus respectively. On average, they found that of the nearly 890 t N $yr^{-1}$ and 75 t P $yr^{-1}$ that enter the swamp system from external sources, 384 t N $yr^{-1}$ and 45 t P $yr^{-1}$ leave the system into the lake.

This nutrient buffering function is higher than that predicted from a previous study for the same wetland which reported a retention capacity of 43% and 22% for total nitrogen and phosphorus respectively (COWI/VKI, 1998). Studies in other regions around the world report removals within 16-75% for nitrogen and 12-73% for phosphorus (Reddy & De Busk, 1987), and 74-77% for both nitrogen and phosphorus (Alaerts *et al.*, 1996).

In studying the potential of wetlands as nutrient sinks, the role of their macrophytes vegetation must be taken into account. Nutrient uptake and excretion/recycling by wetland vegetation play an important role in the overall net nutrient retention capacity of swamps. However, the only effective means of nutrient removal from wetlands via macrophytes is harvesting. Otherwise, most of the nutrients taken up by plants would be recycled back into the system after the growth cycle if no harvesting takes place. On average, in Lake Victoria swamps, 0.079 g N $m^{-2}$ $day^{-1}$ and 0.03 g P $m^{-2}$ $day^{-1}$, or 28% of nitrogen and 60% of phosphorus uptake by plants is recycled into the system through leaching (Kansiime & Nalubega, 1999). For Lake Naivasha in Kenya, Gaudet & Muthuri (1981) reported similar values of 0.087 g N $m^{-2}$ $day^{-1}$ and 0.041 g P $m^{-2}$ $day^{-1}$ for leaching and decomposition of papyrus tissues in swamps. In Nakivubo swamp, sustainable harvesting of papyrus would permanently remove every year a further 5.33% of phosphorus and 3.54% of nitrogen loadings into the swamp (Kansiime & Nalubega, 1999).

#### *1.3.3.1. Current threats to the Lake Victoria wetlands*

Although the importance of wetlands in providing goods and environmental services to local population is generally recognized, most of the swamps around Lake Victoria are still being reclaimed and degraded. This process is driven by the high growth of the local human population and the consequent increasing demand for settlements, food and grazing lands. Past governments interventions and the common property condition of wetlands resources have also been among the key factors in promoting swamps reclamation.

In Uganda, until recently the official government policy encouraged the drainage of wetlands. This policy has changed in the last few years and swamps are now protected by the National Environment Statute of 1995 (NEMA, 2001). Many swamps around urban areas were drained in the 1950s, in part as a malaria-eradication measure (MISR, 1998). Large-scale drainage for agricultural and industrial activities was realized in the 1970s (NEMA, 2001). Currently, an estimated 2,376 $km^2$ of wetland area have been reclaimed in Uganda, which represents a 99.3% increase in area reclaimed since the 1960s (rate of about 2.9% per annum). That is, by 1964, only about 0.05% of the original area under wetlands had been reclaimed compared to 7.9% by the late 1990s (NEMA, 2001). At present, pressure on wetlands is still high because the government does not have the capability to enforce wetlands protection. The anthropogenic pressure varies from district to district. Among the Ugandan districts along the shores of Lake Victoria the most degraded wetlands are in the northern part. The district of Kampala has already lost 19.7% of its original 41 km$^2$ of wetland area, Jinja 43.2% of 176 km$^2$, Mukono 10% of 1,096 km$^2$, Tororo/Busia 32.2% of 1,160 km$^2$, and Iganga/Bugiri 32.7% of 1,806 km$^2$ (UBOS, 1999). The situation is much better in the south-western part where the district of Mpigi has lost only 1.4% of its original 1,068 km$^2$ of wetland area, Rakai 3.3% of 1,322 km$^2$, and Masaka/Sembabule 0.8% of 1,436 km$^2$ (UBOS, 1999).

Also in Kenya natural wetlands are threatened by human development, although statistical data are not readily available. Along the shores of the Kenyan part of Lake Victoria there are many seasonal and permanent swamps. The two largest swamps are Nyando and Yala. The latter has been heavily affected by human development. Much of the area has been demarcated for reclamation since the mid 1960s. An extensive reclamation project by the Kenyan government took place in two different phases. After 2,300 ha of the original 17,500 ha were drained in phase I, further 9,200 ha were to be reclaimed for agriculture in phase II, following the directions of the Lake Basin Development Authority Five Years Plan 1989-1993 (Riedmiller, 1994). Even though it has been planned that a portion of wetland would be preserved, the ecological and economic consequences of the reclamation project are not known.

## 1.4. Problem statement, scope and layout of the thesis

Watershed degradation in many countries threatens the livelihood of millions of people and constrains the ability of society to develop a healthy agricultural and natural resource base. This also seems to be the case for the Lake Victoria basin, where a growing population is facing decreasing trends of food per capita availability. Watershed degradation is thought to be a major cause in the recent drop of agricultural productivity and fisheries yield in the region.

Humans are currently a dominant component of the Lake Victoria food web. The introduction of new species, cultural eutrophication and fishery exploitation have the potential to alter community structure and dynamics within the lake ecosystem. Over the last century the food web of the lake has been radically altered, partly due to over-exploitation of the fisheries and introduction of the piscivorous Nile perch and four tilapiine species. There is also evidence that the nutrient chemistry of Lake Victoria has changed due to cultural eutrophication. Changes in the nutrient loading and/or alterations in the food-web structure can change phytoplankton productivity and phytoplankton species composition through either bottom-up or top-down effects. Recent statistics show that these processes ultimately impact the fisheries of Lake Victoria.

On the other hand, the agricultural sector in the region is also confronting trends of decreasing productivity. Soils deficient in the major nutrients, nitrogen and phosphorus, have been identified as the major problem affecting crop productivity in much of sub-Saharan Africa (Mokwunye *et al.*, 1996; Sanchez *et al.*, 1997; Smaling *et al.*, 1997) due to a long-term process of soil fertility loss. This is often imputed to the negligible application rates of mineral fertilizers (Table 1.11) in the three basin countries. The average rates of fertilizer application in sub-Saharan Africa range between 1-10 kg $ha^{-1}$ (Simpson *et al.*, 1996), whilst FAO (1993) estimates that applications of 90-130 kg $ha^{-1}$ are required for food sufficiency to be achieved. Short-term effects of soil nutrient depletion may be reversed by corrective measures such as fertilizers or manure applications (Francis, 1990). However, in the long-run this would require high rates of soil amendments which would be costly for many developing countries (Forster *et al.*, 1985; Kefeni, 1992).

**Table 1.11:** Total fertilizer ($N + P_2O_5 + K_2O$) consumption trends in some African countries ('000 metric tonnes).

| Country | 1969/70 | 1971/72 | 1979/80 | 1985/86 | 1990/91 | 1995/96 |
|---|---|---|---|---|---|---|
| Uganda | 5 | 8 | - | - | - | 3 |
| Kenya | 42 | 47 | 51 | 109 | 116 | 75 |
| Tanzania | 11 | 17 | 30 | 39 | 51 | 22 |
| Zimbabwe | 94 | 133 | 120 | 170 | 177 | 164 |
| Malawi | 7 | 15 | 21 | 34 | 48 | 34 |
| Ghana | 1 | 3 | 19 | 13 | 13 | 12 |
| Cote d'Ivore | 14 | 24 | 52 | 42 | 36 | 65 |
| Egypt | 348 | 372 | 604 | 864 | 965 | 1134 |
| South Africa | 509 | 617 | 941 | 879 | 792 | 748 |

***Source*:** FAO Fertilizer Yearbook, several years.

There is also a strong link between soil fertility and soil erosion. Plant nutrients are relatively more abundant in the top soil and associated with finer soil fractions that are more subject to erosion (Odendo & Wasike, 1999). In addition, erosion reduces moisture holding capacity of the soil, restricts rooting depth, and makes fertiliser application less efficient by increasing nutrient losses (Lake & Shady, 1993; Thomas, 1997). Thus, technologies that add new nutrients to agro-ecosystems are not sufficient. They need to be supplemented by actions that prevent nutrients run-off. Soil conservation, in the form of appropriate land use and management, is then a major issue for sustainable agricultural development in the Lake Victoria basin.

This is particularly true considering the external effect exerted by soil erosion on nutrients dynamics in Lake Victoria. Estimates of nutrient loading into the lake have been taken using pollution factors for the quantification of domestic, agricultural and industrial loads from Tanzania (Scheren, 1995), Kenya (Kirugara & Nevejan, 1996), and the entire catchment area (Zanting, 1996). These studies conclude that the largest fraction of organic matter stems from domestic and industrial waste, whereas the nutrient loading is caused by agriculture and to a larger extent atmospheric deposition.

In agricultural soil nutrient balances soil erosion ranks second only after losses through harvested material (Schnier, 1993), and atmospheric deposition is a direct consequence of soil erosion. For this reason, it can be inferred that nutrients loading into Lake Victoria is mainly due to this physical process. Then, the direct link between soil erosion, land use and fisheries via eutrophication is quite obvious for Lake Victoria.

Sustained population growth and increasing population densities will further increase land degradation with negative effect for both agriculture and fishery productivity. Government plans following experts' recommendations (Baligar & Bennett, 1986; Vlek, 1990) of promoting higher fertilizer applications may even worsen the eutrophication problem if not accompanied by appropriate land use management. In this framework the role of wetlands is crucial, as it seems difficult to implement policy measures influencing land use in the short-term. Laying in the middle between the land and lake systems, wetlands are able to buffer the external effect on fisheries from an agricultural sector which needs to develop quickly in order to meet the increasing food demand.

In the long-run, however, an optimal land use accounting for both user costs of the soil and off-site costs of erosion, together with appropriate fishery management and wetlands conservation measures, may offer a solution to the reducing trend in per capita food production for both agriculture and fishery sectors at the same time. As it concerns the future of millions of people, the problem of long-term food security in the Lake Victoria basin needs to be addressed. Nevertheless, among many studies concerning the declining fishery yields in Lake Victoria or the agricultural nutrient depletion in its catchment, there are no approaches able to link the two systems under an economic optimisation problem. This is of course a complex task as it requires a deep understanding of all the physical, chemical, biological, ecological and socio-economic interactions among three different systems: land, wetland, and lake.

This thesis addresses this complex problem by developing a multidisciplinary and multisectoral approach in order to identify the appropriate policy solutions for the entire watershed. The thesis is organized in two parts. The first part concerns theory and applications issues which introduce the model and its three different components: land, wetland, and lake. Chapter 2 gives an overview of the theories that provide a framework for the land use component of the model. Chapter 3 introduces the wetland component reviewing the literature of empirical models of nutrient retention by wetlands. The theoretical background for the lake component of the model is provided in Chapter 4. This chapter treats issues such as limnology, plankton ecology and bioeconomic fishery models.

The second part of the thesis applies the theoretical framework to the Kenyan part of Lake Victoria. In Chapter 5, the Ecopath approach is used to explain the role of eutrophication and overfishing on the freshwater ecosystem dynamics during the last decades. The trend in phytoplankton biomass estimated in Chapter 5 is then used as environmental variable for the parameterization of a modified Gordon-Schaefer fishery model in Chapter 6. Chapter 7 is showing an example of watershed approach to fishery management embedding the land use and wetland components into the fishery model. Global and sectoral social decision problems are shown. Policy implications and scenarios are finally considered. Chapter 8 presents the final conclusions and identifies potential future developments and implications of this research.

# PART I:

# *Theory and Methodology*

# Introduction to Part I

Lakes are integrated parts of their basins. This emphasizes the need for modelling the entire watershed in relation to the complex ecosystem of the lake. Our approach thus includes the assessment of several points following a "black-box" model (Figure I.a) in which the final objective is to estimate the economic impact of land use change, through its onsite and offsite externalities. The approach considers the following points:

i) The socio-economic dynamics in the catchment area of a given lake which influence nutrient loading, runoff, and fishing effort;

ii) The role of buffering ecosystems such as wetlands, which reduce nutrient inflow into the lake due to their retention function;

iii) A description of the of lake's nutrients dynamics in terms of limnological mass flow and balance, i.e. how much matter enters, leaves and remains in circulation in the water;

iv) A description of the dynamics of soil quality in terms of soil nutrient balance assessment;

v) The relationship between nutrient concentration in the water and the trophic productivity level of the lake, eventually affecting fish biomass and fisheries;

vi) The relationship between soil quality and agricultural productivity.

This kind of approach consists of three structural elements: abiotic, biotic, and anthropogenic. The anthropogenic element is represented by the socio-economic factors driving the model. In attempting to improve their well-being, individuals are facing choices in the use of the resources available to them. However, their behaviour may not be rational if they fail to internalise the external effect of their actions due to lack of information. Their suboptimal choices may be a threat for long-term sustainable development. In our model these choices are represented by private decisions on agricultural land use and fishing effort. These private decisions, however, can be modified and corrected by government intervention – the institutional factor – to obtain the maximum level of welfare for the society as a whole, including future generations as well.

The biotic component of the model is represented by the freshwater ecosystem of the lake and the agroecosystem of its basin. The biotic elements in the lake's basin are the harvested crops, while lake ecosystem includes several species interacting among them. Primary producers such as phytoplankton (algae and diatoms) and higher plants in wetlands (macrophytes) produce biomass from their uptake of nutrients. Within the lake ecosystem, phytoplankton represents the base of the trophic pyramid. Zooplankton, together with planktivorous, detritivorous, and piscivorous fish species, is regulating and being regulated by primary producers through a complex structure of food web dynamics.

The abiotic components are the chemical elements of the model, such as nutrients. Within this category we are considering the dynamics of nitrogen (N) and phosphorus (P) as they are widely significant for the biological productivity of lakes and streams, not to mention their importance as input for the agricultural production in the basin. The amount of phosphorus available is commonly the limiting factor for freshwater primary productivity due to its small natural supply. However, in tropical climates nitrogen is often found to be limiting. Generally, a distinction between the two cases of nutrient limitation can be made on the basis of the respective N:P ratio, which implies that the nutrient controlling the maximum amount of plant biomass is the nutrient with the lowest concentration relative to the specific biotic uptake.

Nitrogen is composed of various forms, such as particulate organic nitrogen, soluble organic nitrogen (SON), and the three forms of inorganic nitrogen: ammonium ($NH_4$-N), nitrite ($NO_2$-N) and nitrate ($NO_3$-N). Nitrogen dynamics are given by chemical transformation through the processes of mineralization-ammonification, volatilisation, nitrification and denitrification which are happening either in the soil or in the water. Nitrogen mineralization refers to a series of biological transformations that converts organically bound nitrogen to ammonium nitrogen ($NH_4$-N), as the organic matter is being decomposed and degraded. This pathway occurs under both anaerobic and aerobic conditions and is often referred to as ammonification. In a lake, ammonium is produced by ammonification of detritus at the bottom. Once the ammonium ion ($NH_4^+$) is formed, it can take several possible pathways. It can be absorbed by algae, plants or anaerobic microorganisms and converted back to organic matter. Otherwise, under high pH conditions ($pH > 8$), the ammonium ion can be converted to $NH_3$ which is then released to the atmosphere through volatilisation. In the soil, another possible pathway is that the ammonium ion, which is positively charged, can also be immobilized through ion exchange onto negatively charged soil particles. Finally, in aerobic environments, ammonium nitrogen can be oxidized through the process of nitrification in two steps. First by microbacteria *Nitrosomonas sp.*:

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 2H_2O + 4H^+ + \text{energy}$$

and then by *Nitrobacter sp.*:

$2NO_2^- + O_2 \rightarrow 2NO_3^- + \text{energy}$

In lakes, although nitrification can happen under aerobic conditions in the water column as well, it is reported to occur predominantly in the sediments, where ammonium concentrations are normally high.

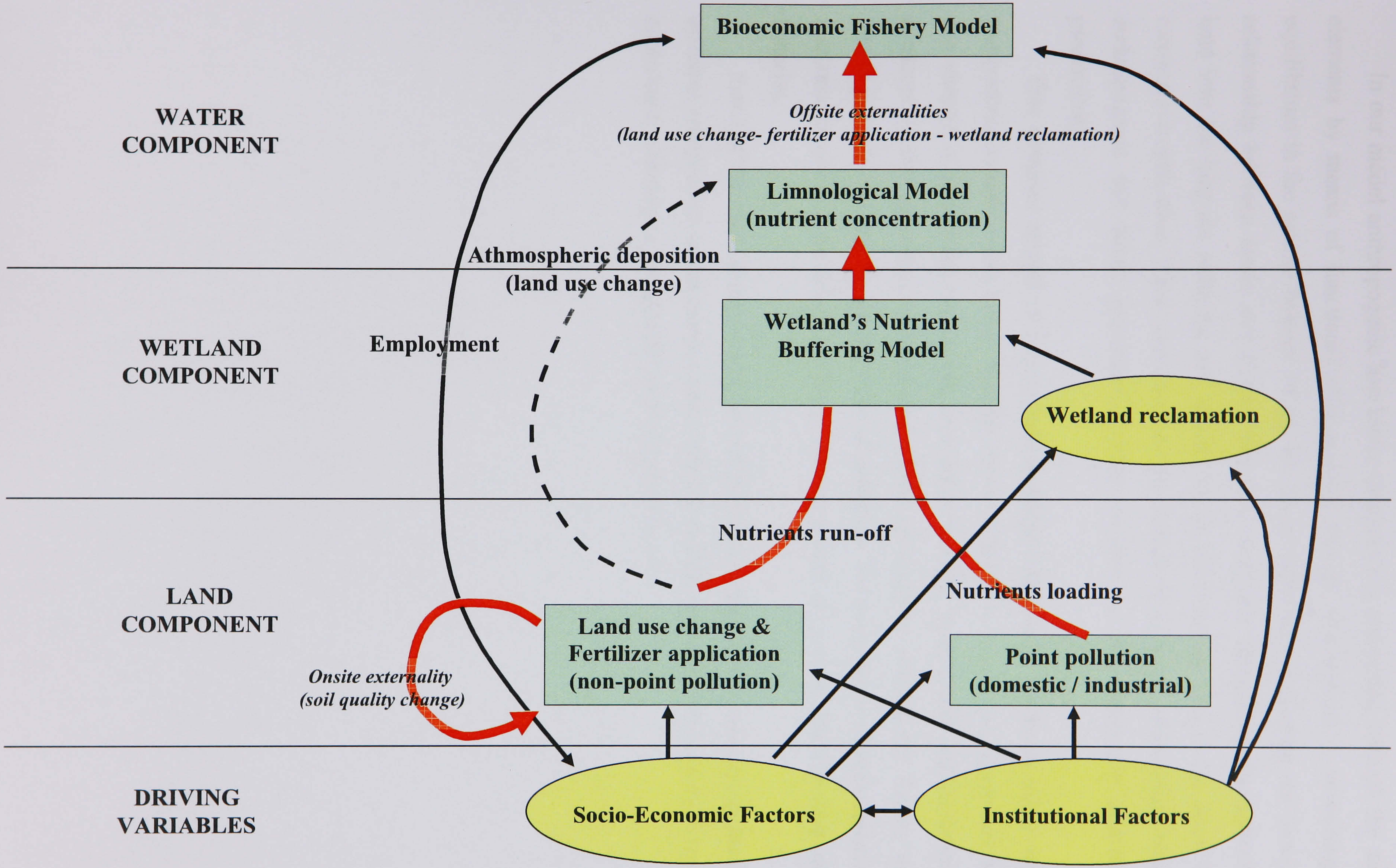
Soil nitrate ($NO_3$), which has a negative ion rather than the positive ammonium ion, is not subject to immobilization by negatively charged soil particles and is, thus, much more mobile in solution and then subject to leaching. If not assimilated immediately by plants or microbes (assimilatory nitrate reduction) or, for the specific case of soil nitrate, it is not lost through groundwater or superficial water flow (leaching), nitrate has the potential to undergo dissimilatory nitrogenous oxide reduction. This term refers to several pathways of nitrate reduction. The most prevalent are reduction to ammonia and denitrification. Denitrification, carried out by microorganisms under anaerobic conditions, results in the loss of nitrogen as it is converted to gaseous nitrous oxide ($N_2O$), which will dissipate into the atmosphere, and to molecular nitrogen ($N_2$) which can not be used as nutrient by most plants and algae:

$C_6H_{12}O_6 + 4NO_3 \rightarrow 6CO_2 + 6H_2O + 2N_2$

Because denitrification itself requires anaerobic conditions, but its substrate (nitrate) is produced under aerobic conditions, denitrification works mainly where both conditions co-occur or alternate in time.

Like nitrogen, also phosphorus occurs in many different forms which can be grouped in particulate and soluble phosphorus. The latter can be further divided in soluble reactive and soluble unreactive forms according to its immediate availability to plants uptake. In lakes however, unlike ammonium and nitrate, phosphate is absorbed by sediment particles and precipitates to insoluble forms. Therefore, in eutrophic lakes there is a strong accumulation of phosphorus in the sediments.

**Figure I.a:** Structure of the model.



In our model anthropogenic and biotic elements are interacting through the abiotic elements by means of functional relationships. Human intervention is perturbing the equilibrium in the natural balance of the abiotic environment, modifying the functional relationship between biotic and abiotic elements. Nutrients flows from the agricultural land into the lake are reducing soil quality for crop harvesting and, at the same time, causing eutrophication. These perturbations are finally affecting the relationship between anthropogenic and biotic elements in terms of reduced fish yield and agricultural productivity.

This complex system of interrelated elements requires an approach using several components as submodels linked through flows of inputs and outputs (Figure I.a). Land use change is generating an on-site externality through agricultural soil depletion of nutrients. It also generates unidirectional offsite externalities towards the fishery sector that is partially reduced by the presence of wetlands. The importance of point sources of nutrients should also be accounted for in order to isolate the partial effect of land use on fisheries.

Part I of the thesis presents a review of theory and applications concerning the three different components of the model (land, wetland, and lake) which introduces to the model application referring to the specific case of Lake Victoria.

# Chapter 2

# THE LAND COMPONENT

## 2.1. Introduction

Environmental assessments show nonpoint sources of soil and water pollution as a major cause of environmental damage in both developed and developing countries (World Resources Institute, 1988; OECD, 1991; Duda, 1993). There are many possible downstream or offsite impacts of anthropogenic land use that result from nutrients runoff and sedimentation. Eutrophication is one of them. In addition, downstream externalities are often accompanied by onsite impacts.

The onsite impacts of nutrient loading are most frequently studied by analysing the effect of soil degradation on crop production. There are many cases in which soil degradation has been measured in terms of soil erosion. Under this perspective several studies show the relevance of the problem particularly for developing countries. Magrath & Arens (1989), for instance, estimated the present-value cost of the onsite annual impact of soil erosion in Java to be US$ 323 million, or just below half of one percent of GDP. Bishop & Allen (1989) estimated the onsite annual cost of soil erosion in Mali at US$ 31-123 million, or 4-16% of agricultural GDP. More recently, Barbier & Bishop (1995) estimated the cost of land degradation in developing countries to vary from 1% to over 15% of their GDP.

Other studies focus on a broader view of the onsite impact of soil degradation. Nutrient balance in topsoil is regarded as a more appropriate measure of soil quality influencing crop productivity. This approach includes soil erosion as one element of soil nutrients dynamics. It seems to be a better measure of soil quality as it accounts for the impact of fertilizer applications. These studies report a dramatic decline in soil productivity in many developing countries, mostly due to unsustainable land use (Stoorvogel & Smaling, 1990; Stoorvogel *et al.*, 1993).

The processes and determinants of soil degradation are key to the sustainability of development patterns in third world countries, and are of particular importance in our study. In this chapter, we analyse the theories and methodologies that provide a framework for studying onsite and offsite impacts of land use and soil degradation. The physical prediction of soil erosion is also considered (§ 2.4), as this process is an important element in estimating both onsite and offsite externalities.

## 2.2. The onsite effects of land use: Soil as production factor in agriculture

Soil is an essential input in agricultural production. Economists often treat soil stock as a semi-renewable resource assuming that soil in agriculture is a potentially depletable resource. This is particularly true when considering soil in terms of depth. If land is left undisturbed, soil accumulates, but it does so at an extremely slow rate. According to Myers (1988), under normal conditions, soil forms at rates of 0.01-0.5 mm per year. However, the rate at which topsoil is eroded through cultivation is faster than the rates at which it regenerates (Barbier, 1998d).

For this reason soil is a key element in estimating onsite impacts of land use. These impacts are most frequently studied through the analysis of the effect of soil nutrient runoff on crop production. Because nutrient runoff affects long-term soil productivity, it has potentially significant implications for economic welfare. An understanding of soil productivity dynamics is useful both for agricultural decision-making, and as an input to ex ante and ex post analyses of the benefits and costs of agricultural policies that alter the value of crops land use or cultivation practices. The task for economists is to identify the links between agricultural production and resource degradation under various property regimes and market conditions, and to indicate the appropriate policies in order to maximise social welfare.

### 2.2.1. Economic models of onsite externalities

Agricultural economists recognize that soil management is a dynamic process that has to be adjusted continually to changes in the stock, i.e. the depth or the average nutrient concentration in the topsoil layer. Indeed, the onsite effect of changes in soil quality is a stock externality. A stock externality exists when decisions made in one time period have cumulative effects over time (Nordhaus, 1982; Brito & Intriligator, 1987; Ko *et al.*, 1992).

The concern over the onsite effects of this stock externality has stimulated studies modelling the optimal private agricultural utilization of the soil. Past research, including studies by Burt (1981), McConnell (1983) and Miranowski (1984), treats soil as a renewable resource and focuses on the consequences of decreasing soil productivity for agricultural production. These early economic models provided a framework for analysing the implications of soil dynamics on the optimal decision problem of agricultural household when offsite externalities are not considered.

Barrett (1997), for instance, modifies the model early developed by McConnell (1983) to analyse the smallholders' response to macroeconomics reforms through the optimal control of soil erosion. It is assumed that farmers influence soil dynamics, in this case represented by a measure of topsoil depth, by choosing which crops to plant, a tillage technique, the extent and nature of crop rotation, and terracing. All these actions are collapsed into a single decision variable (*R*), which can be taken to represent the intensity of cultivation, and which influences soil dynamics (d*Q*/dt) in conjunction with a package of soil conservation practice ($X_2$) according to the following expression:

$$\dot{Q}=q\left(M,R(t),X_2(t)\right)=M-R*g(X_2) \qquad [2.1]$$

with,

$$g(X_2)=1$$
$$g'(X_2)\leq 0$$
$$g''(X_2)\geq 0$$

where *M* is a constant representing naturally occurring additions to topsoil; and *g*(•) is a function of soil conservation inputs $X_2$. This expression says that the more intensive the cultivation, expressed by a greater *R*, the greater is soil loss. In this case a farmer can maintain a constant soil depth provided the use of conservation inputs is increased. Barrett (1997) assumed that farm output, given by the aggregate production function *Y*, depends on soil depth (*Q*), cultivation practice (*R*), and traditional non-soil inputs such as labour, fertilizer and water represented by the vector $X_1$. Then, the problem for the farmer is to choose the optimal level of traditional agricultural inputs, soil-conservation inputs, and cultivation intensity which maximizes the net present value (*PV*):

$$Max_{R,X_1,X_2} PV = \int_{t=0}^{\infty} e^{-\delta t}\left[pY(R(t),Q(t),X_1(t)) - c_1 X_1(t) - c_2 X_2(t)\right]dt \qquad [2.2]$$

subject to:

$$\dot{Q} = q(M,R(t),X_2(t)) = M - R * g(X_2)$$

$$Q(0) = Q_0$$

with,

$$Y = f(R(t),Q(t),X_1(t)) \qquad [2.3]$$

where $c_1$ represents the cost of the conventional crop production inputs; $c_2$ is the cost of the soil conservation input package; and $p$ is the price of the crop output. Thus, given the Hamiltonian ($H$) and the continuous function $\mu(\mathrm{t}) = e^{\delta t}\lambda(\mathrm{t})$, where $\lambda(\mathrm{t})$ is the imputed value of an incremental unit in $Q(\mathrm{t})$ from the perspective of $\mathrm{t} = 0$ and $\delta$ is the rate of discount, the first-order conditions are:

$$\frac{\partial H(\bullet)}{\partial X_1} = p\frac{\partial Y(\bullet)}{\partial X_1} - c_1 = 0 \qquad [2.4]$$

$$\frac{\partial H(\bullet)}{\partial R} = p\frac{\partial Y(\bullet)}{\partial R} - \mu(t)g(X_2) = 0 \qquad [2.5]$$

$$\frac{\partial H(\bullet)}{\partial X_2} = -c_2 - \mu(t)Rg'(X_2) = 0 \qquad [2.6]$$

$$\dot{\mu} = -\frac{\partial H(\bullet)}{\partial Q} + \mu(t)\delta = -p\frac{\partial Y(\bullet)}{\partial Q} + \mu(t)\delta \qquad [2.7]$$

Condition [2.4] says that non-soil inputs should be employed up to the point at which the value of the marginal product equals the input price. Condition [2.5] indicates that the marginal profit of the increase in cultivation intensity ($R$) must at all times equal the shadow value of the soil that is lost as a consequence of that increase. Condition [2.6] says that conservation inputs should be employed up to the point where the cost of increasing $X_2$ by one unit just equals the shadow value of the soil saved as consequence. Finally substituting [2.5] in [2.7] yields:

$$\frac{\dot{\mu}}{\mu}+\left[\left(g(X_2)\frac{\partial Y(\bullet)}{\partial Q}\right)\bigg/\frac{\partial Y(\bullet)}{\partial R}\right]=\delta \qquad [2.8]$$

where the first term of the left-hand site is the capital gain on soil depth, the second term is the rate of return on soil depth holding its shadow price ($\mu$) constant, and the whole left-hand expression is the return which the farmer earns by holding on to $Q$. Expression [2.8] says that this return must equal the return that the farmer could earn on an alternative investment – the market interest rate ($\delta$).

Barbier (1990) presented a similar but slightly different model from that by Barrett (1997). He also takes the formal model originally developed by McConnell (1983) and adapts it to describe the soil conservation decision of farmers in the upper watershed in Java. He assumes that the land holding is fixed and only one crop is produced or, if there are multiple crops, their combined production can be explained by a simple crop production function:

$$Y=f(X_1(t),Q(t)) \qquad [2.9]$$

with,

$$f'(X_1)>0$$
$$f''(X_1)\leq 0$$
$$f'(Q)\geq 0$$
$$f''(Q)\leq 0$$
$$f'_{X_1Q}=f'_{QX_1}\geq 0$$

where, as above, $Y$ is agricultural output; $Q$ represents soil depth; and $X_1$ is a vector representing a conventional input package that includes productive inputs (including labour), crop varieties, and cropping patterns and techniques. An increase in $X_1$ leads to increased output, albeit at a declining rate. The depth of the soil ($Q$) also has a beneficial effect on crop production. The positive cross partial derivative $f'_{X_1Q}$ indicates that additional soil depth improves the crop productivity of the traditional input package, while the positive $f'_{QX_1}$ shows that an increase in these inputs adds to the beneficial effects of soil depth on crop production. However, the use of $X_1$ increases soil loss, and at an accelerating rate. Assuming that the household has a choice of adopting an alternative input package ($X_2$) of appropriate soil conservation methods, the change in soil depth is represented by the following state variable expressing soil quality dynamics:

$$\dot{Q}=q(X_1(t),X_2(t)) \qquad [2.10]$$

with,

$$q'(X_1)\leq 0$$
$$q''(X_1)\leq 0$$
$$q'(X_2)>0$$
$$q''(X_2)\leq 0$$
$$q'_{X_1X_2}=q'_{X_2X_1}<0$$

Given these assumptions, the behaviour of the farming household in response to soil erosion is determined by the impact of soil on profits. The objective of the farming household is to maximize the net present value of income stream from farmland through the optimal choice of input packages ($X_1$, $X_2$), subject to the soil quality dynamics and the initial soil quality (i.e. soil depth):

$$Max_{X_1,X_2}PV=\int_{t=0}^{\infty}e^{-\delta t}\left[pY(X_1(t),Q(t))-c_1X_1(t)-c_2X_2(t)\right]dt \qquad [2.11]$$

subject to:

$$\dot{Q}=q(X_1(t),X_2(t))$$
$$Q(0)=Q_0$$

where again $c_1$ represents the cost of the conventional crop production inputs; $c_2$ is the cost of the soil conservation input package; $p$ is the price of the crop output normalized with respect to the price of any new crop associated with $X_2$. Then, the first-order conditions are:

$$\frac{\partial H(\bullet)}{dX_1}=p\frac{\partial Y(\bullet)}{\partial X_1}-c_1+\mu(t)\frac{\partial q(\bullet)}{\partial X_1}=0 \qquad [2.12]$$

$$\frac{\partial H(\bullet)}{\partial X_2} = -c_2 + \mu(t)\frac{\partial q(\bullet)}{\partial X_2} = 0 \qquad [2.13]$$

$$\dot{\mu} = -\frac{\partial H(\bullet)}{dQ} + \mu(t)\delta = -p\frac{\partial Y(\bullet)}{\partial Q} + \mu(t)\delta \qquad [2.14]$$

Condition [2.12] indicates that, for conventional productive inputs, the value of the marginal product $p(\partial Y/\partial X_1)$ must be equal to the total costs. This includes not only the monetary costs ($c_1$) of these inputs, but also their costs in terms of worsening soil quality, $\mu(\partial Y/\partial X_1)$, where the co-state variable $\mu$ can be interpreted as the true value, or shadow price, of soil. Condition [2.13] shows that the money costs of the soil conservation package must be equated with the additional value it generates by controlling soil erosion. Condition [2.14] indicates that the implicit cost of soil loss ($\mu$) must grow at the rate of discount less the soil's contribution to current profits, $p(\partial Y/\partial Q)$. In other words, it is optimal to hold on to soil up to the point where capital gains in terms of improved future value of the land from conserving soil ($d\mu/dt$), plus the contribution of soil to current profits, $p(\partial Y/\partial Q)$, equal the opportunity cost of holding on to soil ($\mu\delta$), which represents the return the household could obtain from alternative investments. This can be seen combining conditions [2.12] and [2.14]:

$$\dot{\mu} + p\frac{\partial Y(\bullet)}{\partial Q} = \delta\mu(t) = \delta\left[\left(p\frac{\partial Y(\bullet)}{\partial X_1} - c_1\right) \Big/ \left(\frac{\partial q(\bullet)}{\partial X_1}\right)\right] \qquad [2.15]$$

where the left-hand side of the expression indicates the marginal gains for holding on to the soil as an asset, i.e. the future and current value of having additional topsoil; and the right-hand side represents the marginal costs or the forgone returns that could be earned from depleting soil today and investing the proceeds elsewhere.

Barbier (1998c) transformed the model previously developed by Barbier (1990) and McConnell(1983) in order to analyse the role of relative producer prices in influencing soil conservation and land management decisions by small farmers in Malawi. In Barbier's (1998c) model farmers have the choice between cultivating crops that result in soil erosion, that is erosive crop $y_1$, and crops that lead to some, but significantly less erosion, that is less-erosive crop $y_2$. Yields of both crops are assumed to be increasing concave functions of soil quality ($Q$) represented by topsoil depth:

$$y_1 = f_1(Q(t)) \tag{2.16}$$

$$y_2 = f_2(Q(t)) \tag{2.17}$$

Changes in topsoil depth depends on erosion control effort $g(Q)$, expressed by the input package ($X_2$) of appropriate soil conservation methods, less the relative impacts of the erosive and less erosive crops, $h(y_1,y_2)$, according to the following expression:

$$\dot{Q} = q(y_1(t), y_2(t), X_2(Q)) = g(X_2(Q)) - h(y_1, y_2) \tag{2.18}$$

Then, the decision problem for the farmer is to maximize the net returns from production over time by choosing the optimal combination of erosive and less-erosive crops:

$$Max_{y_1,y_2} PV = \int_{t=0}^{\infty} e^{-\delta t} \left[ p_1 y_1(Q) + p_2 y_2(Q) - cg(Q) \right] dt \tag{2.19}$$

subject to:

$$\dot{Q} = q(y_1(t), y_2(t), X_2(Q)) = g(X_2(Q)) - h(y_1, y_2)$$

with,

$$h(y_1) > h(y_2) > 0$$
$$g'(Q) < 0$$
$$g''(Q) > 0$$
$$y_1(t), y_2(t), Q(t) \geq 0$$
$$Q(0) = Q_0$$

where the erosion control effort $g(Q)$ is subject to constant cost $c$; and $p_1$ and $p_2$ are the crop prices for erosive crops and less erosive crops respectively. Then, assuming an interior solution, the first order conditions are given by the following expressions:

$$\frac{\partial H(\bullet)}{\partial y_1} = p_1 - \mu(t) \frac{\partial h(\bullet)}{\partial y_1} = 0 \tag{2.20}$$

$$\frac{\partial H(\bullet)}{\partial y_2} = p_2 - \mu(t) \frac{\partial h(\bullet)}{\partial y_2} = 0 \tag{2.21}$$

$$\dot{\mu}=p_1\frac{\partial y_1(\bullet)}{\partial Q}+p_2\frac{\partial y_2(\bullet)}{\partial Q}+\left(c-\mu(t)\right)\frac{\partial g(\bullet)}{\partial Q}+\mu(t)\left(\frac{\partial h(\bullet)}{\partial y_1(\bullet)}\frac{\partial y_1(\bullet)}{\partial Q}+\frac{\partial h(\bullet)}{\partial y_2(\bullet)}\frac{\partial y_2(\bullet)}{\partial Q}\right)+\mu(t)\delta \qquad [2.22]$$

Conditions [2.20] and [2.21] indicate that farmers should chose to cultivate each erosive and less-erosive crop to the point where the marginal returns from cultivation equal the marginal user costs of soil erosion, where $\mu$(t) can be interpreted as the shadow price of the soil. In particular, by rearranging the above first-order conditions, Barbier (1998c) shows that farmers will adjust choice of crops, the rate of soil erosion and soil conservation effort depending on the change of relative crop prices. Assuming that the percentage rate of increase in soil erosion from cultivating additional erosive crops is constant (and equal to $a$), he obtains the following expression:

$$\frac{\dot{p}_1}{p_2}=(a+\delta)\frac{p_1}{p_2}-\left[\frac{p_1}{p_2}-\frac{c}{p_2}\frac{\partial h(\bullet)}{\partial y_1}\right]\frac{\partial g(\bullet)}{\partial Q} \qquad [2.23]$$

This expression says that if the price of more erosive crops is increasing relative to less-erosive crops, then the costs of conserving soil exceed the returns. On the contrary, if the relative price is falling, then the returns to conservation will exceed the costs and the farmer will change his farming system and practices to reflect this.

The problem of land allocation between two different types of land use has been conceptually explored by Larson & Bromley (1990) within the framework of the onsite externalities of land use due to soil quality degradation. They developed a dynamic farm household model of agricultural production that includes the relation between soil quality and food production, the effects of food production on soil, and activities that improve soil quality such as fallow rotation. Larson & Bromley (1990) further assumed that the household also produces firewood ($y_2$) on the fallow land using labour and subject to a total tree biomass whose growth is represented by another state variable included into the model[5]. The model assumes that at the beginning of the planning horizon $t_0 = 0$ the household allocates its fixed quantity of land ($A$) between tillage and fallow, according to a land allocation parameter ($0 \leq \alpha \leq 1$) which remains constant throughout the planning horizon, to produce a staple crop ($y_1$) on its tilled land using a package of variable inputs

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[5] Loehman & Randhir (1999) recently extended the Larson & Bromley's (1990) model embedding the rural household that performs agricultural production in a two-sector economy that also includes urban households which perform manufacturing production.

($X_1$) that in this case is restricted only to labour. It is also assumed that the household harvests firewood from fallow land employing labour ($X_2$). The two production functions are then represented by the following expressions:

$$y_1 = f_1\big(X_1(t), Q(t), \alpha A\big) \tag{2.24}$$

$$y_2 = f_2\big(X_2(t), B(t), (1-\alpha)A\big) \tag{2.25}$$

where $Q$ is an index of soil quality over all of the household's land; $\alpha A$ represents the land allocated to tillage; $(1\text{-}\alpha)A$ is the land allocated to fallow; $B$ represents the total tree biomass which only grows on fallow land; the production function $y_1(t)$ is non-decreasing in $X_1$, $Q$ and $\alpha$; and $y_2(t)$ is non-decreasing in $X_2$ and $B$, and non-increasing in $\alpha$.

In this model the dynamics of the ecosystem are represented by the soil productivity state variable and by the net change in tree biomass under the assumption that fallow is a soil-building period that helps to increase nutrients, retain moisture in the soil, and reduce soil erosion. The net change in soil productivity at each time $t$ is then given by the following equation:

$$\dot{Q} = q\big(Q(t), B(t), X_1(t), \alpha\big) \tag{2.26}$$

with,

$$Q(0) = Q_0$$

where $Q_0$ is the household's endowment of soil productivity at the beginning of the planning horizon; the growth function $q(\bullet)$ is non-increasing in $X_1$ and $\alpha$; and $q(\bullet)$ is positive and increasing in $Q$ up to some critical point beyond which $q(\bullet)$ is decreasing in $Q$. The net change in tree biomass at each time interval is given by the following expression:

$$\dot{B} = b\big(B(t), (1-\alpha)A\big) - y_2(\bullet) \tag{2.27}$$

with,

$$B(0) = B_0$$

where $B_0$ is the household's endowment of tree biomass on fallow land at the beginning of the planning horizon; and the function $b(\bullet)$ describes the natural growth of tree biomass to be non-increasing in $\alpha$ and positive and increasing in $B$ until some critical point beyond which $b(\bullet)$ is decreasing in $B$. In the Larson & Bromley's (1990) model the decision problem for the agricultural household is to maximize the present value of utility, subject to its production functions, environmental state equations and an income constraint. Then, for a given $\alpha$ and $T$, the present discounted value of household utility ($V$) is given by the following maximisation problem:

$$Max_{X_1,X_2,L} V = \int_{t=0}^{T} U(\overline{Y},L)e^{-\delta t}dt + F\big(B(T),Q(T),T,\Phi\big)e^{-\delta T} \qquad [2.28]$$

subject to:

$$\dot{B} = b(\bullet) - y_2(\bullet)$$

$$\dot{Q} = q\big(Q(t),B(t),X_1(t),\alpha\big)$$

$$B(0) = B_0$$

$$Q(0) = Q_0$$

with,

$$\overline{Y} = w(\overline{L}-L) + p_1 y_1(\bullet) + p_2 y_2(\bullet) - w(X_1+X_2) + Y_e \qquad [2.29]$$

$$\overline{L} = L + X_1 + X_2 \qquad [2.30]$$

where the household instantaneous utility function depends on leisure $L$ and on an aggregate consumption commodity or income $\overline{Y}$; $\overline{L}$ is the household's total time endowment in each period; $p_1$ is the price of the staple crop; $p_2$ is the price of firewood; $w$ is the price of labour/opportunity cost of leisure; $Y_e$ is an exogenous flow of income, such as remittance from family members in a city; and $F(\bullet)$ is the terminal value function that depends on terminal time $T$, terminal values for soil productivity and tree biomass, and a vector of parameters $\Phi$. Given the Hamiltonian:

$$H = U(\overline{Y},L) + \mu_1(t)q(\bullet) + \mu_2(t)\big[b(\bullet) - y_2(\bullet)\big] \qquad [2.31]$$

the first order conditions for a finite period of time requires:

$$\frac{\partial H(\bullet)}{\partial L}=\frac{\partial U}{\partial \bar{Y}}\frac{\partial \bar{Y}}{\partial L}+\frac{\partial U}{\partial L}=-w\frac{\partial U}{\partial \bar{Y}}+\frac{\partial U}{\partial L}=0 \qquad [2.32]$$

$$\frac{\partial H(\bullet)}{\partial X_1}=\frac{\partial U}{\partial \bar{Y}}\frac{\partial \bar{Y}}{\partial X_1}+\mu_1(t)\frac{\partial q(\bullet)}{\partial X_1}=\frac{\partial U}{\partial \bar{Y}}\left(p_1\frac{\partial y_1(\bullet)}{\partial X_1}-w\right)+\mu_1(t)\frac{\partial q(\bullet)}{\partial X_1}=0 \qquad [2.33]$$

$$\frac{\partial H(\bullet)}{\partial X_2}=\frac{\partial U}{\partial \bar{Y}}\frac{\partial \bar{Y}}{\partial X_2}-\mu_2(t)\frac{\partial y_2(\bullet)}{\partial X_2}=\frac{\partial U}{\partial \bar{Y}}\left(p_2\frac{\partial y_2(\bullet)}{\partial X_2}-w\right)-\mu_2(t)\frac{\partial y_2(\bullet)}{\partial X_2}=0 \qquad [2.34]$$

$$\dot{\mu}_1=-\left[\frac{\partial U}{\partial \bar{Y}}p_1\frac{\partial y_1(\bullet)}{\partial Q}+\mu_1(t)\frac{\partial q(\bullet)}{\partial Q}\right] \qquad [2.35]$$

$$\dot{\mu}_2=-\left[\frac{\partial U}{\partial \bar{Y}}p_2\frac{\partial y_2(\bullet)}{\partial B}+\mu_1\frac{\partial q(\bullet)}{\partial B}+\mu_2(t)\left(\frac{\partial b(\bullet)}{\partial B}-\frac{\partial y_2(\bullet)}{\partial B}\right)\right] \qquad [2.36]$$

$$\dot{\mu}_1(T)=\frac{\partial F(\bullet)}{\partial Q(T)} \qquad [2.37]$$

$$\dot{\mu}_2(T)=\frac{\partial F(\bullet)}{\partial B(T)} \qquad [2.38]$$

These lead to the optimal choice of land allocation ($\alpha$) represented by the following land allocation condition:

$$\int_{t=0}^{T}\left[\frac{\partial U}{\partial \bar{Y}}p_1\frac{\partial y_1(\bullet)}{\partial \alpha}+\mu_1(t)\frac{\partial q(\bullet)}{\partial \alpha}\right]dt=\int_{t=0}^{T}\left[\frac{\partial U}{\partial \bar{Y}}p_2\frac{\partial y_2(\bullet)}{\partial \alpha}+\mu_2(t)\left(\frac{\partial b(\bullet)}{\partial \alpha}-\frac{\partial y_2(\bullet)}{\partial \alpha}\right)\right]dt \qquad [2.39]$$

This expression says that anything that increases the marginal utility of land in tillage, the left-hand side, or decreases the marginal utility of land in fallow, the right-hand side, will tend to increase the quantity of land in tillage and therefore, decrease soil productivity.

All the economic models presented above help to understand the consequences on soil quality, and thus agricultural productivity, of macroeconomic and sectoral policies that affect market prices or property rights endowments. The first matter of discussion is how a change in market prices will affect the farmer's decision process. In the case of one aggregate agricultural output dependent on both traditional "soil-depleting" and "soil-conserving" input packages, Barbier (1990) shows that in the short run an increase in the costs of the traditional input package will favour the adoption of the soil conservation package over traditional inputs. In contrast, an increase in the relative price of the crop or in the costs of the soil conservation package, will discourage the adoption of conservation inputs and favour the use of the traditional input package. However, in the long run, the steady-state equilibrium shows that the optimal policy of the farming household depends on whether the initial quality level (i.e. soil depth) of soil is high or low. For a similar case, Barrett (1997) showed that price reforms will not necessarily benefit soil conservation by providing incentives for farmers to grow less-erosive crops. He considers that changes in the output price or input prices could have an effect on soil quality, but the direction of the change will in general be indeterminate.

When the choice is between two different kind of crops, which in turn influence the rate of soil erosion and soil conservation effort, Barbier (1998c) shows that, if the price of erosive crops is increasing (decreasing) relative to less-erosive crops, then the farmer would be expected to change cultivation of erosive and less-erosive crops, and their impact on soil erosion and erosion control effort accordingly. A similar conclusion is also supported by Larson & Bromley (1990) in solving the land allocation problem between tillage and fallow land and the respective yields of "soil-erosive" crop and "soil-conservative" firewood. However, as Barbier (1998c) points out, this may not be the case in third world countries when fluctuations in relative crop prices and returns may be exerting a significant impact on the incentives from smallholders to invest in improved cropping systems and land management. In these countries what matters most is the degree of price risk, given the very small margins for risk among smallholders and the widespread prevalence of household food insecurity. Then, the uncertainty arising from fluctuating prices and returns is not conducive to improving farming systems and soil conservation efforts.

Structural reforms that intend to modify the property right regime through land tenure privatisation may not solve the problem of soil depletion either. As showed by Larson & Bromley (1990), degradation can still occur as a response to economic incentives even when property rights have been assigned. This is particularly true for poor farmers in developing countries, as Perrings (1989) shows that the struggle to meet food consumption needs under conditions of poverty or other limited resources can result in degradation. Therefore, the assignment of property rights may not be the panacea.

### 2.2.2. Measuring soil quality dynamics: the soil nutrient balance approach

In analysing the economic implications of soil degradation a major problem is the identification of the appropriate measure of soil quality. Many soil quality indicators have been applied. These include soil quality proxies, such as topsoil depth, organic matter content or water absorption potential (Burt, 1981; Walker, 1982; Van Kooten *et al.*, 1990; Rhoton & Lindbo, 1997), or the construction of a multivariate soil quality index using a battery of quantitative and qualitative indicator with unstated or arbitrary weights (Pierce *et al.*, 1983; Smith *et al.*, 1993; Karlen *et al.*, 1997).

Among soil quality indicators, topsoil depth has been widely used (Barret, 1997; McConnell, 1983; Barbier, 1990, 1998c). Changes in soil depth may be a proxy for changes in nutrient availability. Soil depth has a positive effect on agricultural output because in deeper soils there is more room for plant roots to take hold, and more nutrients available for plant growth. In estimating the reduction in yield associated with soil erosion, these studies compare yields on eroded soils with the yields that would have been observed had the erosion not occurred, holding everything else constant. However, they ignore the potential for farmers to mitigate erosion-induced yield losses by employing more non-soil inputs like fertilizers (Barret, 1997). Moreover, measuring soil state in terms of soil depth change, it implies ignoring all other losses of nutrients from the soil that are not caused directly by erosion. In fact, soil erosion is only one of several factors affecting soil quality. Indicators such as topsoil nutrient balance may be more appropriate in measuring soil quality, because they also account for other factors influencing soil nutrient dynamics.

The use of soil nutrient balance as a measure of soil quality is based on the evidence that nutrients present in the topsoil, such as nitrogen (N), phosphorus (P) and potassium (K), have a positive impact, at least before reaching the saturation point, on plant growth and therefore on agricultural yield. This methodology has recently been conceptualised by the nutrient monitoring approach, also known as NUTMON (Smaling & Fresco, 1993; Smaling *et al.*, 1996). The model has been applied at different spatial scales in assessing soil nutrient depletion due to agricultural exploitation (Stoorvogel *et al.*, 1993; Smaling *et al.*, 1993; Van den Bosh *et al.*, 1998; De Jager *et al.*, 1998). This approach is based on the fact that at any time, a certain amount of organic and inorganic N and P is present in the soil, either in stable or labile plant-available forms. When measured one year later, these amounts are not necessarily the same. This is due to various processes that cause nutrients to flow into and out of the exploitable soil layers and can be summarized by the expression:

$$Q_t = Q_{t-1} + \left( \sum_i In_t^i - \sum_j Out_t^j \right) \tag{2.40}$$

where $Q_t$ is the quantity of inorganic and organic nutrients remaining in the soil at the end of time $t$; $Q_{t-1}$ is the soil inorganic and organic nutrients present at the end of the previous time period; $In_t^i$ is the $ith$ input flow of nutrients to the soil during time $t$; $Out_t^j$ is the $jth$ output flow of nutrients from the soil during time $t$. Following Stoorvogel & Smaling (1990), the soil nutrient balance is quantified by the estimation of five major inputs ($In^1$=chemical fertilizer; $In^2$=organic fertilizer; $In^3$=wet and dry deposition; $In^4$=nitrogen fixation; $In^5$=sedimentation) and five major outputs ($Out^1$=crop harvesting; $Out^2$=crop residues; $Out^3$=leaching; $Out^4$=gaseous losses; $Out^5$=soil erosion) of nutrients. The difference between N and P dynamics is due to the fact that fixation ($In^4$), denitrification-volatilization ($Out^4$), and leaching ($Out^3$) only apply to nitrogen dynamics. Indeed leaching is an important loss mechanism for some nutrients. In tropical soils, however, phosphorus is often tightly bound by soil particles and leaching can be assumed to play a part only with respect to nitrogen and eventually potassium (Stoorvogel & Smaling, 1990).

Nutrient balance is thus the difference between input and output flows of nutrients. However, a good measure of soil quality would need to consider the impact of nutrient balance relative to an initial soil nutrient stock. Therefore, an indicator of soil quality will incorporate both the rate of soil nutrient depletion and the soil nutrient stock. Thus, expression [2.40] can also be defined as:

$$Q_T = Q_{t=0} + \int_{t=1}^{T} \int_{area} \left( \sum_i In_t^i - \sum_j Out_t^j \right) \tag{2.41}$$

That is, at any period of time $T$, simple specification of the balance of nutrients in soils of agro-ecosystems is expressed by the net difference between the inputs and outputs of nutrients integrated over a certain area and time. In this expression it is assumed that the given stock of nutrients into the soil at the initial time $t = 0$ is in a steady state where the inputs equal the outputs. This could be when soil is covered by natural forest or natural vegetation in general. Once soil is cleared of its natural vegetation, the nutrient balance is no longer in equilibrium. That is, the land use change creates a perturbation in the soil's nutrient balance that may be either positive (excess of input flows over outputs) or negative (excess of output flows over inputs).

It should be recognised that different agricultural production systems may lead to different forms of imbalances. Agricultural practises with high external inputs, e.g. in Europe, result in positive soil nutrient balances leading to pollution of ground and surface waters. Agricultural practices with low external inputs, as frequently found in tropical countries, may result in the depletion of soil nutrient stocks, seriously threatening future agricultural production. This is a problem concerning many African countries.

## 2.3. Offsite effects of anthropogenic activities and soil degradation

There are many possible downstream or offsite impacts of anthropogenic activities and soil degradation that result from nutrient runoff and sedimentation. These impacts include reservoir sedimentation, losses to navigation, irregular flow of irrigation, effects on agricultural, fishing and industrial production in lowland and coastal regions, impacts on water supply and potability, and impacts on drought or flood cycles. In estimating the offsite impact of nutrient loading, physical models have been linked with economic models to regulate and control point and nonpoint pollution. The physical models are used to simulate the impact of management practices and weather variables on the formation and movement of pollutants from point and nonpoint sources. The economic models estimate the offsite economic costs of the pollutant's loading and provide an optimal control strategy.

### 2.3.1. Economic models of offsite externalities

The resulting economic costs of offsite impact of soil degradation would be normally measured in terms of the present value of foregone net economic benefits from any loss of downstream economic activity or from any direct welfare effect. In the literature there are many studies that analyse the economic and policy issues concerning offsite pollution externalities, particularly for nonpoint pollution sources (Xepapadeas, 1991, 1992, 1994;

Cabe & Herriges, 1992; Eiswerth, 1993; Shogren, 1993; Kim *et al.*, 1993; Dosi & Moretto, 1993, 1994). Following this line, Shortle & Abler (1997) presented an economic model of downstream pollution for a lake's watershed that is probably one of the most complete as it includes also elements of uncertainty. Their model assumes that the ambient concentration ($a$) for a given pollutant in the water body depends on runoff from nonpoint sources and point sources discharges, and emissions from natural resources according to the following equation:

$$a = a\left(\sum_{i}^{n} r_i, \sum_{j}^{k} h_j, b, w, \lambda\right) \qquad [2.42]$$

where $r_i$ is the runoff from the i*th* nonpoint source, with $i$ representing an agricultural firm; $h_j$ is discharge from point-source firm $j$ that is considered to be observable and nonstochastic; $b$ is the natural generation of the pollutant; $w$ is a vector of weather variables that influence transport and fate and is considered to be stochastic; and $\lambda$ is a vector of watershed characteristics and parameters that is also uncertain. Runoff from the i*th* firm generating nonpoint pollution is assumed to depend on management decisions, weather and sites characteristics:

$$r_i = r_i(X_i, w, \sigma_i) \qquad [2.43]$$

where $X_i$ is a set of inputs chosen by the i*th* firm representing site management; and $\sigma_i$ is a vector of site characteristics such as soil type and topography that can be assumed to be uncertain. Then, assuming that polluters are risk neutral, the expected net social benefit function ($V$) of the resource allocation decisions by point and nonpoint sources is given by:

$$V = \sum_{i}^{n} \pi_i^e(X_i, \theta_i) + \sum_{j}^{k} \pi_j^e(h_j, \theta_j) - E[D(a(\bullet))] \qquad [2.44]$$

where $\pi^e{}_i$ is the expected profit for the i*th* nonpoint-source firm; $\pi^e{}_j$ represents the expected profit for the j*th* point-source firm; $\theta_i$ and $\theta_j$ are vectors of firm specific parameters representing the firm's private knowledge; $D(a)$ represents the economic cost of downstream damages caused by pollution and taken to be increasing in the ambient concentration ($a$) of the pollutant; and $E$ denotes expectations over all the stochastic or inherently unknown variables, i.e. $w$, $\lambda$, and $\sigma_i$. Then, with appropriate continuity and convexity assumptions, the first-order conditions for the efficient plan allocation are:

$$\frac{\partial \pi_i^e}{\partial x_{im}} = E\left[D'(\bullet)\frac{\partial a(\bullet)}{\partial r_i(\bullet)}\frac{\partial r_i(\bullet)}{\partial x_{im}}\right] \tag{2.45}$$

$$\frac{\partial \pi_j^e}{\partial h_j} = E\left[D'(\bullet)\frac{\partial a(\bullet)}{\partial h_i}\right] \tag{2.46}$$

The first condition requires that the expected increase in profit for firm $i$ from its use of input $m$ to equal the expected marginal environmental damage cost induced by the firm's use of the input. Similarly for the condition concerning the point pollution firm. The left-hand side in condition [2.45] can also be interpreted as the opportunity cost of a reduction in the use of input $m$ at site $i$, or the marginal cost of the improvement; while the right-hand site is the expected marginal benefit of the environmental improvement. Conditions [2.45] and [2.46] can be rewritten respectively as:

$$\frac{\partial \pi_i^e}{\partial x_{im}} = E\left[D'(\bullet)\frac{\partial a(\bullet)}{\partial r_i(\bullet)}\frac{\partial r_i(\bullet)}{\partial x_{im}}\right] + Cov\left(D', \frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{im}}\right) \tag{2.47}$$

$$\frac{\partial \pi_j^e}{\partial h_j} = E\left[D'(\bullet)\frac{\partial a(\bullet)}{\partial h_i}\right] + Cov\left(D', \frac{\partial a}{\partial h_j}\right) \tag{2.48}$$

It can be seen then that the expected marginal benefit of the environmental improvement is composed of the expected damage cost foregone at the margin plus a covariance term that is analogous to a risk premium. The implication is that the optimal use of an input affecting pollution will differ from the level of use that would equate the expected cost of an environmental improvement with the expected marginal benefit. The sign and size of the difference will depend on the sign and magnitude of the risk effect (Shortle & Abler, 1997).

### 2.3.2. Physical models estimating nutrient loading

Nutrient loading, which is the amount of nutrient supplied to a water body per unit time, has been shown to be an important variable related to lake enrichment and eutrophication. The export of nutrients to a lake is a result of many variables such as climate, watershed characteristics and human activities, and therefore requires large expenditures of time and money to be measured systematically. For this reason, there has been a growing number of lake nutrient loadings being derived indirectly, contributing to the development of effective methods for pollution assessment in data and resource poor situations. These methodologies can be summarized by the Export Coefficient Approach, which was originally developed in North America (Omernik, 1976) and subsequently adopted to predict nutrient loading on eutrophic waters (Jørgensen, 1980; OECD, 1982; Beaulac & Reckhow, 1982). Such methodology aims at predicting nutrient loading at any site along the surface water drainage network of a catchment as a function of the export of nutrients from each source in the catchment above that site. It has proven to be effective for nutrient pollution assessment in situations where extensive monitoring programs cannot be executed (Reckhow & Simpson 1980; Beaulac & Reckhow, 1982; Scheren *et al.*, 2000). The Export Coefficient Approach is analogous to the pollution sources assessment, often referred to as the Rapid Assessment Method (WHO, 1982; Economopoulos, 1993). These methodologies provide a procedure for evaluation of the pollution sources (industrial, agricultural, domestic activities and atmospheric deposition) by applying pollution load factors or export coefficients to estimate contributions of the respective sources as follows:

$$L_i = \sum_j \left( F_j * EC_j \right) \qquad [2.49]$$

where $L_i$ is the waste load for the $ith$ nutrient/pollutant; $F_j$ is the functional variable representing the $jth$ pollution source; and $EC_j$ represents the pollution intensity or export coefficient for the $jth$ functional variable (see Table 2.1).

**Table 2.1:** Functional variables and pollution intensities for the principle pollution sources.

| Pollution source | Functional variable | Pollution intensity |
|---|---|---|
| Industries | Annual production | Waste production per unit product |
| Households | Population number | Annual waste production per person |
| Livestock | Livestock number | Annual waste production per animal |
| Agriculture | Area of cultivated land | Annual export coefficients of nutrients |
| Atmospheric deposition | Lake surface area | Annual deposition per unit of surface area |

It must be noticed that the actual pollutant load to a given water body may not be equal to what leaves the source at its origin. Wastewater treatment facilities and self-purification, through sedimentation, mineralization and chemical immobilization, in rivers and wetlands may reduce the actual load. Therefore, the general formula should also account for these effects introducing penetration factors in the assessment. The most common penetration factors are efficiency factors for domestic and industrial wastewater treatment facilities based on the purification performances of the operating plants (Scheren *et al.*, 2000).

This kind of methodology carries a significant level of error and uncertainty due to the choice of the appropriate export coefficients or pollution intensities. Then, this problem needs to be taken into consideration during the prediction of the nutrient loadings and a procedure using modelling and error analysis must be used (Reckhow & Simpson, 1980).

## 2.4. Soil erosion in relation to onsite and offsite effects of land use

Soil erosion represents a very important factor in estimating both the onsite and offsite effects of soil degradation. Erosion and runoff of topsoil particles contribute to the depletion of nutrients and it is therefore an important element in nutrient balance models (§ 2.2.2). On the other hand nutrients, which in their organic state are carried away by the eroded soil to which they are bound, represent one of the major sources of nutrient enrichment of receiving water bodies, together with inflows of soluble nutrients from leaching and atmospheric deposition.

### 2.4.1. Predicting soil erosion: the Universal Soil Loss Equation (USLE)

The development of empirical equations to calculate field soil loss began about the 1940 in the USA. Zingg (1940) first published an equation relating soil loss rate to length and percentage slope. In the following years several other studies (Smith, 1941; Browning *et al.*, 1947; Musgrave, 1947), mostly applied to the Corn Belt region in the USA, added other factors influencing soil loss, such as rainfall, crop and conservation practice factors. The accumulation of knowledge over the years finally led Wischmeier & Smith (1978) to propose a soil loss equation which was universally applicable: the Universal Soil Loss Equation (USLE). This equation was empirically derived from over 10,000 plot-years of data in the USA and has recently been revised (McCool *et al.*, 1987, 1989; Renard *et al.* 1991; Moore & Wilson, 1992).

The USLE is an erosion model designed to predict the long-time average soil losses from fields under specific cropping and management systems. This approach is based on the evidence that the most important factor in explaining soil erosion is the rainfall. However, empirical evidence shows how one storm can cause more erosion than another on the same land, and that the same storm will cause more erosion on one field than on another. This is because rainfall erosion is the interaction of two items, the rain and the soil, where the effect of the rain is called erosivity and the effect of the soil is called erodibility (Hudson, 1984).

Erosivity is the potential ability of rain to cause erosion and is a function of the physical characteristics of rainfall. Erodibility, on the other hand, is the vulnerability of the soil to erosion and is a function of both the physical characteristics of the soil (nature of soil and slope of the land) and the management of the soil (kind of crop-land cover and soil management practice). Hence, the impact of rainfall on soil erosion is exerted through both natural physical conditions and human influence.

As soil erosion is influenced by many different variables, the essence of the USLE is to isolate each variable and reduce its effect to a number, so that when the numbers are multiplied together the answer is the amount of soil loss. The factors that control soil erosion are combined in the empirical USLE in the form (Wischmeier & Smith, 1978):

$$A = R * K * LS * C * P \qquad [2.50]$$

where $A$ expresses soil loss (t $ha^{-1}$ $yr^{-1}$); $R$ represents rain erosivity; $K$ is a measure of soil erodibility; $SL$ is slope steepness and length combined in a single index; $C$ is an index of land cover practice; and $P$ represents soil management practice.

#### *2.4.1.1. Rain erosivity*

The rainfall factor ($R$) shows the capability of a rainstorm to erode a given unit of soil. This index corresponds to the potential erosion risk in a given region where sheet erosion appears on a bare plot with 9% slope. Rainfall erosivity is a physical process, which requires expenditure of energy. It may be divided into splash erosion and surface run-off, where the kinetic energy of the drops of rain is spent in breaking down soil aggregates, in splashing them in the air, in scouring and carrying away soil particles through surface run-off (Hudson, 1984).

In the USLE the rainfall factor embeds this energy-related erosion processes being equal to the *EI index* (Wischmeier *et al.*, 1958) in which the kinetic energy of rainfall $E$ is multiplied by $I_{30}$, the maximum intensity of rain in any 30 minutes period during a storm expressed in cm per hour. However, in several studies in the tropics the *EI index* has been found to be less effective as a measure of erosivity. This has led to a search for alternative methods. One, developed by Hudson (1965) in Africa, is based on the fact that little erosion takes place at low intensities. At low intensities rain is composed mainly of small drops falling with low velocity, and hence low energy. Even if a little splash erosion does occur, there is usually no run-off to carry away the splashed particles. The Hudson's (1965) erosivity index $KE>25$ takes into account this threshold, considering only the total kinetic energy of all the rain falling at more then 25mm per hour. The same concept has been modified by Morgan (1977) for temperate regions using a lower threshold value of 10 mm per hour.

The $EI$ and $KE>25$ indexes can be calculated using empirical equations which relate the kinetic energy of rainfall to its intensity (Table 2.2). After choosing the appropriate energy-intensity relationship, it is possible to estimate the kinetic energy of rainfall for a specific intensity interval ($i$) using the interval's central value as independent variable (Table 2.3). Then, the $E$ term in the $EI$ (or $KE>25$) index is the sum over all the intensity intervals of the annual precipitation value (mm). The value ($E$) from this sum is then multiplied by the $I_{30}$ value to obtain the $EI$ index.

**Table 2.2:** Energy-intensity relationships.

| | Energy Unit | Intensity Unit | Country | Source |
|---|---|---|---|---|
| $E_i$=916+331*$\log_{10}(I_i)$ | foot-tons/acre | in/h | USA | Wischmeier et al. (1958) |
| $E_i$=11.9+8.7*$\log_{10}(I_i)$ | J/m$^2$ | mm/h | USA | Wischmeier & Smith (1978) |
| $E_i$=30-(125/$I_i$) | J/m$^2$ | mm/h | Zimbabwe | Hudson (1965) |
| $E_i$=9.81+11.25*$\log_{10}(I_i)$ | J/m$^2$ | mm/h | Italy | Zanchi & Torri (1981) |

**Table 2.3:** Estimated kinetic energy of rainfall.

| Interval intensity ($I_i$) (mm/h) | Energy[a] ($E_i$) (J/m$^2$) | Energy[b] ($E_i$) (J/m$^2$) |
|---|---|---|
| 0-25 | 21.6 | 20.4 |
| 25-50 | 25.6 | 26.7 |
| 50-75 | 27.6 | 28.0 |
| >75 | 28.3 | 28.4 |

Referring to: a = Wischmeier & Smith (1978); b = Hudson (1965).

### *2.4.1.2. Soil erodibility*

The soil erodibility factor (*K*) varies from 0.7, for the most fragile soil, to 0.01, for the most stable soil. It is measured on bare reference plots 22.2 m long on 9% slopes, tilled in the direction of the slope and having received no organic matter for more then two years. If a gradient other than 9% must be used, the data are adjusted by the *LS* factor in the USLE. Soil erodibility depends on the organic matter and texture of the soil, its permeability and profile structure. In taking account of these variables a soil erodibility nomograph is used through the following equation (Wischmeier & Smith, 1978), which has to be multiplied by a conversion factor of 1.292 when using metric measures:

$$K = \frac{\left[2.1M^{1.14}(10^{-4})(12-a)+3.25(b-2)+2.5(c-3)\right]}{100}*1.292 \qquad [2.51]$$

where *M* is a particle-size parameters which equals percent silt times the quantity 100-%clay; *a* is percent organic matter; *b* is the soil structure code value (1:very fine granular; 2:fine granular; 3:med. or coarse granular; 4:blocky, platy or massive); and *c* is the profile permeability class (1:rapid; 2:mod. to rapid; 3:moderate; 4:slow to mod.; 5:slow; 6:very slow).

### *2.4.1.3. Slope steepness and length*

The topographical factor (*LS*) depends on both the length and gradient of the slope. It is computed by the product of the slope factor (*S*), a ratio which compares soil loss with that from a field of a 9% slope, by the length factor (*L*), a ratio which compares soil loss with that from a field of a specified length of 22.6 m. The *LS index* varies from 0.1 to 5, however, in mountainous areas, it may reach 20 or above. In the original model (Wischmeier & Smith, 1978) the index is expressed in English measures and may be estimated from the average slope gradient $s$ (expressed in %) and the average length of the slope $l$ (expressed in feet):

$$LS = L * S \qquad [2.52]$$

$$L = \frac{l^{0.5}}{72.6}$$

$$S = \frac{\left(0.43 + 0.3s + 0.043s^2\right)}{6.613}$$

These figures may be converted in metrics values, where the average length $l$ is expressed in m, as follows:

$$LS = \left(L * S\right) * 3.15 \qquad [2.53]$$

$$L = \left[\frac{l^{0.5}}{22.6}\right] * 1.292$$

$$S = \left[\frac{\left(0.43 + 0.3s + 0.043s^2\right)}{6.613}\right] * 1.292$$

Estimation of the *LS* factor, however, poses more problems than any of the other factors in the USLE (Wilson, 1986; Renard *et al.*, 1991). For this reason there has been substantial research by many authors in revising the *LS* parameter estimation (Moore & Burch, 1986a, 1986b; McCool *et al.*, 1987, 1989)

### *2.4.1.4. Land cover*

The land cover factor (*C*) compares the erosion of a bare soil plot with that from a plot with a specific plant cover. The index combines plant cover, its production level and associated cropping techniques. It varies from 1 on bare soil, to 0.001 under forest, 0.01

under grassland and cover plants, and 0.1 under root and tuber crops. The calculation of *C* is rather complicated. The protective effect of the crop and its management varies during a year, as also does erosivity. Therefore, the factor of *C* is given by the sum of five crop-stage periods: 1) fallow or preparation of seedbed; 2) seedbed (1st month after seeding); 3) establishment (2nd month after seeding); 4) growing cover (3rd month from seeding to harvest); and 5) stubble (from harvest to ploughing).

Many values for the land cover factor are available from the Wischmeier & Smith (1978) study, where the soil loss during the five periods has been measured in many combinations of cover, crop residue, productivity, growing season, tillage practice and residue management. However, they refer to the USA and are not particularly suitable to tropical regions. In this case, land cover factors may be taken by other studies in the region or may be estimated through field measurements.

#### *2.4.1.5. Support practice*

The support practice factor (*P*) is defined in the USLE as the ratio of soil loss with a specific support practice to the corresponding loss with up-and-down-slope cultivation. The most important of these supporting cropland practices are contour tillage, stripcropping on the contour, and terrace systems. In the USLE the support practice factor (*P*) can generally be read for percent of the slope and kind of soil conservation measure used (Table 2.4).

**Table 2.4:** Values for the support practice factor P in the USLE.

| % slope | Contour farming | Contour farming + strip cropping +rotation of crops | Contour farming +terracing |
|---|---|---|---|
| 1 – 2 | 0.6 | 0.30 | 0.12 |
| 3 – 8 | 0.5 | 0.25 | 0.10 |
| 9– 12 | 0.6 | 0.30 | 0.12 |
| 13 – 16 | 0.7 | 0.35 | 0.14 |
| 17 – 20 | 0.8 | 0.40 | 0.16 |
| 21 – 25 | 0.9 | 0.45 | 0.18 |
| 26 – 55 | 1 | 0.50 | n.a. |

***Source:*** Wischmeier & Smith (1978).

From the table above it can be seen that contour farming on slopes of 3-8% can reduce soil loss by 50% compared with cultivating up-and-down hill. Combined with strip cropping the soil loss will be reduced by a further 50%. Terracing combined with contour farming and other conservation practices is even more effective than those practices without terraces. Improved tillage practices, sod-based rotations, fertility treatments, and greater quantities of crop residues left on the field contribute materially to erosion control and frequently provide the major control in a farmer's field. However, these are considered conservation cropping and management practice, and then the benefits derived from them are included in $C$.

#### *2.4.1.6. Weakness and limitations of the USLE method*

The major purpose of the USLE is to supply specific and reliable guides for selecting adequate erosion control practices for farm fields. The procedure is also useful for computing the upland erosion phase of sediment yield as a step in predicting rates of reservoir sedimentation or stream loading. In this last case, however, the amount of sediments that reach the stream system is only a fraction of the soil erosion. Eroded soil materials often move only short distances before a decrease in runoff velocity causes their deposition. They may remain in the fields where they originated or may be deposited on more level slopes that are remote from the stream system. Estimates show that only about one-fourth of the amount of sediment moved by flowing water in the USA annually reaches major streams (Stewart *et al.*, 1975).

Therefore, if the USLE is used in order to estimate sediment yield directly from soil erosion, it provides values that will be clearly overestimated. Soil erosion values will approach sediment delivery values only if the runoff drains directly into a lake or stream system with no intervening obstructions or flattering of the land slope. In all the other cases, models to estimate watershed sediment yields from soil erosion must be used. The general method for estimating sediment yields using the USLE is the Gross Erosion-Sediment Delivery Method (U.S. Department of Agriculture, 1971), which is summarized by the following equation:

$$Y = \frac{A * d}{W_s} \qquad [2.54]$$

where $A$ is the gross erosion estimated by the USLE; $d$ is the sediment delivery ratio; and $W_s$ is the area of the watershed above the point for which the sediment yield is being computed. The sediment delivery ratio ($d$) for a specific drainage area is the ratio of sediment delivered at a given location in the stream system to the gross erosion from the drainage area above that location. This ratio must be estimated for specific drainage areas directly from local data, as a general equation for computing watershed delivery ratios is not available. However, available watershed data indicate that the delivery ratio varies approximately as the 0.2 power of drainage-area size, with indications that the exponent in this relationship may be as small as 0.1 for very large areas (Wischmeier & Smith, 1978).

Another major problem is related to the applicability of the USLE in estimating the gross erosion in the Gross Erosion-Sediment Delivery Method. The USLE can be used to estimate the sediment generated by sheet and rill erosion that is usually, but not always, the major portion of a watershed's gross erosion. Sediment from gully, streambank and streambed erosion, and from uncontrolled roadsides, must be added to the USLE estimates.

For all these reasons the USLE empirical model does not always satisfy scientists who are looking for physical models based on the primary erosion processes and also hope to identify the processes occurring in isolated rainstorms instead of average values collected over 20 years. However, it is not certain that such physical models will be more effective than best locally adapted versions of present empirical models based on the USLE (Wischmeier & Smith, 1978) and its revised version (Renard *et al.*, 1991). Nevertheless, efforts to improve the prediction and estimation of soil erosion are continuing and the research is developing improved, process-based erosion-prediction models, such as the WEPP (Laflen *et al.*, 1991) for instance, aiming to replace the USLE in the near future. At present, however, the USLE is been used as a practical guide for engineers and is still applied in many countries.

## 2.5. Discussion

The flow of nutrients from land to water bodies may generate both onsite externalities on agricultural productivity and offsite externalities on fishery yields and water use. Physical models, such as the soil nutrient balance approach and the export coefficient approach, are used for the prediction of these flows. The soil nutrient balance is most often used to predict nutrient dynamics in cultivated land. The export coefficient model is a more general approach as it includes also point pollution sources. The former then seems more appropriate in predicting onsite effect of agricultural exploitation in terms of soil nutrient depletion, while the latter is more complete for the estimation of total nutrient flow from a watershed to its main water body. Nevertheless, a combination of both appears to be the best solution as the former approach can be an integrated part of the latter, if used to estimate export coefficients from agricultural nonpoint pollution sources.

In this chapter we have also stressed out the importance of soil erosion in nutrient balance models. Predictive methods such as the USLE may be used to estimate the fraction of nutrient outflow due to erosion, knowing the level of nutrient concentrations in the eroded topsoil. The use of the USLE accounts for the outflow of nutrients that are chemically bound to soil particles or are not in soluble forms. Then, it does not lead to doublecounting with respect to nutrient leaching.

The final objective of this thesis is to model the relation between land use change and fish stock biomass through the impact on water quality. A complete land allocation problem should include the economic models of both on-site and off-site externalities. In chapter 7 we are presenting a theoretical model (§ 7.2), which is partly inspired by the literature reviewed in this chapter. However, in applying the model to a specific sub-basin of Lake Victoria (§ 7.5.1), we focus on the off-site effect of land degradation, without embedding any economic model of on-site externalities. Physical models such us the soil nutrient balance approach (§ 2.2.2) and the USLE (§ 2.4.1) are used to simulate the impact of management practices. These physical models lead to the prediction of soil nutrient dynamics that represent a constraint in the economic optimization of land use.

# Chapter 3

# THE WETLAND COMPONENT

## 3.1. Introduction

Wetlands cover around 150 million hectares worldwide, about 1% of the world's land surface. Nevertheless, there is some disagreement among scientists on the perception of what constitutes a wetland system and in defining the generic characteristics of wetlands. This is partially due to their highly dynamic character and to the difficulties in defining their boundaries with any precision (Mitsch & Gosselink, 2000).

The complexity involved is also evident in the definition of wetland given by the Ramsar Convention[6]: "*areas of marsh, fen, peat land or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt including areas of marine water, the depth of which at low tide does not exceed 6 meters*". It is partially because of this complexity that many human activities as well as inconsistent or contradictory policies result in negative effects on wetlands ecosystems (Turner *et al.*, 2000).

According to Turner *et al.* (1995), three types of interrelated failures can be identified as playing an important role in this process: information, market and intervention failures. Indeed, market and intervention failures derive from a fundamental failure of information, caused by the lack of understanding of the multiple values and services that may be associated with wetlands (Turner & Jones, 1991). It follows that general public does not correctly perceive the indirect consequences of land use, water management, agricultural pollution, air pollution and infrastructure for the quality and sustainability of wetlands. Moreover, decision makers may not understand the consequences of their actions. Where the consequences of intervention failures include

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[6] Wetlands are the only single group of ecosystems to have their own international convention. The Ramsar Convention, which came into force in 1975, is an inter-governmental conservation treaty, where a framework for international co-operation was provided for the conservation of wetland habitats to ensure their conservation and wise use. At present 156 countries are Ramsar Contracting Parties, with 1676 wetland sites included in the Ramsar List of Wetlands of International Importance (http://www.ramsar.org/index.html).

effects on the general public or third parties, then it matters that policy makers also do not understand the functions of wetlands.

In fact, wetlands provide many important goods and services to human society, but are at the same time ecologically sensitive and adaptive systems (Turner *et al.*, 2000). Wetland-dependent fish, shellfish, plants and timber provide a valuable goods and income earnings opportunities, particularly in developing countries. At the ecosystem level, wetlands moderate the effects of floods, improve water quality, provide habitat to a very wide range of species, support agriculture, forestry and fisheries, recharge aquifers, and have aesthetic and heritage value. They also contribute to the stability of global levels of available nitrogen, athmospheric sulfur, carbon dioxide and methane (Mitsch & Gosselink, 2000). Some of these services are reflected in market transaction. Many more are not, especially in developing countries.

Because of the many non-market services wetlands provide, wetland management is crucial to developing countries (Barbier, 1998a). These economies are typically highly dependent on primary production as source of long-term sustainable economic development (Barbier, 1994). Wetlands protection and conservation is then essential to support economic activity and human welfare (Barbier, 1998a).

To estimate the net benefits of alternative management regimes requires an economic evaluation of the goods and services that wetlands provide under each regime. Economic analysis of wetlands has generally focused on the choice between two extreme regimes: wetland conservation on the one hand and wetland conversion on the other (Dixon & Lal, 1997). More generally we need estimates of wetlands benefits on a range of management regimes reflecting three types of values, i.e. direct use values, indirect use values, and non-use values (Barbier, 1998b).

Most of the benefits of wetland conservation involve either direct or indirect use values, for which the most effective valuation technique is the production function approach. This is particularly used to estimate the indirect use value of regulatory ecological functions exerted by wetlands. Among the first to provide the theoretical background for the application of this methodology to wetland valuation were Ellis & Fisher (1987). Their aim was to value the environment as an input to the production of a marketed good. They brought the examples of production of marketed goods such as freshwater and shellfish, suggesting using impacts of an environmental change on product revenues as a measure of the value of the change. A previous study by Lynne *et al.* (1981) also suggested that may be possible to isolate the contribution of environmental inputs to production, valuing both wetland acreage and human input in the production of blue crabs in Florida. However, the analysis was not carried through to obtain estimates of consumer and producer surplus associated with specified increases in wetland acreage.

Other recent studies have focused on the linkage between wetland and fisheries. Barbier & Strand (1997) studied a mangrove wetland on the coast of Mexico. Swallow (1994) evaluated the tradeoffs involved in converting wetlands to agriculture rather than maintaining them to protect salinity in estuarine nursery areas. Turner (1982) illustrated the relationship between the area of wetland or the wetland-water interface of coastal wetlands and the production of commercial fisheries for a number of case studies around the world including marine, freshwater and pond raised. Barbier & Sathirathai (2002) showed the economic value of mangrove wetlands for the shrimp farming industry in Thailand.

In the vast wetland valuation literature, many other studies have focused on the impacts of environmental change on activities other than fisheries. A study on the Iberà Wetlands in Argentina (Simonit *et al.*, 2005), for instance, analysed through a production function approach the impact on rice production of the increased wetland water level due to groundwater flow diversion. Extensive research has been done on the groundwater recharge effects of upstream water developments in the Hadejia-Nguru wetlands in northern Nigeria (Acharya, 1997, 2000a, 2000b; Acharya & Barbier, 1997, 2000; Barbier & Thompson, 1998; Barbier *et al.*, 1998). In these studies the impact of diverting upstream surface and groundwater flows for irrigation has been measured in terms of the opportunity cost of forgone hydrological services to downstream activities such as agriculture, livestock and grazing, timber, non-timber products, wildlife tourism, groundwater recharge, and fishery. In other cases, stated preference techniques, mainly contingent valuation techniques, have been used to estimate the recreational value of wetlands (Bell, 1997; Bergström & Stoll, 1993; Bergström *et al.*, 1990; Creel & Loomis, 1992), as well as their "non-use" value, i.e. existence value (Bateman & Langford, 1997). Other studies used methodologies, such as damage costs avoided or preventive expenditure, to estimate wetland functions such as flood control and storm protection. In this context, Costanza & Farber (1985) offer an analysis of the hurricane protection value of wetlands for protection from storm surge and wind damage, whilst Farber (1987) presents a more detailed analysis of the value of coastal wetlands for protection from wind damage only. Techniques based on replacement costs of wetlands have also been used. Folke (1991), for instance, evaluates the life-support functions of a Swedish wetland system comparing the loss of wetland's functions with the costs of replacing them with, where feasible, human technologies.

All the different types of wetland values and valuation techniques mentioned above reflect the peculiar nature of wetlands as multifunctional and multiple use systems. However, in most of the studies wetlands are considered as single use systems to simplify the analysis. In this case the predominant economic value of wetland systems is a single regulatory function or a group of ecological functions providing support or protection for an economic activity (Barbier, 1994). Attempts to aggregate the value of wetland services under alternative uses have been made by few authors (Costanza *et al.*, 1997; Seidl & Steffen-Moraes, 2000). Nevertheless, this is not the same as the value of wetlands under any given management regime. Indeed, we are aware of the perils of aggregation inherent in the ecosystem valuations of Costanza *et al.* (1997). The effort to assign prices to ecosystem services is fundamentally flawed since most of the benefits fall outside the marketplace. Such services are public goods that contribute immeasurably to human welfare without ever being drawn into the money economy. Moreover, even if prices to ecosystem services can be reasonably calculated, they cannot reflect the full value of these services, which reaches well beyond their importance to the world economy.

An alternative approach to the valuation of wetlands as multifunctional systems is provided by energy analysis (Costanza, 1980). Energy analysis is based on the total amount of energy captured by natural ecosystems, as a measure of the totality of the ecosystem functions. Assuming a linear relationship between embodied energy and price, energy flow can be evaluated in terms of prices. The energy analysis method, although imprecise because of the several estimates used, appeals to many wetland scientists because it is based on the inherent productivity of the ecosystem, not on perceived anthropogenic values that may change from generation to generation and from location to location (Mitsch & Gosselink, 2000). For an economic perspective, however, this is exactly what makes it inappropriate.

In conclusion, despite their crucial role in supporting many ecological and socio-economic activities, wetlands are under considerable pressure from anthropogenic sources around the world. This is largely due to the fact that many wetland values are ignored, because they are not captured in market transactions. To understand the consequences of market failures requires an economic valuation of wetland functions. This shows their importance to policy makers and society. Several approaches to the valuation of wetlands have been advanced, falling broadly into two classes: ecological evaluation and economic valuation. The former is generally necessary before attempting the latter, since the estimated ecological functions determine the monetary value (Mitsch & Gosselink, 2000). However, the important thing to bear in mind in attempting any wetland valuation is that wetlands may be multiple use systems – that is regulatory functions may support or protect many different economic activities, and a simple economic activity may depend on more than one regulatory ecological function (Barbier, 1994). In this case, assumptions concerning the possible ecological relationships among these various multiple uses need to be carefully constructed (Aylward & Barbier, 1992; Ruitenbeek, 1992). The following section considers a set of functions relating to one important ecological service provided by wetlands: the retention of nutrients.

## 3.2. Nutrient retention in wetlands

In many countries around the world, the emission of pollutants into water bodies has caused considerable environmental and economic damage. The loading of nutrients such as nitrogen and phosphorus from agriculture, industrial and domestic wastewater is the primary cause of eutrophication in many lakes and coastal waters. Because wetlands function as pollutant sinks, in particular for nutrients, one of the aims of wetland management and restoration is pollution control.

The capacity of wetlands to purify watersheds reducing nutrients and other pollutant levels has been pointed out in several studies (Nichols, 1983; Mitsch & Jorgensen, 1989; Reckhow & Quian, 1994; Richardson *et al.*, 1997; Wang & Mitsch, 2000) and has been recently analysed also at catchment scale (Mitsch & Wang, 2000; Arheimer & Wittgren, 2002; Trepel & Palmeri, 2002). At this large scale, predictive models have been developed to characrerise the ability of wetlands to reduce point and non-point pollution from the upstream watershed.

From an economic perspective, the nutrient retention function of wetlands represents a potentially low cost means of reducing point and non-point pollution (Gren, 1993, 1995; Gren *et al.*, 1997; Byström, 1998; Byström *et al.*, 2000). In the economic literature, pollution abatement has been studied in relation to artificially restored wetlands for wastewater treatment, rather than in relation to natural wetlands. In general, the marginal value of pollution abatement is estimated by reference to alternative pollutant abatement measures. These measures include, for instance, sewage treatment plants and the reduction of pollutant discharge from agricultural activities. However, as shown in Turner *et al.* (1995), their marginal value may differ considerably.

Indeed, the value of restored wetlands as pollution sinks includes other benefits from wetland restoration, such as water buffering, biodiversity conservation, and life-support functions. The value of these ancillary benefits, due to the multifunctionality of wetlands, often makes the net value of pollution abatment exceed the value of pollution abatment via other methods. Turner *et al.* (1995) show that if $W_t$ is the marginal value of restoring the wetland at time $t$ and $S_t$ is the marginal value of an increase in the stock of sewage treatment plants at time $t$:

$$W_t = \int_t^{\infty} e^{-(r-k)(r-t)}(V + A)\delta t \qquad [3.1]$$

$$S_t = \int_t^{\infty} e^{-(r+\rho)(r-t)}(V)\delta t \qquad [3.2]$$

$$N = V \qquad [3.3]$$

where $N_t$ is the marginal value of reducing farmers' uses of polluting fertilizers at time $t$; $V$ is the marginal value of water quality; $A$ is the marginal value of ancillary benefits; $r$ is the discount rate; $k$ is the natural regeneration rate of wetlands; and $\rho$ is the rate of depreciation of the treatment plants. It follows that if $A = 0$, then $W_t > S_t$ if $(r + \rho) > (r - k)$. The difference between restored and natural wetlands resides in the fact that production of environmental services by natural wetlands may not change over time if these are at an equilibrium or steady-state. In this case there is no growth term ($k$) for natural wetlands unless these wetlands are regularly harvested. On the other hand, recently constructed wetlands change over time in the capacity to function as pollutants sinks. Their capacity increases over time as the plants grow and spread, until they reach their mature state. In the expressions above, the marginal value of improved water quality is included in the marginal values of all the pollutant abatement measures.

Considering fishery output, the marginal value of water quality can be obtained through its impact on fishery production. We have already seen how the relationship between wetland area and commercial fish stock, measured in terms of fish catches, has been widely studied through the production function approach (Lynne *et al.* 1981; Turner, 1982; Ellis & Fisher, 1987; Swallow 1994; Barbier & Strand, 1997; Barbier & Sathirathai, 2002). In these studies, the main objective is to estimate the ecological function of wetlands in supporting offshore fisheries by serving as both spawning ground and nursery for fry. To our knowledge there is no study attempting to estimate the marginal value of wetland area through the impact of water quality on fisheries.

## 3.3. Methodological approaches for modelling nutrients retention in wetlands

Several studies estimated the capacity of wetlands to retain the nutrients in water flows from the upstream catchment area. Wetland retention of nutrients can be modelled as a function of wetland area and nutrient loading (Byström, 1998; Dortch & Gerald, 1995). A positive linear relationship is often assumed. Even though nutrient retention, particularly for nitrogen, is positively correlated with inflow load (Jansson *et al.*, 1998), if nutrient retention increases with nutrient loading, there should be a point of satiation. At this point it does not matter whether nutrient loading increases, the retention rate will still be unchanged. This second view supports a non-linear relationship.

Which relationship is right requires estimation of both linear and non-linear models. This may require a considerable amount of data. An alternative approach is represented by the mass balance model, which is a simplified predictive method that requires little input of data and minimizes the cost of implementation.

### 3.3.1. Linear relationships approach

Linear relationships have often been used to explain nutrient retention as function of nutrient loading and wetland area. A simple linear model has been presented by Jansson *et al.* (1998). They applied a linear wetland load-retention model to quantify wetlands nitrogen retention in the Baltic Sea basin. The estimated equation assumes the form:

$$r = a * L_{in} \qquad [3.4]$$

and it shows a linear relationship between nutrient retention $r$ and nutrient load $L_{in}$ from the upstream catchment.

Another linear regression model, presented by Byström (1998), also includes wetland area as explanatory variable:

$$\frac{r}{A_W} = a + b\frac{L_{in}}{A_W} + \varepsilon \qquad [3.5]$$

where $r$ is the amount of nutrient retained by the wetland, $L_{in}$ is the amount of nutrient entering the wetland, $A_W$ is wetland area, $a$ and $b$ are regression coefficients, and $\varepsilon$ is the error term. This approach requires observing mean values for nutrient retention, wetland nutrient load and wetland area for several wetlands, estimating the equation coefficients through cross-sectional econometric analysis.

### 3.3.2. Non linear relationship approach

Another option in modelling nutrients retention by wetlands is to use a non-linear relationship. Byström (1998), for instance, found that a Cobb-Douglas function was more appropriate than a linear function in estimating nitrogen retention for catchments in southern Sweden. The estimated model was as follows:

$$R = a * L_{in}^{b} * A_W^{c} \qquad [3.6]$$

where $R$ is the amount of nutrient retained by the wetland; $L_{in}$ is the amount of nutrient entering the wetland; $A_w$ is wetland area; and $a$, $b$ and $c$ are parameters subject to estimation through regression analysis. The parameter estimation requires a log linear model of the form:

$$\ln\left(\frac{R}{A_W}\right) = \alpha + \beta \ln\left(\frac{L_{in}}{A_W}\right) + \varepsilon \qquad [3.7]$$

which after solving for $R$ gives,

$$R = e^{\alpha} * L_{in}^{\beta} * A_{W}^{(1-\beta)} \qquad [3.8]$$

A similar model was developed by Kadlec & Newman (1992) to estimate storage and loss rate of phosphorus from a database of over 100 freshwater wetlands throughout United States. The relationship between loading input and loading output was:

$$L_{out} = a * L_{in}^{b} * A_{W}^{c} \qquad [3.9]$$

which is equivalent to a linear log-log relationship between nutrient input and output per unit area of wetland:

$$\ln\left(\frac{L_{out}}{A_W}\right) = \alpha + \beta \ln\left(\frac{L_{in}}{A_W}\right) + \varepsilon \qquad [3.10]$$

Other related models (eg. Kadlec & Knight, 1996) explain outflow nutrient concentrations $C_{out}$ (g $m^{-3}$) as a function of inflow nutrient concentrations $C_{in}$ and hydraulic loading rate $\omega$ (cm $day^{-1}$). The latter can be obtained dividing $Q_{in}$, the hydraulic flow rate entering the wetland ($m^3$ $day^{-1}$), by $A_w$, the wetland surface area ($m^2$), to estimate the coefficients of the following expression:

$$C_{out} = a * C_{in}^{b} * \left(\frac{Q_{in}}{A_W}\right)^{c} \qquad [3.11]$$

Indeed the precise functional form may vary according to the type of nutrient being studied. Therefore, several other empirical equations for the estimation of outflow concentrations or nutrient retention can be found from field experiments.

### 3.3.3. Mass balance approaches

An alternative method for estimating the wetland retention is to use predictive models that have a theoretical base (Mitsch & Gosselink, 2000). One of these, PREWET, was developed by Dortch & Gerald (1995) and is based on a mass-balance approach. The primary assumption made in this mass-balance model is that the wetland is in a steady state (i.e., flow and concentrations are constant in time). Although wetlands may not be at steady-state, steady-state analyses are useful for evaluating long-term, average values where mean annual input conditions (e.g., flows and depth) are consistent with this assumption (Dortch & Gerald, 1995). The basic equation of the PREWET model is as follows:

$$L_{out} = L_{in} - kV_W C_W \tag{3.12}$$

where $L_{out}$ and $L_{in}$ are respectively the amount of nutrient (tonnes) exiting and entering the wetland at time $t$; $V_W$ is the volume ($m^3$) of the wetland; $C_W$ is the concentration ($g\ m^{-3}$) of nutrient into the wetland; and $k$ is the bulk loss or removal rate of nutrient because of physical, chemical or biological processes. Since the amount of nutrient entering and exiting the wetland is the product of the concentration and the hydraulic flow, $Q$ ($m^3$), the expression can be written as:

$$Q_{out} C_{out} = Q_{in} C_{in} - kV_W C_W \tag{3.13}$$

Considering several wetland inflow points ($j$) and outflow points ($i$), the general expression becomes:

$$L_{out} = \sum_i Q_{out}^i C_{out}^i = \sum_{j \neq i} Q_{in}^j C_{in}^j - kV_W C_W \tag{3.14}$$

In order to predict wetland nutrient retention, however, we need to accommodate specific wetland characteristics. In general, nutrient abatement can be written as a function of wetland area $A_w$, for given levels of nutrient load $L_{in}$ into the wetland and wetland characteristics $Z$ (Byström, 1998):

$$R = R(A_W, L_{in}, Z) \tag{3.15}$$

The specific wetland characteristics may depend on wetland vegetation, composition of soil or wetland mixing patterns. This last aspect distinguishes between two different conditions of spatial gradients in nutrients concentration within the wetland: "*fully mixed*" or "*longitudinal gradient condition*". In the first case there are no gradients and concentration within the wetland is constant and equal to that exiting the wetland. In the latter case, concentration decays exponentially within the wetland with gradients along the main flow axis. With perfect mixing the mass balance equation is stated as (Dortch & Gerald, 1995):

$$\frac{d(V_W C_W)}{dt} = Q_{in} C_{in} - Q_{out} C_{out} - k(V_W C_W) = 0 \qquad [3.16]$$

From wetland hydrology, the theoretical detention time $\tau$, without considering the porosity of medium, is defined as $\tau = V_W/Q_W$ (Mitsch & Gosselink, 2000), with the flow rate through the wetland, $Q_W$, defined as $Q_W = (Q_{in}+Q_{out})/2$. Assuming $Q_{in} = Q_{out}$ and with $C_W = C_{out}$ from the fully mixed spatial assumption, the nutrient outflow from the wetland is given by:

$$L_{out} = \frac{L_{in}}{1 + k\tau} \qquad [3.17]$$

On the other hand, in a wetland with longitudinal gradient conditions the mass balance equation, which includes the fractional distance $y$ from inlet to outlet, is (Dortch & Gerald, 1995):

$$\frac{d(Q_W C_W)}{dy} = k\tau Q_W C_W \qquad [3.18]$$

Integrating this equation over the entire length of the wetland, the solution can be expressed as a first order areal model where the inflow nutrient concentration $C_{in}$ is assumed as the concentration at the first transect of the wetland $C_W^{in}$ and the outflow nutrient concentration $C_{out}$ is that in the last transect $C_W^{out}$ :

$$\frac{Q_{out} C_W^{out}}{Q_{in} C_W^{in}} = \exp(-k\tau) \qquad [3.19]$$

or,

$$L_{out} = L_{in} \exp(-k\tau) \qquad [3.20]$$

where again it can be assumed for simplicity $Q_{in} = Q_{out} = Q_W$.

In the above equations the hydraulic residence time $\tau$ is defined as the theoretical maximum detention time $V_W/Q_W$. However, the true detention time of water parcels can be less than $V_W/Q_W$ because of dominant flow paths that result in dead zones and short-circuiting. Therefore, the true detention time $\bar{\tau}$ (days) in wetlands can be estimated as (Thackston *et al.*, 1987):

$$\bar{\tau} = 0.84 \frac{V_W}{Q_W} \left( 1 - e^{-0.59 \frac{l_W}{v_W}} \right) \qquad [3.21]$$

or,

$$\bar{\tau} = 0.84 \frac{d_W * A_W}{Q_W} \left( 1 - e^{-0.59 \frac{l_W}{v_W}} \right) \qquad [3.22]$$

where $h_w$, $l_w$ and $v_w$ are wetland depth, length and width respectively expressed in m; $A_w$ is wetland area ($m^2$); and $Q_w$ is the mean flow through the wetland ($m^3\ day^{-1}$).

A variation of the PREWET model has been developed by Kadlec & Knight (1996). Their approach, which assumes longitudinal gradient conditions within the wetland, considers the removal rate constant as an areal value reflecting background nutrient concentrations:

$$\omega \frac{dC}{dy} = k_A \left( C - \overline{C} \right) \qquad [3.23]$$

where $\omega$ is the inflowing hydraulic loading rate ($m\ yr^{-1}$); $C$ is nutrient concentration ($g\ m^{-3}$); $\overline{C}$ is residual or background nutrient concentration ($g\ m^{-3}$) below which it is generally agreed that wetlands cannot go, even in absence of external nutrient loading; and $k_A$ is the areal removal rate constant (m yr$^{-1}$). In this equation the hydraulic loading rate is defined as:

$$\omega = \frac{Q_{in}}{A_W} \qquad [3.24]$$

where $Q_{in}$ is the hydraulic flow rate entering the wetland (m$^3$ yr$^{-1}$) and $A_w$ is the wetland surface area (m$^2$). In this expression the hydraulic loading rate is equivalent to the depth (m) of flooding over the wetland per unit time (Mitsch & Gosselink, 2000).

Integrating this equation over the entire length of the wetland, the solution can be expressed as a first-order areal model:

$$\frac{C_{out} - \overline{C}}{C_{in} - \overline{C}} = \exp\left(\frac{-k_A}{\omega}\right) \qquad [3.25]$$

where $C_{in}$ is the inflow concentration of nutrient (g m$^{-3}$) and $C_{out}$ is the outflow concentration. This expression can then be rearranged as:

$$C_{out} = \overline{C} + \exp\left(\frac{-k_A A_W}{Q_{in}}\right)\left(C_{in} - \overline{C}\right) \qquad [3.26]$$

or in terms of flows,

$$L_{out} = Q_{out}\overline{C} + \exp\left(\frac{-k_A A_W}{Q_{in}}\right)\left(Q_{in}C_{in} - Q_{in}\overline{C}\right) \qquad [3.27]$$

where again it can be assumed in the long term that $Q_{in} = Q_{out}$.

To simplify the model, this approach has recently been applied by Arheimer & Wittgren (2002) without considering the background concentration values, leading to the following expression:

$$C_{out} = C_{in} \exp\left(\frac{-k_A A_W}{Q_{in}}\right) \qquad [3.28]$$

## 3.4. Nitrogen retention in wetlands

Nitrogen cycling in wetlands is controlled by various processes, such as settling, mineralization, hydrolysis, uptake and release by biota, nitrification, and denitrification (Dortch & Gerald, 1995). Through the processes of denitrification, uptake in biomass and sedimentation, wetlands have the capacity of reducing nitrogen concentration in aquatic environments (Mitsch & Gosselink, 2000).

Denitrification, however, is the only major net, long-term removal process for total nitrogen (Gale *et al.*, 1993; Howard-Williams, 1985) and it accounts for 60-90% of the total nitrogen retention in wetlands (Jansson *et al.*, 1994). Denitrification in freshwater wetlands may account for as much as 0.35 g N $m^{-2}$ $day^{-1}$ (Sloey *et al.*, 1978). Some nitrogen is also lost to burial in the sediments, but this effect is small compared with denitrification (Dortch & Gerald, 1995). Denitrification in wetlands takes place primarly in the anaerobic bottom sediments where nitrate is reduced during organic matter decomposition to gaseous $N_2$, which is released to the atmosphere. Nitrate is supplied to the sediments by the deposition and mineralization of organic nitrogen, nitrification, and diffusion of nitrate from the water column (Dortch & Gerald, 1995).

Even assuming that long-term nitrogen removal is due solely by denitrification, explaining the retention of nitrogen in wetland is a complex task because it is controlled by a number of factors such as temperature, seasonal variation, precipitation, nitrogen load, wetland area and soil quality (Mitsch & Gosselink, 2000; Baker, 1992; Gale *et al.*, 1993; Arheimer & Wittgren, 1994; Gumbricht, 1993). In particular, nitrogen removal is related to temperature, as increasing temperature accelerates all metabolic processes, including denitrification and biological uptake. The removal is also related to hydrology, as a longer residence time allows more nitrogen to be removed from the water phase (Arheimer & Wittgren, 2002).

In general, it is often assumed that wetland retention of nitrogen, mainly in terms of denitrification, can be modelled as a function of wetland area and nitrogen load (Byström, 1998; Dortch & Gerald, 1995). However, the effectiveness of wetlands as nutrient sinks may vary both spatially and temporally due to their hydrological regime and vegetation cover (Howard-Williams, 1985; Bowden, 1987). In estimating nitrogen retention in wetlands, it is possible to choose either the model presented by Dortch & Gerald (1995) or the approach by Kadlec & Knight (1996). The two models are slightly different, in particular regarding the variable identifying the wetland's removal rate. In the first case we need to estimate the nitrogen removal rate *k*, while in the latter the nitrogen areal removal rate $k_A$ is needed.

Following the Dortch & Gerald (1995) approach, the denitrification process is described as a first order removal rate where the loss of nitrate through denitrification can be used to estimate total nitrogen loss. In this process $k_{TN}$ is the first order removal rate for total nitrogen:

$$k_{TN} = \frac{C_{NO_3-N}}{C_{TN}} k_{dn} \qquad [3.29]$$

with $C_{NO_3\text{-}N}$ as mean nitrate concentration (mg $l^{-1}$); $C_{TN}$ is mean total nitrogen concentration (mg $l^{-1}$); and $k_{dn}$ represents the specific denitrification rate (per day).

This expression requires a specific estimate for the denitrifcation rate, together with the value expressing the proportion between nitrate and total nitrogen concentration. While the former value can be taken from studies of similar wetlands in the same region, the latter is more site-specific and requires field measurements. For this reason the Dortch & Gerald's (1995) approach, equation [3.20], should be applied to a single specific wetland as it would lose accuracy in predicting nitrogen retention at a regional scale. Moreover, a site-specific application of this methodology is also required when including the true hydrological retention time, equation [3.22], which is a function of width, length and depth of the wetland under study.

Several studies on many wetlands around the world report values for the denitrification rate $k_{dn}$. Palmeri & Trepel (2002) estimate a denitrification rate of 0.12 per day for a surface flow wetland in Northern Germany. Graetz *et al.* (1980) found that $k_{dn}$ varied from 0.04 to 0.192 per day for 15 wetlands in Florida. Dortch & Gerald (1995) applied a denitrification rate value of 0.2 per day to predict nitrogen retention to a riparian wetland in the United States. However, due to the lack of a universally applicable value for denitrification and removal rate of nitrogen, Dortch & Gerald (1995) report that denitrification rates should be normally within the range of 0.04-0.19 per day. They also suggest a removal rate for total nitrogen within a range of $0.05 \leq k_{TN} \leq 0.30$ per day and a default value of $k_{TN} = 0.14$. In Lake Victoria, however, a lower value of $k_{TN} = 0.02$ per day has been estimated for the Nakivubo wastewater treatment wetland in Uganda (Kansiime & Nalubega, 1999).

The Kadlec & Knight's (1996) approach requires a value for the areal removal rate of total nitrogen ($k_A^{TN}$). The literature provides useful values for nitrogen areal removal rate from many field studies. Arheimer & Wittgren (2002) calculate $k_A^{TN}$ values for several wetlands in Sweden ranging from 0.0002 to 0.012 (m day$^{-1}$ °C$^{-1}$). Given their high variability, a single coefficient was then obtained through simultaneous calibration, yielding a regional value of 0.0023 (m day$^{-1}$ °C$^{-1}$). This needs to be multiplied by the water temperature expressed in C°. As a general guideline, Kadlec & Knight (1996) suggest areal removal rate values for wastewater wetlands ranging from 22 (m yr$^{-1}$), with prevalence of surface flow, to 27 (m yr$^{-1}$) with subsurface flow.

## 3.5. Phosphorus retention in wetlands

Phosphorus dynamics in wetland ecosystems are much more complex than nitrogen. Phosphorus can exist in water in dissolved, particulate, organic, and inorganic forms, which are affected by biotrasformation, sorption, settling, sedimentation, mineralization, and hydrolysis. However, unlike the denitrification process, microbial activity does not result in a net loss of total phosphorus from the system (Dortch & Gerald, 1995). Most research has shown that long-term storage of phosphorus is dependent primarily on inorganic sediment adsorption and accretion, while uptake by microorganisms and vegetation is only a short-term sink (Richardson, 1985; Richardson & Craft, 1993; Craft & Richardson 1993a; Craft & Richardson, 1993b).

It has also been suggested that natural wetlands are not particularly effective as phosphorus sinks compared to lakes and ponds, due to the fact that macrophytes uptake of phosphorus from sediments is later released into the water column through litter pathways of dead biomass (Prenki *et al.*, 1978). In fact, Wang & Mitsch (2000) showed that the presence of macrophytes may increase total phosphorus in the water column during the non-growing season. Even though most of the phosphorus obtained from sediments by macrophytes is reincorporated into the sediments and does not leave the wetland immediately, they predicted that total phosphorus retention would increase by 5% if macrophytes were absent from the wetland. It follows that harvesting macrophytes has the potential to enhance long-term phosphorus removal in wetlands eliminating the "phosphorus pumping effect" of macrophytes.

Regardless of the presence of macrophytes, natural or old constructed wetlands are much less effective as phosphorus sinks than newly created wetlands. The latter, with low soil P content, can effectively remove phosphorus for a while. This is explained by the fact that, while total nitrogen retention will remain practically constant with time, total phosphorus retention decreases over time as the sediments become saturated until the sorption of phosphorus in the upper sediment horizons reaches equilibrium (Nichols, 1980; Richardson, 1985; Hammer & Kadlec, 1983). After the sediments reach the saturated equilibrium, the removal rate becomes relatively constant with time and is proportional to the sediment burial rate. Research suggests that in old constructed or natural wetlands, permanent storage of total phosphorus is below 1 g $m^{-2}$ $yr^{-1}$ and usually averages 0.5 g $m^{-2}$ $yr^{-1}$ (Nichols, 1983; Richardson, 1985; Richardson & Marshall, 1986; Richardson & Craft, 1993; Craft & Richardson, 1993b). This loading threshold suggests that the process that controls phosphorus retention capacity for wetlands is saturated at loadings above 1 g $m^{-2}$ $yr^{-1}$ once short-term uptake processes are saturated. Therefore, because of this "aging phenomena", the threshold of 1 g $m^{-2}$ $yr^{-1}$ is usually considered as the long-term wetlands storage capacity for total phosphorus. Then, phosphorus loadings into natural or old constructed wetlands above 1 g $m^{-2}$ $yr^{-1}$ will result in a significant increase in phosphorus concentrations in outflows above the baseline, which is usually around 40 μg $l^{-1}$(Richardson *et al.*, 1997). As explained above, in the presence of macrophytes this long-term wetlands storage capacity can be further increased at a higher threshold if machrophytes are periodically harvested.

Adsorption and retention of phosphorus in freshwater wetland soil is controlled not only by the grade of phosphorus saturation of the soil or by the biological uptake, but also by chemical reactions (Lindsay, 1979; Faulkner & Richardson, 1989; Richardson & Vaithiyanathan, 1995). Faulkner & Richardson (1989) reviewed the effect of the interaction of redox potential pH, *Fe*, *Al*, *Ca* minerals and the amount of soil phosphorus in controlling the mobility of phosphorus in wetland environments. In acid soils, inorganic phosphorus is absorbed on hydrous oxides of *Fe* and *Al* and may precipitate as insoluble Fe-phosphates (*Fe-P*) and *Al-P*. Precipitation as insoluble *Ca-P* is the dominant transformation at pH's greater than 7.0 (Qualls & Richardson, 1995; Richardson & Vaithiyanathan, 1995). Insoluble forms of phosphorus are not available for biological uptake by macrophytes and are independent of the grade of phosphorus saturation in wetlands soil. Because of these precipitation processes, it has been shown that long-term phosphorus removal can take place above the normal wetland storage capacity if a continual source of *Ca*, *Fe* or *Al* is available in the wetland (Cooke *et al.*, 1992).

As for nitrogen, we can apply the method proposed by Dortch & Gerald (1995) to estimate phosphorus outflow from a wetland, given a certain inflow. Using this methodology, we need to estimate the wetland retention coefficient $k_{TP}$ for total phosphorus. Recent modelling efforts have shown that removal rates in wetlands can be approximated by a first order rate equation relating surface water total phosphorus removal to soil phosphorus accretion rates (Kadlec, 1994; Walker, 1995). An alternative is to compute the net removal rate coefficient $k_{TP}$ by using steady-state coupled total phosphorus mass balance equations for the water column and active layer of the sediment bed, considering settling and resuspension of particulate phosphorus, burial of total phosphorus, and diffusion of dissolved phosphorus between the water column and the bed. This latter procedure has been suggested by Dortch & Gerald (1995) through the following expression:

$$k_{TP} = \frac{V_n}{H} \frac{\left(V_s f_{pw} + V_d f_{dw}\right)}{\left(V_s + V_d f_{dp} \frac{\rho_b}{S}\right)} \qquad [3.30]$$

where $V_n$ is the net settling rate of total suspended solids (m day$^{-1}$); $V_s$ is the settling velocity of total suspended solids (m day$^{-1}$); $V_d$ is the mass transfer velocity (m day$^{-1}$) across the sediment-water interface resulting from diffusion of dissolved phosphorus; $H$ is the mean water depth of the wetland (m); $f_{pw}$ is the fraction of particulate to total phosphorus in the water column; $f_{dw}$ is the fraction of dissolved to total phosphorus in the water column; $f_{dp}$ is the fraction of phosphorus in the sediment pore water to total phosphorus in the bed layer; $\rho_b$ is the active sediment layer bulk density (g l$^{-1}$); and $S$ is the suspended solids concentration (g l$^{-1}$). Following this procedure they found a wetland-depth dependent removal rate value ($k_{TP}^{h}$) of 0.00042 per day for the Cache River wetlands in Arkansas. This value can then be used as a reference $k_{TP}$ value by multiplying it for the average wetland depth in meters. In Lake Victoria, Kansiime & Nalubega (1999) provide a value of $k_{TP}$ = 0.01 per day which can be used as a reference for other wetlands in this region.

On the other hand, the method proposed by Kadlec & Knight (1996) requires the estimation of the areal removal rate ($k_A^{TP}$) for total phosphorus. Kadlec (1994) found values of $k_A^{TP}$ varying from 2.4 to 23.7 (m yr$^{-1}$), with an average value of 11.6, for a group of 53 wetlands around the United States. This would suggest that a value around $k_A^{TP} = 12$ (m yr$^{-1}$) for both surface and subsurface-flow wetlands could be an appropriate approximation for the areal removal rate constant (Kadlec & Knight, 1996).

## 3.6. Discussion

The important role played by wetlands in supporting primary production and providing many environmental services, must be seen in an ecological and socio-economic context to provide an adequate support to decision makers. In our study, despite their nature as multifunctional systems, we limit our analysis to the nutrient buffering function of wetlands. Among the available methodologies, it is possible to choose between a model that requires several site-specific wetland characteristics and one that requires only a general wetland characteristic variable, such as extent of wetland area. The latter can therefore be applied at a larger scale, allowing for the estimation of the average marginal value of wetlands in a region that has a number of separated swamps.

For a site-specific approach, the model presented by Dortch & Gerald (1995) under longitudinal gradient conditions, equation [3.20], may be used. This is summarized by the equations:

$$L_{out}^{TN} = L_{in}^{TN} \exp\left( -k_{TN}\left[ 0.84\frac{d_W * A_W}{Q_W}\left( 1 - e^{-0.59\frac{l_W}{v_W}} \right) \right] \right) \qquad [3.31]$$

and,

$$L_{out}^{TP} = L_{in}^{TP} \exp\left( -k_{TP}\left[ 0.84\frac{d_W * A_W}{Q_W}\left( 1 - e^{-0.59\frac{l_W}{v_W}} \right) \right] \right) \qquad [3.32]$$

for nitrogen and phosphorus retention respectively.

For a regional approach the option is to adopt the model by Kadlec & Knight (1996). Then, without considering background concentration, the general expression [3.27] becomes:

$$L_{out}^{TN} = L_{in}^{TN} \exp\left( \frac{-k_A^{TN} A_W}{Q_{in}} \right) \qquad [3.33]$$

and,

$$L_{out}^{TP} = L_{in}^{TP} \exp\left( \frac{-k_A^{TP} A_W}{Q_{in}} \right) \qquad [3.34]$$

for nitrogen and phosphorus retention respectively. Either of these two approaches requires the estimation of the removal rate constant $k$ and areal removal rate constant $k_A$ respectively for both nitrogen and phosphorus.

A third possible alternative to these two mass-balance models is to adopt a statistical based predictive model using econometric analysis. Several linear and non-linear models are available in the literature. Nevertheless, this approach is data intensive. However, while for nitrogen retention it requires site-specific data, for the case of phosphorus there is more room for the estimation of equations that result to be more regionally applicable. Its justification resides in the fact that the effectiveness of wetlands as nitrogen sinks may vary both spatially and temporally due to their hydrological regime and vegetation cover more than it does for phosphorus. This is because the denitrification process, which plays the main rule in nitrogen retention by wetlands, is particular sensitive to these variables.

The primary focus of the thesis is the role played by wetlands in moderating the effect of land based pollution and hence on eutrophication and fishery yields in Lake Victoria. To address this problem, in chapter 7 we model the impact of nutrient loading on fisheries assuming the watershed drains into wetlands before entering the lake. In doing so, we will use the Kadlec & Knight's mass balance approach to predict nutrient retention.

# Chapter 4

# THE LAKE COMPONENT

## 4.1. Introduction

In studying the impact of eutrophication on fisheries using a watershed approach, it is necessary to link nutrient loadings to lake's nutrients concentration. The estimation of nutrient concentrations requires understanding of the chemical, physical and hydrological factors that regulate nutrient dynamics in lakes. The theoretical framework for such dynamics is provided by the vast literature on limnological models reviewed in this chapter.

An overview of the basic fishery models (§ 4.3) will then introduce the problem of accounting for an environmental variable (§ 4.4). Particular emphasis is given to the Gordon-Schaefer logistic model (§ 4.3.1) and the ecosystem model Ecopath (§ 4.3.2).

Since one candidate for the environmental variable in our fishery model is phytoplankton population density, we first present several empirical studies that use econometric analysis to study the relationship between nutrients concentration and phytoplankton biomass (§ 4.5.1). The resulting environmental function can then be incorporated in a bioeconomic Gordon-Schaefer fishery model. On the other hand, if an ecosystem fishery model is used to estimate the marginal impact of eutrophication, phytoplankton species composition may be used as environmental variable. In this latter case, the change in relative abundance of competing species within the algae community can be predicted through a phytoplankton competition model. This approach, however, must include a phytoplankton growth function, which explains most of the competing interactions among algae species. The theoretical basis of both phytoplankton growth models (§ 4.5.2) and algae competition models (§ 4.5.3) are explored.

Finally, the theory of trophic state indexes is also reviewed (§ 4.6) in order to consider their potential use as alternative environmental variables, particularly in Gordon-Schaefer-type fishery models.

## 4.2. Limnological models for lakes

Lakes are open flow-through systems whose content of a given element is determined by the dynamic balance between supply through the inflows, loss through the outflows, and internal sources and sinks (Andersen, 1997). In eutrophic lakes, controlling nutrient inputs and predicting responses to changes in nutrients loading are the major concerns. Prediction is commonly based on a family of phosphorus mass balance models directly descending from the pioneering work of Vollenweider (1968b; 1969). Based on the broad consensus that phosphorus is the mineral nutrient most likely to limit primary production in lakes (Schindler, 1977, 1978), since Vollenweider's initial work many models have been developed that describe the mass balance of phosphorus in lakes (Dillon & Rigler, 1974a; Vollenweider, 1975, 1976; Chapra, 1975; Nurnberg, 1984).

In this context, Carpenter *et al.* (1999a, 1999b) describe phosphorus limnological dynamics for the particular case of shallow lakes, where there is a consistent nutrient recycling activity from the bottom sediments, according to the following set of equations:

$$P_{t+1} = (1 - s - h)P_t + L_t + rM_t f(P_t) \qquad [4.1]$$

with,

$$M_{t+1} = (1 - b)M_t + sP_t - rM_t f(P_t) \qquad [4.2]$$

$$f(P_t) = \frac{P_t^q}{(m^q + P_t^q)} \qquad [4.3]$$

where *P* is the amount of phosphorus (mass or concentration) in the water, *L* the annual phosphorus input (mass or concentration per unit time), *M* the amount of phosphorus in the mud, *r* the maximum recycling rate of phosphorus (mass or concentration per unit time), *b* the proportion of phosphorus in the mud lost at each time step to permanent burial, *s* the proportion of phosphorus sedimented each year, and *h* the proportion of phosphorus flushed each year. Sediments are assumed to be the major source of recycled phosphorus ignoring the recycling by consumers. The overall recycling rate is assumed to be a sigmoid function, $f(P_t)$, of phosphorus concentration where *m* is the value of *P* at which recycling reaches half of the maximum rate. The exponent $q \geq 2$ affects the steepness of the sigmoid curve. Empirical limnological models provide a rough estimate of $q$ ranging from approximately 20 for shallow warm lakes, to about 2 for deep cold lakes (Janssen, 2001).

On the other hand, the original model by Vollenweider (1968b, 1969) is more generally applicable to pollutants other than phosphorus. He adopted the assumption that concentrations in the lake and in the outflowing water are equal. This assumption is known in the engineering literature as the continuously stirred tank reactor (Reckhow & Chapra, 1983) and is equivalent to consider the lake's waters as completely mixed. Under this assumption the total lake content of the i*th* pollutant, the product of the lake volume ($V$) and the i*th* concentration in the lake ($C_i$), is the sum of a) the total i*th* loading rate, which is the product of the total water flow into the lake ($Q_{in}$) and the average concentration of the i*th* pollutant in the inflowing waters ($C_{in}^i$); b) the i*th* pollutant lost through the lake's hydraulic discharge at a rate proportional to the rate of superficial outflow of water ($Q_{out}$) and the i*th* concentration in the lake ($C_i$); and c) net losses or gains due to internal processes ($S_i$):

$$\frac{dVC_i}{dt} = \dot{V}C_i + V\dot{C}_i = Q_{in}C_{in}^i - Q_{out}C_i - VS_i \qquad [4.4]$$

In this expression a change in either lake volume or lake concentration leads to a change in lake content. The change in lake volume is equal to the difference between total inflow and outflow. It is often assumed the maintenance of a constant lake volume which would require equal in- and outflows. In this case, the mass-balance equation, in terms of concentration, can be simplified assuming that the change in concentration of the pollutant in a lake is equal to the supply per unit volume minus loss through sedimentation and loss by outflow:

$$\frac{dC_i}{dt} = \frac{L_i(t)}{V} - \rho C_i - s_i C_i \qquad [4.5]$$

with $L_i$ representing the total input of the i*th* pollutant; $\rho$ is hydraulic flushing rate per unit of water volume; and $s$ is the sedimentation rate. The internal processes may vary according to the type of pollutant to be considered. Loss due to sedimentation and burial in the mud at the bottom of the water body is often the most important in-lake nutrients process. However, for the specific case of nitrogen, the loss due to denitrification should also be included. Expression [4.5] can be written in more detail as follows:

$$\frac{dC_i}{dt} = \frac{L_i(t)}{V} - \frac{Q_{out}}{V}C_i - \frac{v_i}{H}C_i \tag{4.6}$$

where $C_i$ is the mean concentration (g m$^{-3}$) of i*th* pollutant in the lake; $L_i$ represents the total input of $i$ (g day$^{-1}$); $V$ is the lake volume (m$^3$); $Q_{out}$ is the hydraulic superficial outflow from the lake (m$^3$ day$^{-1}$); $H$ is the mean depth of the water body (m); and $v_i$ is the settling velocity rate (m day$^{-1}$), which can be inferred by the sediment deposition rate according to the formula (Scheren, 1995):

$$v_i = \frac{r_S S_i}{C_i} \tag{4.7}$$

where $r_s$ is the dry weight sedimentation rate (g m$^{-2}$ day$^{-1}$) and $S_i$ is content (mg g$^{-1}$) of the i*th* pollutant in the deep water sediments.

Assuming constant $V$, $\rho$ and $s$ and with initial boundary condition $C_i = C^i_{t=0}$, the mass-balance equation [4.5] can be solved analytically by integration over time interval $t_0 = 0$ to $t$ as follows (Vollenweider, 1968b, 1969):

$$C_i = \frac{L_i}{V(\rho + s_i)}\left(1 - \exp[-(\rho + s_i)t]\right) + C^i_{t=0}\exp[-(\rho + s_i)t] \tag{4.8}$$

which is equivalent to the integration of equation [4.6]:

$$C_i = \frac{L_i}{\left(Q_{out} + \frac{v_i}{H}\right)}\left\{1 - \exp\left[-\left(\frac{Q_{out}}{V} + \frac{v_i}{H}\right)t\right]\right\} + C^i_{t=0}\exp\left[-\left(\frac{Q_{out}}{V} + \frac{v_i}{H}\right)t\right] \tag{4.9}$$

The steady state conditions, which follow by determining the limit for $C_i$ when $t \to \infty$, are respectively:

$$C_i^* = \frac{L_i}{\rho + s_i} \frac{1}{V} \tag{4.10}$$

and,

$$C_i^* = \frac{L_i}{\left( Q_{out} + \frac{v_i}{H} V \right)} \tag{4.11}$$

Under steady-state conditions equations [4.10] and [4.11] can be used to represent a description of overall nutrient processes in a lake and then to evaluate or predict changes in the nutrient balance of a water body as a response to a step input, when the input changes from one constant level to another load level $L_i$.

The basic equation, [4.5] or [4.6], of the Vollenweider's (1968b; 1969) model implies that, as each new concentration of the *ith* pollutant enters the lake and is immediately mixed throughout the lake, a new concentration is produced after a fraction leaves through the outlet and a fraction sediments to the bottom. Both are functions of the new, slightly changed concentration (Welch & Lindell, 1996). The steady-state equation shows that over the long term the lake would reach equilibrium at the given loading. If the loading is changed, then time will be required for adjusting to the new loading. If a first order rate reaction is assumed, the times to 50% and 90% of equilibrium will be respectively (Vollenweider, 1969):

$$t_{50} = \frac{\ln 2}{s + \rho} \tag{4.12}$$

$$t_{90} = \frac{\ln 10}{s + \rho} \tag{4.13}$$

Vollenweider's (1968b, 1969) model and models that derive from it require the estimation of the sedimentation parameter (*s*), which may be difficult to obtain. The alternative, however, is to use a dimensionless retention coefficient $R_i$ (Vollenweider & Dillon, 1974; Dillon & Rigler, 1974a), which is the fraction of the loading of the *ith* pollutant that is not lost via outflow. In this case, as reported in Jørgensen (1980), equation [4.8] can be rewritten as:

$$C_i = \frac{L_i(1-R_i)}{\rho V} - \left(1 - \frac{\rho V C_{t=0}^i}{L_i(1-R_i)}\right)\exp\left[-\frac{\rho}{1-R_i}t\right] \qquad [4.14]$$

and the corresponding steady state condition [4.10] reported in Vollenweider & Dillon (1974) as:

$$C_i^* = \frac{L_i(1-R_i)}{\rho}\frac{1}{V} \qquad [4.15]$$

or more simply:

$$C_i^* = \frac{L_i(1-R_i)}{Q_{out}} \qquad [4.16]$$

which is equivalent to the expression in areal terms used by Dillon & Rigler (1975):

$$C_i^* = \frac{L_A^i(1-R_i)}{\rho_A} \qquad [4.17]$$

where $L_A{}^i$ is annual *ith* nutrient loading per unit of lake surface area (mg m$^{-2}$ yr$^{-1}$); and $\rho_A$ is the areal hydraulic outflow (i.e. the outflow of the lake divided by its surface area) expressed in m yr$^{-1}$ as the ratio of $Q_{out}$ (m$^3$ yr$^{-1}$) over $A$ (m$^2$).

Equations [4.15] and [4.17] are used when $R_i$ is provided from the literature and based on empirical relationships between retention coefficient and hydraulic conditions. One of these relationships has been estimated by Kirchner & Dillon (1975) for predicting $R_i$ through multiple regression analysis based on several lake studies in Scandinavia:

$$R_i = 0.426\exp(-0.271\rho_A) + 0.574\exp(-0.00949\rho_A) \qquad [4.18]$$

This equation has been estimated specifically for phosphorus, but it can also be applied to nitrogen by adjusting the predicted $R_i$ to a 16% lower value (Jørgensen, 1980).

Larsen & Mercier (1976), basing their models on data from lakes in which the initial phosphorus concentration was lower than 25 mg m$^{-3}$, provide three empirical formulations that relate the retention coefficient for phosphorus $R_P$ to hydraulic variables as follows:

$$R_P = 0.86 - 0.143 \ln \rho_A \qquad [4.19]$$

$$R_P = 0.482 - 0.112 \ln \rho \qquad [4.20]$$

$$R_P = \frac{1}{1+\rho^{0.5}} \qquad [4.21]$$

From equation [4.21] the steady-state condition for phosphorus concentration can be written as:

$$C_P^* = \frac{L_P}{\rho+\rho^{0.5}} \frac{1}{V} \qquad [4.22]$$

Another relationship which circumvents the problem of estimating sedimentation rate is provided by Ahlgren *et al.* (1988), who show how the relationship between sedimentation rate ($s$), flushing rate ($\rho$) and inflow concentration of phosphorus, that was previously estimated by Canfield & Bachmann (1981) as:

$$s_P = 0.129 \left( \rho \frac{L_A^P}{q_A} \right)^{0.549} \qquad [4.23]$$

it leads to the following steady-state condition (Ahlgren *et al.*, 1988):

$$C_P^* = \frac{\left(L_A^P / q_A\right)}{1 + 0.129 \rho^{-0.451} \left( \frac{L_A^P}{q_A} \right)} \qquad [4.24]$$

where $q_A$ is the areal water inflow (m yr$^{-1}$), given by the product of total water inflow $Q_{in}$ (m$^3$ yr$^{-1}$) over lake area $A$ (m$^2$).

Alternatively, Welch *et al.* (1986) presented an expression where sedimentation rate is a function of flushing rate only:

$$s_P = \rho^{0.78} \qquad [4.25]$$

Using least squares regression from 47 north temperate lakes, Reckhow (1979) found that the apparent settling velocity for phosphorus could be fitted using a weak function of the areal water loading. Then, he presented the following steady-state equation, which is only applicable to the north temperate climatic zone:

$$C_P^* = \frac{L_A^P}{11.6 + 1.2q_A} \qquad [4.26]$$

Following a different approach, Chapra (1975) gave the following equation for the phosphorus retention coefficient:

$$R_P = \frac{v_P}{v_P + \rho_A} \qquad [4.27]$$

which leads to the steady-state condition for phosphorus concentration:

$$C_P^* = \frac{L_A^P}{v_P + \rho_A} \qquad [4.28]$$

where $v_P$ is the apparent settling velocity of total phosphorus, which, according to Chapra (1975) is 16 m $yr^{-1}$, while following Dillon & Kirchner (1975) is 13.2 m $yr^{-1}$.

From the equations presented so far, it must be noticed that the derivation of the mass-balance model for nutrient concentration assumes steady-state conditions. This implies that there is no accumulation or change in concentration in the water column. Indeed, steady state can be assumed on annual basis if morphometry, climate, and nutrient supply do not vary significantly from year to year. However, lakes undergoing cultural eutrophication are unlikely to be at steady state. One way to correct for accumulation is to leave it in the nutrient budget model, upon which the relationship between loading and mean annual concentration is based. Then, referring in this case to equation [4.28], Chapra & Tarapchak (1976) suggested the following expression:

$$C_P = \frac{L_A^P - H\dfrac{dC_P}{dt}}{v_P + \rho_A} \qquad [4.29]$$

where $H$ represents the mean depth of the lake and $dC_P/dt$ is the change in concentration with respect to time. An alternative to including the accumulation term is to estimate confidence limits for the predicted steady-state equation. If limits are small enough, it can be inferred that the variability introduced by non-steady conditions does not seriously affect the general predictive capability of the steady-state equation (Chapra & Tarapchak, 1976).

## 4.3. Fishery models

Gordon's (1954) classical study, which developed the theory of optimum utilisation of fishery resources and the reasons for its overexploitation, is considered the base of the economic theory of fish resource utilization. Since then, many studies have attempted to refine and extend the economic theory of fishing (Scott, 1955; Schaefer, 1957; Smith, 1968, 1969; Plourde, 1970; Clark, 1973a, 1973b, 1980; Brown, 1974; Clark & Munro, 1975). A common feature among them is that they are all based on the theoretical relationships between species stock levels, growth functions, fishing effort and fish catch. These bioeconomic models look at the biomass of fish stock as an indicator of population size and are often referred to as biomass dynamics models.

Single-species biomass dynamics models, which deal mainly with changes in aggregate biomass of the fish stock, represent the simplest models used in fisheries (Hilborn & Walters, 1992). However, this is not the only approach available to fishery economics and management. Other models may be used to extend this basic approach. Age structure models (Beverton & Holt, 1957; Ricker, 1975), for instance, have been introduced in order to consider the age structure of fish population, which is ignored in the basic biomass models. Multispecies models (Larkin & Gazey, 1982; Chaudhuri, 1986; Sainsbury, 1988) may also be used to account for the interactions between species, in form of competition or predator-prey relationships, according to the basic theory formulated by Lotka (1925, 1956) and Volterra (1931). These models are particularly useful in studying mixed-stock fisheries where several stocks are harvested jointly (i.e. Paulik *et al.*, 1967;

Hilborn, 1976). The multispecies approach has been brought even further by the ecosystem models (Laevastu *et al.*, 1982; Polovina, 1984; Christensen & Pauly, 1992), which consider species interactions in rigorous quantitative terms by modelling interactions in the whole ecosystem. On the other hand, aggregate production models (Ralston & Polovina, 1982), which aggregate species and treat the aggregates as stocks to be analysed via production models, have been used as an alternative to multispecies production models.

### 4.3.1. Biomass dynamics models

The first widely used biomass dynamics model was formulated by Schaefer (1954), based on early work by Graham (1935). The model is characterized by two components, the population growth function $F(X)$ and the production function $Y(E,X)$ that can be written respectively as:

$$F(X) = rX\left(1 - \frac{X}{K}\right) \qquad [4.30]$$

$$Y = qEX \qquad [4.31]$$

where $X$ represents the stock biomass; $r$ is the intrinsic growth rate; $K$ is the carrying capacity; $Y$ is the fishery yield; $E$ represents fishing effort; and $q$ is the catchability coefficient. The basic fishery model is then derived by the combination of the growth and production functions, where the change in the size of the stock is expressed as the difference between the natural growth and harvest:

$$\frac{dX}{dt} = rX\left(1 - \frac{X}{K}\right) - qEX \qquad [4.32]$$

which can also be expressed in discrete time dimension using a simple difference equation as shown by Walters & Hilborn (1976):

$$X_{t+1} = X_t + rX_t\left(1 - \frac{X_t}{K}\right) - qE_tX_t \qquad [4.33]$$

Following Gordon (1954) and Schaefer (1954; 1957) this leads to the sustainable yield function of the so-called Gordon-Schaefer fishery model:

$$Y = qEX = qKE\left(1 - \frac{qE}{r}\right) \qquad [4.34]$$

This basic model allows for as many variations as the possible combinations of different functional forms for the growth and production function. The logistic growth function first applied to fisheries by Schaefer (1954) can be replaced by any of the following functional forms:

$$F(X) = rX^{\alpha}\left(1 - \frac{X}{K}\right) \quad ;\alpha > 1 \qquad [4.35]$$

$$F(X) = X\left|e^{-r(1-X/K)} - 1\right| \qquad [4.36]$$

$$F(X) = rX\ln(K/X) \qquad [4.37]$$

$$F(X) = r(X - K_{\min})\left(1 - \frac{X}{K}\right) \quad ;K_{\min} > 0 \qquad [4.38]$$

$$F(X) = rX\left(\frac{X}{K_{\min}} - 1\right)\left(1 - \frac{X}{K}\right) \quad ;K_{\min} > 0 \qquad [4.39]$$

Also the production function can assume different functional forms from the linear model used by Schaefer (1954, 1957). The basic assumption underlying this linear model is that catch per unit effort is proportional to stock biomass level for all levels of effort and stock. This assumption may be unrealistic in cases of saturation of fishing gear or congestion of fishing vessels. In such cases a Cobb-Douglas production function, which exhibits decreasing marginal returns to stock and effort, is preferable:

$$Y = qE^{\alpha}X^{\beta} \qquad [4.40]$$

where $\alpha$ and $\beta$ are positive constants depicting the marginal returns to scale (Clark, 1976; Conrad & Clark, 1987).

The multispecies extension of the Gordon-Schaefer model applies the Lotka-Volterra theory to the logistic growth function for both competitive interactions and predator-prey interactions. The competitive interaction among species is considered in the form of an interspecific competition term included in the growth function equation of the *ith* species as follows (Hilborn & Walters, 1992):

$$F(X_i) = r_i X_i \left( 1 - \frac{X_i}{K_i} - \sum_{j \neq i} \alpha_{ij} X_j \right) \qquad [4.41]$$

However, other authors propose a slightly different functional form considering the ratio between the interspecific competition coefficient $\alpha_{ij}$ and the carrying capacity of the *ith* species, and including an intraspecific competition coefficient $\alpha_{ii}$ (Jørgensen, 1994; Begon *et al.*, 1996):

$$F(X_i) = r_i X_i \left( 1 - \alpha_{ii} \frac{X_i}{K_i} - \frac{\sum_{j \neq i} \alpha_{ij} X_j}{K_i} \right) \qquad [4.42]$$

Predator-prey interactions have been considered in a growth function for a fishery model by Clark (1976), following the dynamical studies by Gause (1935), and identified by the following growth function equation for the *ith* prey species:

$$F(X_i) = r_i X_i \left( 1 - \frac{X_i}{K_i} \right) - \sum_{k \neq i} \beta_{ik} X_i X_k \qquad [4.43]$$

where $\beta_{ik}$ represents the predation coefficient of the k*th* predator on the *ith* prey species. Following this model, Chaudhuri (1986) shows mathematically that combined harvesting of two interacting fish species may drive one species to extinction, while the fishery continues to maintain a bionomic equilibrium with the support of the other species. Dynamic optimisation of combined harvesting of a two-species fishery is analysed by Chaudhuri (1988).

### 4.3.2. Ecosystem models: "Ecopath" and "Ecosim"

Among the ecosystem models, one methodology used for the construction, parameterization and balancing of steady-state trophic models of aquatic ecosystems is

"Ecopath with Ecosim"[7]. This system is built on an approach initially developed by Polovina (1984), later implemented by Christensen & Pauly (1992) and by the following additions of the dynamic and spatial dimension of the Ecosim (Walters *et al.*, 1997, 2000) and Ecospace (Walters *et al.*, 1999) components respectively.

The Ecopath model bases the parameterization on an assumption of mass balance over an arbitrary period, usually a year, at the end of which the system has the same biomass state as at the beginning. The trophic mass-balance model is defined by the master equation that describes how the production term for each functional group (*i*) within the ecosystem can be split in components (Christensen *et al.*, 2000):

Production = catches + predation mortality + other mortality + biomass accumulation + net migration

or more formally,

$$P_i = Y_i + M_i + M0_i + BA_i + E_i \qquad [4.44]$$

which can be re-expressed as:

$$B_i * (P/B)_i = Y_i + \left[\sum_{j=1}^{n} B_j * (Q/B)_j * DC_{ji}\right] + \left[(P/B)_i * B_i * (1 - EE_i)\right] + BA_i + E_i \qquad [4.45]$$

or in terms of production if (*i*) is utilized within the system:

$$B_i * (P/B)_i * EE_i = Y_i + \left[\sum_{j=1}^{n} B_j * (Q/B)_j * DC_{ji}\right] + BA_i + E_i \qquad [4.46]$$

with,

$$P_i = B_i * (P/B)_i \qquad [4.47]$$

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[7] The Ecopath with Ecosim (EwE) software is available on-line at http://www.ecopath.org. The user's guide by Christensen *et al.* (2000) is also available for download together with some key publications and a list of about two hundred published applications by several authors. Among the key bibliography it must be mentioned: Christensen & Pauly (1992) for an explanation of the Ecopath methodology; Walters *et al.* (1997, 2000) for the basic theory of the Ecosim dynamic component of Ecopath; Walters *et al.* (1999) for an overview of the Ecospace spatial component of the package; Christensen & Walters (2003) for a detailed insight of the new features of EwE, including the quasi-Bayesian re-sampling technique and nutrient limitation; and Pauly *et al.* (2000) for an explanation of the Ecopath's routine Ecoranger, which quantifies the uncertainty associated with Ecopath input.

$$M_i = \sum_{j=1}^{n} Q_{ij} = \sum_{j=1}^{n} Q_j * DC_{ji} = \sum_{j=1}^{n} B_j * (Q/B)_j * DC_{ji} \qquad [4.48]$$

$$M0_i = P_i * (1 - EE_i) \qquad [4.49]$$

where $P_i$ is the total production rate of the group ($i$) during the period in question with the system having $i$ = 1,…,n functional groups; $B_i$ is the biomass of ($i$); $M_i$ is the total predation rate for ($i$); $M0_i$ is the mortality rate not accounted for within the system; $(P/B)_i$ is the production/biomass ratio of ($i$); $EE_i$ is the ecotrophic efficiency which represents the fraction of the production that is consumed within the system, harvested, used for biomass accumulation, migration or export; $Y_i$ is the yield of ($i$), i.e. its catch in weight, with $Y_i = F_i{*}B_i$, where $F$ is the fishing mortality; $B_j$ is the biomass of the consumer or predator ($j$); $(Q/B)_j$ is food consumption per unit biomass for consumer ($j$), and $DC_{ji}$ is the fraction of ($i$) in the diet of ($j$) with $DC_{ij} = 0$ when ($j$) does not eat ($i$).

In an ecosystem without biomass accumulation and external migration, this formulation generates a systems of linear equations, where at equilibrium we have:

$$B_i * (P/B)_i - F_i * B_i - M0_i - \sum_{j=1}^{n} Q_{ij} = 0 \qquad [4.50]$$

From this set of equations it is possible to estimate through parameterisation the biomass of each of the $n$ functional groups in our system at a specific period of time. This requires information on production/biomass ratio ($P_i/B_i$), consumption biomass ratio ($Q_i/B_i$), ecotrophic efficiency ($EE_i$), catch rate ($Y_i$), net migration rate ($E_i$), biomass accumulation rate ($BA_i$), diet composition ($DC_{ji}$) and food assimilation rate for each group.

The Ecopath parameterization method does not rely on solving a full set of linear equations, i.e. there may be fewer equations than there are groups in the system. This is due to a number of algorithms included in the parameterization routine that will try to estimate iteratively as many missing parameters as possible before setting up a set of linear equations. The following loop is carried out until no additional parameters can be estimated (Christensen *et al.*, 2000). The set of linear equations, including groups for which parameters are still missing, is then solved using a generalized inverse method for matrix inversion described by Mackay (1981).

This approach allows the estimation of parameters at one period of time only, without considering any dynamic in the system. Nevertheless, it is possible to re-express a system of linear equations as a system of differential equations which predicts biomass change over time for each *ith* functional group. The previous equation in dynamics terms then becomes (Walters *et al.*, 1997):

$$\frac{dB^i}{dt} = f\left(B_t^i\right) - F_t^i * B_t^i - M0_t^i - \sum_{j=1}^{n} Q_t^{ij} \qquad [4.51]$$

This expression represents a set of equations which can be integrated with fishing mortality $F_i$ (where $F_i$ can be considered as the product of fishing effort by the catchability coefficient) varying in time to provide dynamic biomass predictions of each (*i*), as affected directly by fishing and predation on (*i*) and changes in food available to (*i*), and indirectly by fishing or predation on other pools with which (*i*) interacts.

The difference between the dynamic and the static equation is that in the dynamic model we replace the static $Q_{ij}$ pool-to-pool consumption estimates with functional relationships predicting how consumption will change with changes in the biomass of $B_i$ and $B_j$:

$$\sum_{j=1}^{n} Q_t^{ij} = c_{ij}\left(B_t^i, B_t^j\right) \qquad [4.52]$$

The second difference is that we provide a functional relationship $f(B_i)$ to predict changes in $(P/B)_i$ with biomass $B_i$. This functional relationship varies depending on whether it refers to primary producers:

$$\Delta(P/B)_i = f\left(B_t^i\right) = \frac{r^i B_t^i}{\left(1 + h^i B_t^i\right)} \qquad [4.53]$$

or consumers,

$$\Delta(P/B)_i = f\left(B_t^i\right) = g_i \sum_{j=1}^{n} c_{ji}\left(B_t^i, B_t^j\right) \qquad [4.54]$$

The functional relationship for primary producers [4.53] is a simple saturating production relationship where $r^i$ is the maximum $(P/B)$ that $(i)$ can exhibit when $B_i$ is low and $(r^i/h^i)$ is the maximum net primary production rate for pool $(i)$ when biomass is not limiting to production. In this formulation $r^i$ can be calculated providing an estimate $\eta$ of the ratio of maximum to initial $(P/B)$, with then $\eta^i = r^i/(P/B)^i_{t=0}$. Also $h_i$ can be estimated from the parameters of the static model, which represents the initial conditions of the dynamic model at time $t = 0$, as:

$$h^i = \frac{\left[\dfrac{r^i}{(P/B)^i_{t=0}} - 1\right]}{B^i_{t=0}} \qquad [4.55]$$

The functional relationship referring to consumers [4.54], where the term $g_i$ is the growth efficiency, may represent predator-prey encounter patterns and physiological-behavioural phenomena such as satiation of predators. Several of these functional relationships have been proposed in the ecosystem and fisheries modelling literature for predicting consumption flows. Following the Lotka-Volterra or mass-action modelling assumptions, the functional relationship predicting consumption flows would be:

$$c_{ij}\left(B^i_t, B^j_t\right) = a_{ij} B^i_t B^j_t \qquad [4.56]$$

where $a_{ij}$ represents the instantaneous mortality rate on prey $(i)$ caused by one unit of predator $(j)$ biomass. This catchability interpretation of $a_{ij}$ corresponds in the ecological literature to interpreting $a_{ij}$ as the "rate of effective search" (Holling, 1959) of the consumer, measured per unit consumer biomass. In this case $a_{ij}$, if assumed to be constant, can be estimated directly from the corresponding flow estimate of the model at time $t = 0$:

$$a_{ij} = \frac{Q^{ij}_0}{B^i_0 B^j_0} \qquad [4.57]$$

One of the most serious weaknesses of the mass-action functional relationship in predicting consumption flows is that it does not account for spatial and behavioural limiting mechanisms. In nature predator-prey encounter patterns are seldom random in space, and are most often associated with behavioural or physical mechanisms that limit the rates at which prey become available to encounters to predators.

Such limitations are introduced into the system by viewing each prey pool $B_i$ as having an available component $V_{ij}$ of their biomass to each consumer ($j$) at any moment in time. This available biomass $V_{ij}$ may exchange fairly rapidly with the unavailable biomass according to the exchange equation (Walters *et al.*, 1997):

$$\frac{dV^{ij}}{dt} = v_{ij}\left(B_t^i - V_t^{ij}\right) - v_{ij}V_t^{ij} - a_{ij}V_t^{ij}B_t^j \qquad [4.58]$$

which represents the change in the biomass of ($i$) that is available to predator ($j$). That is, $V_{ij}$ gains biomass from the currently unavailable pool ($B_i - V_{ij}$) at rate $v_{ij}$, biomass returns ($v_{ij}V_{ij}$) to the unavailable state at the same rate, and biomass is removed ($a_{ij}V_{ij}B_j$) from $V_{ij}$ by the consumer at a mass-action encounter rate ($a_{ij}B_j$). Assuming that exchange process between $V$ and $B$ operates on short time scales relative to changes in $B_i$ and $B_j$, then $V_{ij}$ should stay near the equilibrium. It follows that, with $dV/dt = 0$, at any moment in time we have an equilibrium value for $V_{ij}$ which varies with $B_i$ and $B_j$:

$$V_t^{ij} = \frac{v_{ij}B_t^i}{2v_{ij} + a_{ij}B_t^j} \qquad [4.59]$$

under these assumptions the functional relationship predicting consumption flows becomes:

$$c_{ij}\left(B_t^i, B_t^j\right) = a_{ij}V_t^{ij}B_t^j \qquad [4.60]$$

or,

$$\sum_{j=1}^{n} Q_t^{ij} = c_{ij}\left(B_t^i, B_t^j\right) = \frac{a_{ij}v_{ij}B_t^iB_t^j}{2v_{ij} + a_{ij}B_t^j} \qquad [4.61]$$

This allows rewriting the expressions representing the set of differential equations for the system under consideration as:

$$\frac{dB^i}{dt} = g_i\left(\frac{a_{ki}v_{ki}B_t^iB_t^k}{2v_{ki} + a_{ki}B_t^i}\right) - Y_t^i - \left[P_i\left(1 - EE_i\right)\right] - \frac{a_{ij}v_{ij}B_t^iB_t^j}{2v_{ij} + a_{ij}B_t^j} \qquad [4.62]$$

and,

$$\frac{dB^i}{dt} = \frac{r^iB_t^i}{1 + h^iB_t^i} - \left[P_i\left(1 - EE_i\right)\right] - \frac{a_{ij}v_{ij}B_t^iB_t^j}{2v_{ij} + a_{ij}B_t^j} \qquad [4.63]$$

for the i*th* functional group being consumer [4.62] or primary producer [4.63] respectively. These expressions depict the biomass dynamics over time accounting for spatial and behavioural limiting mechanisms.

These biomass estimates are very sensitive to changes in the transfer rate $v_{ij}$ between the available and unavailable components of the biomass of the i*th* prey. The transfer rate is what determines if control is top-down or bottom-up within the system. This allows representing a strategic range of alternative hypotheses about top-down versus bottom-up control of trophic structure and abundance (Hunter & Price, 1992; Matson & Hunter, 1992).

Setting low values for the vulnerability ratios leads to bottom-up control of flow rates from prey to predators. In this case, replacement of depleted biomass from the invulnerable to the vulnerable part of the prey's population will be slow, and the amount that predators consume will be largely determined by the low value of $v_{ij}$, rather than by their own biomass. Then, flow rates are determined by prey biomass only, such that increases in prey productivity will lead to prey biomass increases and then to increased availability to predators. This is the situation when a prey is protected most of the time and becomes available to predators only when it leaves the features that protect it. Here, being caught is a function of the prey's behaviour. Bottom-up control usually leads to unrealistically smooth biomass changes in the prey and predators concerned that usually do not propagate through the other elements of the food web (Christensen *et a*l., 2000).

On the other hand, high values for the vulnerability ratios leads to top-down control and trophic cascade effects (Carpenter & Kitchell, 1993). In this case, the rapid replenishment of vulnerable biomass depleted by predator will rapidly drain the invulnerable part of the biomass. The amount of prey consumed by the predator is the product of predator and prey biomass and then the predators co-determine how much of the prey is consumed. This situation may occur where the prey has no refuge, and is thus always caught upon encounter with predators. Top-down control, also known as Lotka-Volterra dynamics, easily leads to rapid oscillations of prey and predator biomasses and/or unpredictable behaviour.

## 4.4. Using environmental variables in fishery models

Environmental factors play a significant role in determining the productivity of freshwater fisheries. Many of them are natural processes, ranging from nutrient enrichment and variation in temperature and precipitation, to patchy distributions of food organisms. Nevertheless, current environmental concerns focus on the effects of human activities on fish habitats. Despite many evidences of the serious consequences that environmental factors may bear on fishery, up to now to our knowledge, few fishery studies have tried to include environmental variables.

Among those few, Ikeda & Yokoi (1980) incorporate an environmental variable in a biomass dynamic model in order to study the behaviour of the system as a function of nutrient enrichment and fish catch. They proposed a mathematical model of an ecosystem which comprises nutrient, plankton and two types of fish, small and large. The system of three population variables is represented by three ordinary differential equations as follows:

$$\dot{X} = r\left(1 - \frac{X}{K[n(p)]}\right)X - \alpha XY \qquad [4.64]$$

$$\dot{Y} = (\alpha k_1 X - d_1(p) - e_1 Y)Y - \beta YZ - F_1 Y \qquad [4.65]$$

$$\dot{Z} = (\beta k_2 X - d_2(p) - e_2 Z)Z - F_2 Z \qquad [4.66]$$

where $X$, $Y$, and $Z$ are respectively the biomasses of plankton, small fish and large fish; $r$ is the plankton growth rate; $K$ represents the plankton carrying capacity; $n$ the amount of nutrient as a function of nutrient loading $p$; $\alpha$ is the grazing coefficient of plankton by small fish; $\beta$ is the grazing coefficient of small fish by large fish; $d_1$ and $d_2$ are the natural death rates of small and large fish respectively; $e_1$ and $e_2$ represent the self-restraint coefficients of small and large fish; $k_1$ and $k_2$ are the conversion rates of small and large fish; $F_1$ and $F_2$ are the catching rates for $Y$ and $Z$ respectively; and given function, $n(p)$, $K[n(p)]$, $d_1(p)$ and $d_2(p)$ respectively as:

$$n(p) = bp \qquad [4.67]$$

$$K[n(p)] = K_0 + (K_m - K_0)(1 - \exp[-a_0(p - p_0)]) \qquad [4.68]$$

$$d_1(p) = d_{10} \exp[a_1(p - p_0)] \qquad [4.69]$$

$$d_2(p) = d_{20} \exp[a_2(p - p_0)] \qquad [4.70]$$

where $a_0$, $a_1$, $a_2$, and $b$ are positive constants; $K_0$, $d_{10}$, and $d_{20}$ are respectively the values of $K$, $d_1$, and $d_2$ at $p = p_0$; and $K_m$ is the upper limit of $K$ or maximum attainable carrying capacity. Applying their model to an inland sea in Japan, Ikeda & Yokoi (1980) were finally able to explain the decrease in large fish biomass and the general trend of decreasing fish catches as partially caused by nutrient enrichment in an eutrophic environment.

More recently Fréon *et al.* (1993) applied environmental variables in surplus models, such as the Gordon-Schaefer fishery model. They suggest that environmental factors can affect fish biomass in three different ways: through an effect on the carrying capacity ($K$), through an effect on the intrinsic growth rate ($r$), and through a combined effect on both $K$ and $r$. An analysis of the three cases found out that the latter results the most appropriate, specifically for the case where the ratio of these two parameters ($r/K$) is constant (Fréon *et al.*, 1993). Unfortunately the mathematical function that links the environmental variable with growth in biomass is generally unknown. In most cases a general flexible function of the form:

$$X = a + bW^c \qquad [4.71]$$

is used to express the relationship between fish biomass $X$ and the environmental variable $W$, providing as many possible cases as the combinations of the different values for the coefficients $a$, $b$, and $c$ respectively. Moreover, when a single environmental variable may have opposite effects according to its value, where the relationship is non-monotonic, the parabolic function:

$$X = aW + bW^c \qquad [4.72]$$

or the Ricker form:

$$X = aWe^{-bW} \tag{4.73}$$

can be used (Fréon *et al.*, 1993). This implies that at lower values of *W*, increasing the value of the environmental variable leads to an increase in productivity up to a certain point beyond which further increases would cause a decline in productivity.

Following this approach, Kasulo & Perrings (2002) include the influence of changes in environmental conditions introducing into an aggregated Gordon-Schaefer fishery model a biodiversity variable in the form of bioeconomic indexes that capture both biological and economic factors. This tractable model of biodiversity in freshwater fisheries was then applied to a freshwater fishery in Lake Malawi describing the typical Gordon-Schaefer bioeconomic model as follows:

$$\dot{X} = r\left(1 - \frac{X}{K} - eW\right)X - qEX \tag{4.74}$$

where the growth of fish is affected by the environmental variable according to the estimated value of its coefficient *e*, which gives the amount by which a unit change in the environmental variable depresses the natural growth rate of fish biomass. Their model implies that, if the level of the environmental variable negatively affects the growth of fish stocks, then a further negative variation in environmental conditions would lead to a decrease in the Marginal Sustainable Yield, open access and profit maximising levels of effort, and stock size. Moreover, they found that a Gordon-Schaefer model modified to include a bioeconomic diversity index fits the data much better than a standard Gordon-Schaefer model. This suggests that it is possible to explain a very high proportion of the variation in estimated stock size, catch and effort, if a 'single-species' model includes an environmental variable (Kasulo & Perrings, 2002).

## 4.5. Phytoplankton as environmental variable

Among the anthropogenic effects that directly or indirectly affect fish productivity, cultural eutrophication is the fastest growing concern in lakes. A question of long standing concern in natural resource management is the amount of fish harvest that can be derived from primary production in aquatic systems (Oglesby, 1977, Nixon, 1988). Primary production is linked to fishery yield through bottom-up processes. As a consequence, nutrient enrichment will have a positive effect on fishery productivity in nutrient-limited environments such as oligotrophic or mesotrophic lakes. This has been shown to be true for many freshwater ecosystems around the world (Melack, 1976a; Liang *et al.*, 1981; Hoyer & Jones, 1983; Downing *et al.*, 1990; Quirós, 1990; Gomes *et al.*, 2002).

However, there is growing evidence from studies of lakes and estuaries that, in some instances, sustainable harvest of fish populations at upper trophic levels may remain constant or even decline as these aquatic ecosystems become highly eutrophic (Beeton, 1969; Lee *et al.*, 1991; Caddy, 1993). In an eutrophic environment, excess of nutrients affects fish productivity through changes in the amount of food (Bootsma & Hecky, 1993) and volume of habitat (Hammer *et al.*, 1993) available to some fish species. In the latter case this is due both to the increase in the volume of deoxygenated water, which boosts natural mortality of fish, and to a higher sedimentation rate, which spoils nursery grounds and may damage fish eggs. This decreases the transfer efficiency of primary productivity along the food web and, in concomitance with high fishing pressure, it can have a severe impact on fish stock biomasses and fishery yields (Kemp *et al.*, 2001).

Phytoplankton is often used as a measure to determine the trophic state of a water body and is well used as water quality variable in eutrophication models. Phytoplankton is composed of single-celled algae, which is largely microscopic in size ranging from a few microns to a few hundred microns in the longest dimension, or as colonies or filaments composed of many cells. The three determinations usually made in characterizing the phytoplankton as a water quality variable in standing waters are productivity, biomass and species composition. All these characteristics are directly influenced by nutrients loading and nutrients concentration in the water. Therefore, at the base of any eutrophication model there is a need for studying the relationship between nutrients availability and phytoplankton.

In estimating the relationship between phytoplankton dynamics and nutrients concentration, the literature presents two different approaches. One is to relate nutrients concentration to phytoplankton biomass as a whole. Algal biomass in turn is estimated from the chlorophyll-a concentration in the water assuming a dry-weight:chlorophyll ratio of 70 as a rule of thumb (Scheffer, 1998), even though the chlorophyll content of phytoplankton may vary within the range 0.5-2% of the dry weight depending on nutrient status, light and temperature (Ahlgren *et al.*, 1988). The second approach is to model the impact of nutrients availability on the dynamics of several phytoplankton species through an algae competition model. When following the first approach, econometric analysis is used to estimate the equation underlying the relationship between nutrients and chlorophyll-a concentration. The second approach, on the other hand, applies the wide literature of phytoplankton growth models in order to build an inter-specific algae competition model.

### 4.5.1. Nutrients concentration and overall phytoplankton biomass

The dependence of the overall phytoplankton biomass on nutrients is a general feature in freshwater ecosystems. Since Sakamoto (1966) first demonstrated the strong empirical links between phosphorus, nitrogen and chlorophyll and its subsequent generalization by Dillon & Rigler (1974b), many chlorophyll/nutrient relationships have been published.

In most cases these focus on the relationship between chlorophyll-a and total phosphorus concentration in linear models (Megard, 1972; Schindler *et al.*, 1978; Edmondson & Lehman, 1981), logarithmic models (Sakamoto, 1966; Dillon & Rigler, 1974b; Jones & Bachmann, 1976; OECD, 1982) and models with defined maximum levels (Ahl & Wiederholm, 1977; Straškraba, 1980). While linear models imply a constant chlorophyll-phosphorus ratio, logarithmic models plotted on linear scales produce curves with varying slopes. This curvilinear relationship implies an increasing chlorophyll-phosphorus ratio with increasing phosphorus concentrations. This ratio varies due to variations in the chlorophyll and total phosphorus content of phytoplankton and in the amount of total phosphorus which is bound to phytoplankton (Ahlgren *et al.*, 1988). The increasing trend of such ratio is likely due to the fact that a lower fraction of phosphorus generally is bound to phytoplankton in oligotrophic lakes than in eutrophic lakes (Heyman & Lundgren, 1988). Moreover, increasing ratios as phosphorus in a lake increases it implies that phosphorus becomes more efficient at producing a standing crop of algae (Ahlgren *et al.*, 1988). In the models with defined maximum levels this curvilinear relationship presents an upper limit of chlorophyll concentration where factors other than phosphorus are becoming limiting. This upper limit for chlorophyll concentration has been reported to be within a range of 30-50 $\mu g\ l^{-1}$ (Straškraba, 1980) and 50 (deep lakes)-150 (shallow lakes) $\mu g\ l^{-1}$ (Ahl & Wiederholm, 1977).

According to Welch & Lindell (1996), the two most often used expressions among the models mentioned above are those by Jones & Bachmann (1976) and Dillon & Rigler (1974b). Jones & Bachmann (1976) presented a log-linear relationship, where chlorophyll-a (*Chla*) and total phosphorus (*TP*) are both expressed in $\mu g\ l^{-1}$, using data from 143 lakes covering a broad range in trophic state:

$$\log(Chla) = -1.09 + 1.46\log(TP) \qquad [4.75]$$

The value of the regression coefficient for *TP*, which expresses the partial effect of a variation of *TP* on chlorophyll concentration, does not differ significantly from that estimated by Dillon & Rigler (1974b). They developed another widely used relationship for estimating the average summer chlorophyll-a concentration using data from 46 lakes, mostly in eastern Canada:

$$\log_{10}(Chla) = -1.136 + 1.449\log_{10}(TP) \qquad [4.76]$$

Their equation, however, is consistent, and then applicable, only when the *TN:TP* water concentration ratio is > 12 by weight. Values above this threshold correspond to water environments in which *TP* is the limiting nutrient, while values below follow the point first made by Vollenweider (1968a) that in certain situations, in which phosphorus is supplied in abundance, some other nutrients might become limiting.

Despite this observation, total nitrogen (*TN*) has been used infrequently as an indicator. However, as it may be suggested by the Dillon & Rigler's (1974b) study, nitrogen could be a pertinent indicator in highly eutrophic lakes where, with generally low *TN:TP* ratios, it could be expected to control productivity. Indeed, Smith (1982) showed that chlorophyll concentrations are strongly associated with the concentrations of both phosphorus and nitrogen, and consequently the *TN:TP* ratio. From a large sample of 127 north latitude lakes (with all values expressed in μg $l^{-1}$), he presented a chlorophyll predictive equation that also includes nitrogen and may be more useful in some highly eutrophic systems than phosphorus-chlorophyll relationship alone:

$$\log(Chla) = -1.517 + 0.653\log(TP) + 0.548\log(TN) \qquad [4.77]$$

This equation was then compared by Smith (1982) with another estimated from a separate dataset of 101 Florida lakes that confirmed the importance of nitrogen and the validity of multiple regression expression containing both *TN* and *TP*:

$$\log(Chla) = -2.488 + 0.374\log(TP) + 0.935\log(TN) \qquad [4.78]$$

Similarly, Prairie *et al.* (1989) used a dataset of 133 lakes from the northern hemisphere to estimate the following equation, where all variables are measured in μg $l^{-1}$:

$$\log_{10}(Chla) = -2.213 + 0.517\log_{10}(TP) + 0.838\log_{10}(TN) \qquad [4.79]$$

All these studies confirm the contention that both *TP* and *TN* are important variables in the prediction of chlorophyll-a concentrations in lakes. However, trying to explain the chlorophyll-nutrients relationships through a single equation does not always fit well to the plots of observed data from field experiments (Schindler, 1978; Smith, 1982; Vollenweider & Kerekes, 1980). According to Prairie *et al.* (1989), for the specific case of the log-log chlorophyll-phosphorus relationship, the chlorophyll concentration appears to reach a maximum beyond which further increase in phosphorus concentration does not result in higher chlorophyll concentrations. This observation is also at the base of the logarithmic models with defined maximum levels presented by Ahl & Wiederholm (1977) and Straškraba (1980). The asymptotic, or at least strongly curvilinear, nature of the general chlorophyll-phosphorus relationship, which is not new to the literature (Forsberg & Ryding, 1980; Straškraba, 1980; Canfield, 1983), tends to appear similar to a logistic model once transformed to arithmetic units (Straškraba, 1980). On the other hand, the log-log chlorophyll-nitrogen relationship is better described by a sigmoidal curve rather than by a straight line (Prairie *et al.*, 1989).

This curvilinear nature of the chlorophyll-nutrients relationship is explained by the fact that a varying *TN:TP* ratio affects the slope, intercept, and correlation coefficient of both chlorophyll-phosphorus and chlorophyll-nitrogen relationships in lakes. According to Prairie *et al.* (1989), this observation shows that the constancy of slope implicit in all *Chla* = *f*(*TP,TN*) multiple regression models (i.e. Smith, 1982) is inappropriate. In a cross sectional database including several lakes or in a time series database of a single lake, such *TN:TP* relationship may vary considerably. For this reason, Prairie *et al.* (1989) suggest a set of equations explaining the chlorophyll-nutrients relationship varying both slope and intercept over a set of *TN:TP* ratios. However, as they found that *TP* provides better predictions more consistently for lakes of any given *TN:TP* ratio, it has been suggested that a chlorophyll-phosphorus regression equation is best suited to predict chlorophyll concentration than a multiple *Chla* = *f*(*TP,TN*) regression model. Then, in explaining the chlorophyll-phosphorus relationship, they estimated a set of $\log_{10}$-$\log_{10}$ equations as showed in Table 4.1, where all concentrations are in $\mu g\ l^{-1}$.

**Table 4.1:** Coefficients of Chla=f(TP) relationships for different TN:TP ratios.

| TN:TP | Intercept | Slope |
|---|---|---|
| 5 | 0.028 | 0.643 |
| 10 | -0.620 | 1.058 |
| 15 | -0.823 | 1.204 |
| 20 | -0.910 | 1.302 |
| 25 | -0.979 | 1.365 |
| 30 | -0.912 | 1.318 |
| 40 | -0.776 | 1.210 |
| 50 | -0.645 | 1.110 |
| 60 | -0.530 | 1.030 |

***Source*:** Prairie *et al.* (1989).

In conclusion we present (Table 4.2) a summary of the chlorophyll-nutrient empirical equations from the most significant studies found in the literature. Even though there is certainly a dependence of chlorophyll concentration from phosphorus concentration, the impression from a comparison of the results of the different studies is that the *Chla:TP* relationship is very variable. Phosphorus-limited environments will make the ratio *Chla:TP* less variable. However, factors other than phosphorus would still tend to obscure the picture, particularly in eutrophic systems. For this reason, relationships that account for both phosphorus and nitrogen (Smith, 1982) or the value of the *TN:TP* ratio (Prairie *et al.*, 1989) may be more appropriate in modelling eutrophication.

**Table 4.2:** Cholorophyll-nutrients models.

| Model | Source |
|---|---|
| Chla = -7.3 + 1.19*TP | Schindler et al. (1978) |
| Chla = 4.2 + 0.58*TP | Megard (1972) |
| Chla = -4.8 + 0.55*TP | Edmondson & Lehman (1981) |
| Chla = 0.0735*$TP^{1.583}$ | Sakamoto (1966) |
| Chla = 0.0731*$TP^{1.449}$ | Dillon & Rigler (1974b) |
| Chla = 0.0813*$TP^{1.46}$ | Jones & Bachmann (1976) |
| Chla = 0.028*$TP^{0.98}$ | OECD (1982) |
| Chla = -0.2194*$TP^{0.653}$*$TN^{0.548}$ | Smith (1982) |
| Chla = -0.0061*$TP^{0.517}$*$TN^{0.838}$ | Prairie *et al.* (1989) |
| Chla = 150[1 - exp(-0.000867*TP – 0.0000111*$TP^2$)] | Ahl & Wiederholm (1977)$^1$ |
| Chla = 50[1 – exp(-0.0026*TP - 0.000102*$TP^2$)] | Ahl & Wiederholm (1977)$^2$ |
| Chla = 40.1/[1 + 130exp(-0.114*TP)] | Straškraba (1980) |

1 = for shallow lakes; 2 = for deep lakes.

### 4.5.2. Phytoplankton growth models and the effect of resource limitation

The trophic state of a water body can be characterized not only by the level of phytoplankton concentration, measured by chlorophyll-a concentration, but also by phytoplankton species composition. Indeed, phytoplankton community is a composition of several species of algae. The taxonomic orders of algae representing the major portion of the phytoplankton are the Chlorophyta (green algae), Chrysophyta (diatoms and yellow-green algae), Cyanophyta (blue-green algae, also cyanobacteria), Pyrrhophyta (dinoflagellates), Euglenophyta (euglenoids) and Cryptophyta (cryptomonads), although groupings may vary among authors (Welch & Lindell, 1996). These species often tend to out compete each other due to a change in nutrients availability and other environmental factors, generating species succession over seasons or trophic states.

In this latter case, for instance, an oligotrophic lake is generally populated by diatoms and green algae while an eutrophic lake is dominated by cyanobacteria. Indeed, in eutrophic lakes the normal ratio of *N* over *P* concentration changes considerably. It is usually characterized by an excess of $P$ over $N$, the latter becoming the limiting nutrient for algae growth. In this case the N-fixing cyanobacteria have a competitive advantage over green-algae that is further enhanced by the fact that cyanobacteria are more resistant to light limitation generated by the shadow effect of its own biomass. Under such circumstances they easily out-compete green algae generating a trophic cascade effect (Carpenter *et al.*, 1985). Such a shift in phytoplankton species composition may be expected to reduce fish productivity since cyanobacteria are a poor source of food compared to benthic algae, particularly for zooplankton (Bootsma and Hecky, 1993).

Many eutrophication studies use algae competition models to predict the impact of nutrient availability on the dynamics of phytoplankton species composition in lakes (Lehman *et al.*, 1975; Scavia, 1980; Smith & Kalff, 1983; Scavia *et al.*, 1988). These competition dynamics are generally modelled through the effect of resource limitation on species-specific growth rates using phytoplankton growth models.

Phytoplankton growth is basic to the functioning of the normal aquatic ecosystem and is the key process of eutrophication. For this reason, over the last decades, phytoplankton growth models have been widely used in eutrophication models (Jørgensen, 1976; Jørgensen *et al.*, 1978; Scavia & Park, 1976; Riley & Stefan, 1988), where the former are embedded within the latter models in order to predict the effect of resource limitation on primary productivity and then eutrophication. In this case, primary production encompasses the growth of phytoplankton and is mostly represented as the amount of biomass produced per unit volume and time.

In phytoplankton growth models two major controls on photosynthesis are nutrient availability and light intensity (Scavia & Park, 1976). However, also water temperature plays a crucial effect, particularly in temperate lakes where seasonal variations are the main cause for phytoplankton species successions over the year. Therefore, the growth rate of phytoplankton depends on the interaction among these three limiting factors.

Laboratory experiments justify the classical approach that these effects interact among them through a multiplicative construct (Chen, 1970; DiToro *et al.*, 1971) rather than following a construct which mimics Liebig's law of the minimum (Larsen *et al.*, 1973). Adopting a multiplicative construct, phytoplankton growth rate can be written as:

$$r^{phyt} = r_{\max}^{phyt} * (temperature) * (light) * (nutrients) = r_{\max}^{phyt} * G(T) * G(I) * G(C) \qquad [4.80]$$

and,

$0 \le G(T) \le 1$
$0 \le G(I) \le 1$
$0 \le G(C) \le 1$

where $G(T)$, $G(I)$, and $G(C)$ are expressions representing respectively the limiting effect of water temperature, solar radiation and water nutrients concentration on the maximum growth rate ($r_{max}$) of phytoplankton measured at 20°C and optimal light and nutrient conditions.

The processes that contribute to primary productivity exhibit complex environmental relationships. It is therefore of great importance to understand the interacting factors regulating the growth.

### *4.5.2.1. Nutrient limitation*

There are two different approaches to modelling the nutrient limitation effect on phytoplankton growth. In most phytoplankton models the growth rate is related only to the external nutrient concentration (DiToro & Connolly, 1980; Scavia, 1980; Walters, 1980). The alternative approach focuses on the limiting effect of the internal nutrient concentration (Lehman *et al.*, 1975; Jørgensen *et al.*, 1978).

The effect of varying nutrient levels in the water substrate can be quite significant. At a nutrient level of 0 there is, of course, no growth since the phytoplankton requires at least some critical level of nutrients to begin stimulating the growth of the population. As the nutrient level increases, growth commences. However, as nutrient levels continue to increase the effect on the growth rate of phytoplankton is reduced and asymptotically approaches unity. This relationship is stated in a well-known Monod model (Monod, 1949) for nutrient limitation according to Michaelis-Menten kinetics:

$$G(C^i) = \frac{C^i}{C^i + K^i} \qquad [4.81]$$

where $C^i$ is the substrate concentration or the concentration of the *ith* nutrient in the water and $K^i$ represents the Michaelis constant or half saturation constant for a certain type of nutrient. While phytoplankton uptake rate is dependent on the ambient concentration of nutrient according to the Michaelis-Menten kinetics, the growth rate is dependent on the cellular concentration, which may be higher than ambient if there is excess storage. In this latter case, the growth rate would be higher than the uptake rate, while, if cells have below quota concentrations, uptake could be higher than growth rate. Therefore, only under the assumption that internal nutrient content of phytoplankton is always in equilibrium with the nutrient concentration in the water column, then nutrient limitation of growth rate can be predicted from ambient concentration according to equation [4.81]. This assumption is based on the high rate of uptake relative to the growth rate (DiToro, 1980).

It is widely recognised that nitrogen, carbon, phosphorus and, for the specific case of diatoms, silica are the nutrients able to limit phytoplankton growth. Nitrogen and phosphorus are the nutrients that are usually most scarce relative to the needs of organisms and therefore most likely to limit growth and biomass. In very low alkalinity waters carbon is usually in sufficient supply so as not to be limiting at all or to be limiting only in the short term if nitrogen and phosphorus are plentiful (Welch & Lindell, 1996).

The effect of multiple nutrient limitation is often presented as a multiplicative effect (Chen, 1970; DiToro *et al.*, 1971; Dahl-Madsen & Strange Nielsen, 1974). Following this approach it is assumed that all the nutrients might be limiting phytoplankton growth at the same time. If we consider only ambient concentrations of nitrogen ($C^N$) and phosphorus ($C^P$) as limiting nutrients, the multiplicative effect is expressed by the following equation:

$$G(C) = \frac{C^P}{C^P + K^P} * \frac{C^N}{C^N + K^N} \qquad [4.82]$$

In other models the interaction among nutrients is described through an additive function using an average of the limiting factors (Patten *et al.*, 1975):

$$G(C) = \frac{1}{2}\left( \frac{C^P}{C^P + K^P} + \frac{C^N}{C^N + K^N} \right) \qquad [4.83]$$

Another form of averaging treats the reciprocal of the reduction terms according to the following general expression (Bloomfield *et al.*, 1973):

$$G(C) = \frac{n}{\sum_i^n \frac{1}{U^i}} \qquad \text{with, } U^i = \frac{C^i}{C^i + K^i} \qquad [4.84]$$

where $n$ is the number of limiting functions and is used to normalize the total limitation term. Considering only nitrogen and phosphorus as limiting nutrients, this expression becomes:

$$G(C) = \frac{2}{1 \Big/ \left( \frac{C^P}{C^P + K^P} \right) + 1 \Big/ \left( \frac{C^N}{C^N + K^N} \right)} \qquad [4.85]$$

Other authors (i.e. Larsen *et al.*, 1973), following the Liebig's law of the minimum, use the minimum of the limiting factors, which is more in accordance with the limiting nutrient concept:

$$G(C) = \min\left\{ \frac{C^P}{C^P + K^P}, \frac{C^N}{C^N + K^N} \right\} \qquad [4.86]$$

Another possible formulation is a threshold model reported by Welch & Lindell (1996), in which the type of limiting nutrient depends on the nitrogen/phosphorus ratio concentration in the substrate:

$$G(C) = \frac{C^P}{C^P + K^P} \quad \text{if N:P} > 13.5{:}1 \text{ (by weight); } 30{:}1 \text{ (by atoms)}$$

$$G(C) = \frac{C^N}{C^N + K^N} \quad \text{if N:P} < 13.5{:}1 \text{ (by weight); } 30{:}1 \text{ (by atoms)} \qquad [4.87]$$

The choice among the nutrient limiting models depends on many considerations. The multiplicative model [4.82], for instance, has as its biological basis the multiplicative effects of the enzymatic processes involved in photosynthesis (Scavia & Park, 1976). However, several authors (Bloomfield *et al.*, 1973; Lassiter & Kearns, 1973; Riley & Stefan, 1988) point out that the multiplicative method results in an extremely severe limitation of growth due to the reduction effect of multiplying fractions. This severe limitation is probably not found in natural systems, where adaptive shifts can occur in the algal assemblage (Scavia & Park, 1976).

It is believed that the single limiting nutrient more closely simulates the fluctuations of algal populations (De Groot, 1983). In fact, the effect of multiple limiting nutrients is probably neither singular nor multiplicative (Monod, 1949). Indeed, the construct [4.86], which mimics the Liebig's law of the minimum, can be expected to produce reasonable results when dealing with pure cultures (Scavia & Park, 1976). Nevertheless, when natural assemblages are considered, the adaptive ability of the assemblage precludes the use of this construct as well (Steele, 1974). Then, when it is reasonable to assume that adaptation and new species replacement in a natural assemblage will moderate the limiting effect of nutrients, additive models of the average [4.83] or reciprocal [4.84] of the limiting factors and threshold models [4.87] may better represent the actual limitation process at the ecosystem level.

The ambient nutrient limitation models above are all based on the assumption that cell concentrations are at quota amounts which correspond to the constant stoichiometric ratios, or ratios in cell material, of carbon, nitrogen and phosphorus in the phytoplankton. The Redfield ratio, or average ratio found in marine plankton, is often used to judge the requirements of phytoplankton (Redfield *et al.*, 1963; Sverdrup *et al.*, 1942):

$$C : N : P = 106 : 16 : 1 \text{ (by atoms)} = 41 : 7.2 : 1 \text{ (by weight)} \qquad [4.88]$$

The ratio $C : N : P$ in the ambient water is often used to indicate which nutrient may be depleted first, assuming that the nutrients were taken up by the algal cells in the required Redfield ratio, and thus could be identified as the potentially most limiting nutrient.

However, the assumption of cell concentration at quota amount is not strictly accurate, particularly for highly eutrophic lakes where, as shown by Jørgensen (1976), it is necessary to take the varying cell composition into account. In this case, internal nutrient concentration models may be preferred to the ambient nutrient concentration models. The former are based on laboratory experiments which have shown that phytoplankton growth is a two-step process: 1) uptake of nutrient; and 2) cell division controlled by the internal nutrient concentration.

The basic internal nutrient concentration model refers to the early studies by Lehman *et al.* (1975) and Jørgensen *et al.* (1978). The model proposed uses the Droop's (1973) derivation of the internal nutrient limitation term for phytoplankton growth which is assumed to dependent solely on internal nutrient concentration:

$$G(Q^i) = \frac{Q^i - Q^i_{\min}}{Q^i} \qquad [4.89]$$

where $Q^i$ is the quantity or quota of *ith* nutrient concentration per cell; and $Q^i{}_{\min}$ is the minimum cell nutrient content. This expression is complemented by the Lehman *et al.* (1975) expression for nutrient uptake which recognizes uptake rates as functions of both internal and external nutrient levels according to the following equation:

$$u = u_{\max} \frac{Q^i_{\max} - Q^i}{Q^i_{\max} - Q^i_{\min}} * \frac{C^i}{C^i + K^i} \qquad [4.90]$$

where $u_{\max}$ is the maximum uptake velocity; and $Q_{\max}$ is the maximum nutrient storage capacity of a cell. In this expression ambient concentration influences the growth through a two-step procedure which requires modelling of the effect of the uptake rate on the internal nutrient pool. The internal nutrient pool dynamics are modelled by Jørgensen *et al.* (1978) according to the following equation:

$$\frac{dQ^i}{dt} = u^i * C_{phyto} - \left( \frac{v_{phyto}}{H} + \frac{r_{zoo} Q^i_{zoo}}{Y_{zoo}} + \frac{S}{V} \right) * Q^i \qquad [4.91]$$

where $C_{phyto}$ is the phytoplankton concentration in the water substrate; $V$ is the volume of the water body; $S$ is the water outflow, $H$ is the mean depth of the water body; $v_{phyto}$ is the settling velocity rate of phytoplankton; $Y_{zoo}$ is the yield of feeding zooplankton; $r_{zoo}$ is the growth rate of zooplankton; $Q^i{}_{zoo}$ is the proportion of *ith* nutrient in zooplankton.

The Lehman *et al.* (1975) and Jørgensen *et al.* (1978) model for the internal nutrient concentration seems more in accordance with the observed natural processes than the ambient concentration models, especially for the cases where the concentrations of soluble nutrients are fluctuating (Jørgensen, 1980).

### *4.5.2.2. Light limitation*

The light-limitation construct on phytoplankton primary production takes into account the reduction of the photosynthetic rate for light intensities. Light intensity ($I_Z$) and its attenuation with depth ($Z$) of the water layer are determined from the Beer-Lambert law for light extinction based on a vertical light path:

$$I_Z = I_0 \exp(-\varepsilon Z) \tag{4.92}$$

where $I_0$ is the incident light intensity which is the solar radiation at the surface expressed in cal $m^{-2}$ $s^{-1}$; $I_Z$ (cal $m^{-2}$ $s^{-1}$) is the remaining light intensity at depth Z; $\varepsilon$ ($m^{-1}$) is the vertical attenuation coefficient for water and dissolved and particulate matter combined; and $Z$ is depth. Some studies (i.e. Jørgensen, 1976) included the influence of the phytoplankton concentration on the extinction coefficient. Then, the vertical light attenuation coefficient can be further defined as:

$$\varepsilon = \varepsilon_{wd} + \varepsilon_p Chla \tag{4.93}$$

where $\varepsilon_{wd}$ ($m^{-1}$) is the attenuation by water and dissolved matter or background turbinity; $\varepsilon_p$ ($m^2$ $mg^{-1}$ Chla) is the attenuation coefficient due to particulate matter, with values often between 0.02 (Matsuoka *et al.*, 1986) and 0.025 (Welch & Lindell, 1996); and chlorophyll-a (*Chla*) is the measure of particulate matter expressed in mg $m^{-3}$.

For modelling purposes it is convenient to calculate average conditions over a given depth of water. Thus, this approach uses an average value ($I$) of light intensity. Following the first models by Steele (1965) and Vollenweider (1965), values for $I$ are approximated as the vertical integration of an exponential light attenuation function over a certain depth divided by that depth:

$$I = \frac{\int_0^Z I_0 e^{-\varepsilon Z} dZ}{Z} = -\frac{I_0}{\varepsilon Z}\int_0^Z d\left(e^{-\varepsilon Z}\right) = -\frac{I_0}{\varepsilon Z}\left(e^{-\varepsilon Z} - 1\right) = \frac{I_0}{\varepsilon Z}\left(1 - e^{-\varepsilon Z}\right) \tag{4.94}$$

Under mixing conditions the depth of the mixing water layer may limit the light available to a mixing population of plankton cells. For this reason it could be more appropriate to estimate the average amount of light available ($I$) to mixing cells considering the depth of the mixing water layer $Z_m$. Thus, equation [4.94] becomes:

$$I = \frac{\int_0^{Z_m} I_0 e^{-\varepsilon Z_m} dZ_m}{Z_m} = -\frac{I_0}{\varepsilon Z_m} \int_0^{Z_m} d\left(e^{-\varepsilon Z_m}\right) = -\frac{I_0}{\varepsilon Z_m}\left(e^{-\varepsilon Z_m} - 1\right) = \frac{I_0}{\varepsilon Z_m}\left(1 - e^{-\varepsilon Z_m}\right) \qquad [4.95]$$

Following Kemp *et al.* (2001), equation [4.94] or [4.95] can be used directly as light limiting factor for algal photosynthesis, together with the self-shading expression [4.93], which implies that light absorption by phytoplankton cells contributes substantially to total attenuation:

$$G(I) = \frac{I_0}{\varepsilon Z}\left(1 - e^{-\varepsilon Z}\right) \qquad [4.96]$$

An alternative is to describe the relation between light intensity and production rate by means of Michaelis-Menten's equation as shown in Jørgensen (1980):

$$G(I) = \frac{I}{I + K_I} \qquad [4.97]$$

where $K_I$ is the half saturation constant for light.

A shade-impact function of a Monod type describing a decline of photosynthesis with shade has been presented by Scheffer (1998):

$$G(I) = \frac{K_S}{K_S + \varepsilon Z} \qquad [4.98]$$

where $\varepsilon$ is given by equation [4.93] and $K_S$ is the half saturation constant for light, which represents the shade tolerance for algae. Note that in expression [4.98] the half saturation constant ($K_S$) differs from that ($K_I$) of equation [4.97]. Since the function is formulated in terms of shading ($\varepsilon Z$) rather than absolute light ($I_0 e^{-\varepsilon Z}$), the shade tolerance is not a universal physiological property of the algae, as it depends on $I_0$ and thus on latitude and season (Scheffer, 1998).

Several other models (Forsberg & Shapiro, 1981; Megard *et al.*, 1984) predict light limitation of algae growth. One of the more widely used has been presented by Steele (1962). It is based on the observation that as radiation increases, photosynthesis increases up to a maximum level, beyond which further increases in light levels tend to result in photoinhibition. The model predicts photosynthetic rate at depths throughout the water column according to the equation:

$$G(I) = \frac{I}{I_s} \exp\left(1 - \frac{I}{I_s}\right) \qquad [4.99]$$

where $I_s$ (cal m$^{-2}$ s$^{-1}$) denotes the saturation light intensity or the light intensity at which photosynthetic rate is saturated. Following this equation, light limitation is approximated by a time and depth integrated expression of the form used by Steele (1965) which implies rate reductions both at low light intensities and at very high light intensities:

$$G(I) = \int\int G(I(t,Z))dtdZ = \frac{2.718f}{\varepsilon Z}\left[\exp\left(-\frac{I_0}{I_s f}\exp(-\varepsilon Z)\right) - \exp\left(\frac{I_0}{I_s f}\right)\right] \qquad [4.100]$$

where $f$ represents the daylight photoperiod.

Other models presented by (Jørgensen, 1976; Jørgensen *et al.*, 1978) consider a light limitation expression which includes a half saturation constant for light intensity:

$$G(I) = \frac{1}{\varepsilon Z} \ln\left(\frac{I_0 + K_I}{I_0 \exp(-\varepsilon Z) + K_I}\right) \qquad [4.101]$$

where $K_I$ is the Michaelis constant for the light intensity set to 400 cal m$^{-2}$ 24h$^{-1}$ according to Gargas (1975), and $\varepsilon$ represents a self-shading expression according to equation [4.93].

### *4.5.2.3. Temperature limitation*

The growth rates of all phytoplankton populations are also affected by water temperature. Different functions, representing the limiting effect to phytoplankton growth rate due to water temperature, have been used in phytoplankton growth models (Lehman *et al.*, 1975; Scavia & Park, 1976; Scavia, 1980; Walters, 1980; Bierman & Dolan, 1981).

Among them, probably the most acceptable form without an excessive number of parameters is given by Lehman *et al.* (1975) as:

$$G(T)=\exp\left[-2.3\left(\frac{T-T_{opt}}{T_{opt}-T_{\min}}\right)^2\right] \quad \text{if } T<T_{opt} \qquad [4.102]$$

$$G(T)=\exp\left[-2.3\left(\frac{T-T_{opt}}{T_{\max}-T_{opt}}\right)^2\right] \quad \text{if } T>T_{opt} \qquad [4.103]$$

where $T_{opt}$ represents the optimum temperature for nutrient saturated growth; $T_{min}$ is the low temperature at which growth is reduced by 90% from optimum (≈ 0°C); and $T_{max}$ is the high temperature at which growth is reduced by 90%.

Alternatively, other authors (Scavia & Park, 1976; Scavia, 1980) presented an expression depicting the temperature effect on phytoplankton growth when $T < T_{\max}$:

$$G(T)=\left(\frac{T_{\max}-T}{T_{\max}-T_{opt}}\right)^x \exp\left[x\left(1-\frac{T_{\max}-T}{T_{\max}-T_{opt}}\right)\right] \qquad [4.104]$$

with,

$$x=\left[\frac{W\left(1+(1+40/W)^{0.5}\right)}{20}\right]^2$$

$$W=\left(T_{\max}-T_{opt}\right)*\ln Q_{10}$$

where $Q_{10}$ represents a simplified Van't Hoff-Arrhenius expression whose value has been estimated as approximately constant at 1.88 (Eppley, 1972; Goldman & Carpenter, 1974).

### 4.5.3. Inter-specific algae competition models

Several studies model the competitive interaction among phytoplankton species due to changes in water nutrient concentrations. These models may have several levels of complexity depending on the number of trophic levels included. Some may include only one level of complexity without considering the effect of predation on algae. In this case inter-specific competition dynamics among phytoplankton are regulated solely by limiting resources such as phosphorus (Smith & Kalff, 1983) or both phosphorus and light availability (Scheffer *et al.*, 1997; Scheffer, 1998). However, in order to predict more closely the dynamics of natural environments, many models (Lehman *et al.*, 1975; Rose *et al.*, 1988) also account for the interaction of phytoplankton-zooplankton, considering the different ingestion rates of zooplankton on each phytoplankton species. In most cases (Scavia, 1980; Scavia & Park, 1976; Matsuoka *et al.*, 1986), they are detailed ecosystem models considering the whole set of relationships phytoplankton-zooplankton-fishes.

Generally, in an algae competition model the phytoplankton growth equation, which provides predictions of changes in productivity based on changing environmental conditions, is further embedded within a mass-balance equation. This equation is used to model phytoplankton biomass dynamics in a completely mixed lake as follows:

$$V\frac{dC^{phyt}}{dt} = V\left(r^{phyt} - \mu^{phyt} - Z^{zoo}_{phyt} - \frac{v_s^{phyt}}{H}\right)C^{phyt} + Q_{in}C_{in}^{phyt} - Q_{out}C^{phyt} \qquad [4.105]$$

where $C^{phyt}$ is the phytoplankton concentration per unit volume; $r^{phyt}$ represents the phytoplankton growth rate as given by expression [4.80]; $\mu^{phyt}$ represents phytoplankton losses due to respiration and senescence; $Z^{zoo}_{phyt}$ is the loss due to predation by zooplankton; $v_{phyt}$ is the sedimentation rate of phytoplankton; $C_{in}^{phyt}$ is the inflowing concentration of phytoplankton; $V$ is the lake water volume; $H$ is the lake mean depth; and $Q_{in}$ and $Q_{out}$ represent the hydraulic inflowing and outflowing respectively.

Following this approach, for instance, Scheffer *et al.* (1997) presented a two-species competition model between cyanobacteria and green algae. In this model the gross growth rate ($r$) for algal biomass is limited both by light and nutrients, where the nutrient limitation is described as a function of phosphorus only. Then, for the *ith* algal species, (where $i$ =green algae or cyanobacteria) the two species model is (Scheffer *et al.*, 1997):

$$\dot{A}^i = r^i A^i \frac{1}{1+q^i\varepsilon}\frac{P_F}{h_P^i + P_F} - \left(l^i + f\right)A^i \qquad [4.106]$$

or alternatively (Scheffer, 1998):

$$\dot{A}^i = r^i A^i \frac{h_s^i}{h_s^i + \varepsilon Z_m} \frac{P_F}{h_P^i + P_F} - \left(l^i + f\right) A^i \qquad [4.107]$$

with,

$$\varepsilon = \sum_i \varepsilon^i A^i$$

$$P_F = C^P - \sum_i c_P^i A^i$$

where $A^i$ is biomass of the i*th* algae expressed in chlorophyll-a concentration; $r^i$ is the gross productivity rate; $Z_m$ is depth of the mixed water layer; $h_s^i$ is the species-specific shade tolerance; $l^i$ is the fixed loss rate due to respiration, senescence and sinking; $q^i$ is the sensitivity to turbidity; $\varepsilon$ is the vertical light attenuation coefficient; $\varepsilon^i$ is the specific light attenuation coefficient for the i*th* algae; $P_F$ is the directly available "free" phosphorous concentration; $h_p^i$ is the half-saturation concentration of phosphorus for the i*th* algae; $C^P$ is the total phosphorous concentration in the water; $c_p^i$ is the average phosphorus content of the i*th* species; and *f* is the species-independent flushing rate.

This model is based on field observations of cyanobacteria dominance patterns that show how algal communities may resemble hysteretic systems with multiple equilibria. This multiplicity of stable state has already been discussed for eutrophic shallow freshwater systems (Scheffer, 1989; Scheffer, 1990). Such dynamics are also predicted in the two-algae models (Scheffer *et al.*, 1997; Scheffer, 1998), due to the differences in physiological parameters of cyanobacteria and green algae. The basic mechanism is that cyanobacteria are the superior competitors under condition of low light, but also promote such conditions as they can cause a higher turbidity per unit of biomass than other algae. All qualitative results follow from the assumption that cyanobacteria have a lower gross productivity (*r*), loss rate (*l*), and sensitivity to turbidity (*q*), but they have a higher shade tolerance ($h_s$) and cause a higher turbidity per unit of biomass ($\varepsilon$). In general, where the suffixes "c" and "g" identify cyanobacteria and green algae respectively, the four basic inequalities of the models are:

$$r^c < r^g \qquad l^c < l^g \qquad q^c < q^g \qquad h_s^c > h_s^g \qquad \varepsilon^c > \varepsilon^g$$

These inequalities are confirmed by data (Table 4.3) from field experiments where a typical cyanobacteria, *Planktothrix agardhii*, and a typical green algae, *Scenedesmus* *protuberans*, are used to represent the two respective groups (Scheffer *et al.*, 1997; Scheffer, 1998).

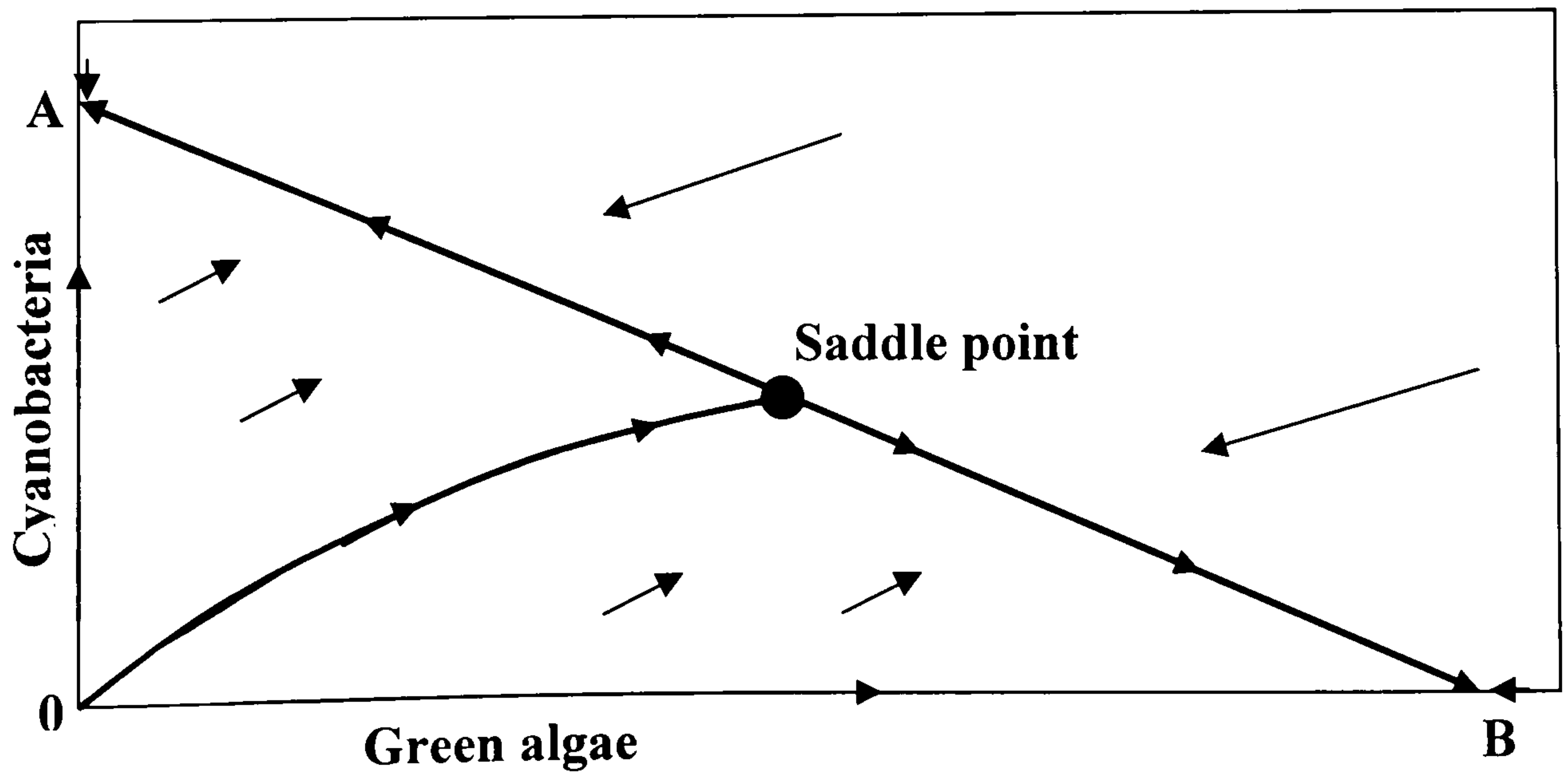
**Table 4.3:** Parameter dimensions of the two-species phytoplankton competition model.

| Parameter | | Green algae | Cyanobacteria | |
|---|---|---|---|---|
| Gross productivity rate[1] | $r^i$ | 1.2 | 0.6 | $day^{-1}$ |
| Loss rate[1] | $l^i$ | 0.12 | 0.06 | $Day^{-1}$ |
| Specific extinction coefficient[1] | $\varepsilon^i$ | 5 | 10 | $M^2 g^{-1}$ |
| Shade tolerance[2] | $h_s^i$ | 1.5 | 3 | - |
| Sensitivity to turbidity[1] | $q^i$ | 2 | 1 | M |

**Sources:** 1= Scheffer *et al.* (1997); 2= Scheffer (1998).

At a given flushing rate and phosphorus concentration, in addition to one trivial monoculture equilibria (points A & B in Figure 4.1) for each of the two species and the point at the origin, the model presented by Scheffer has only one equilibrium point in which both species coexist. However, this coexistence is always unstable as the corresponding equilibrium is a saddle for all the parameters value satisfying the above inequalities. In this case, the boundary of their basin of attraction is the stable manifold of the saddle connecting the origin with the unstable equilibrium point. Out of this line, simulation will end in either of the trivial monoculture equilibria depending on which side of this boundary they start.

**Figure 4.1:** Unstable equilibrium in competition dynamics (Scheffer *et al.*, 1997).



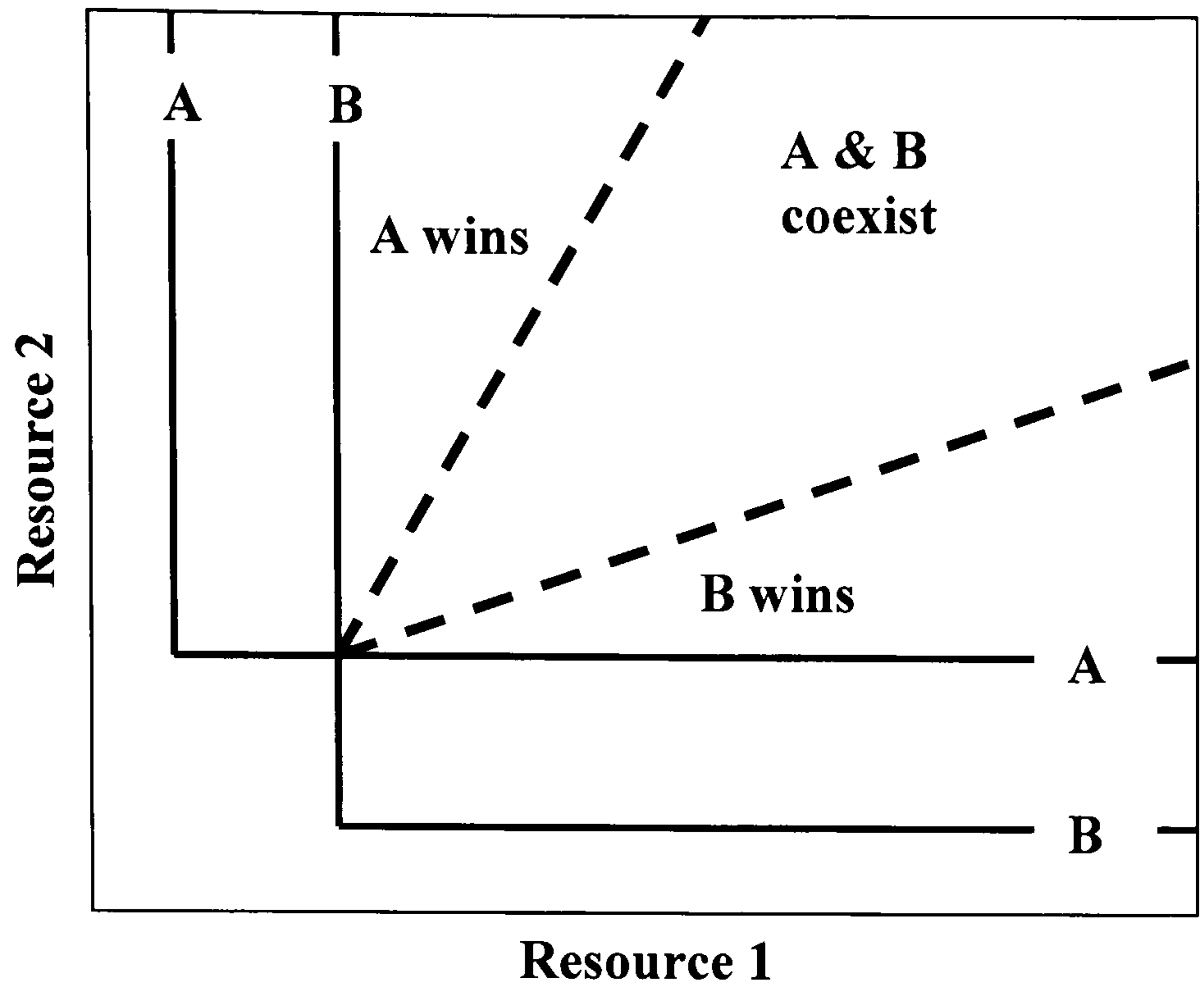
In Volterra models unstable coexistence occurs if inter-specific competition is stronger than intra-specific competition. Indeed, green algae have a competitive advantage in an environment dominated by them because there is more light available, whereas cyanobacteria have an advantage in the turbid condition caused by their own dominance. However, as Scheffer *et al.* (1997) pointed out, the monoculture prediction of this minimal model is rarely observed in natural environments. Many other factors than those included in the model operate in lakes. Some, such as photoinhibition and inedibility of cyanobacteria, enhance the hysteresis, while others, such as spatial heterogeneity and temporal variation of the environment, help preventing competitive exclusion.

More accurate results, which allow for a more stable coexistence of competitive species, may be predicted when considering phytoplankton growth models with more than one limiting nutrient. Several authors (Harris, 1980; Smayda, 1980) argue that a single limiting nutrient approach would be too simplistic to explain a phenomenon as complex as algal succession. For this reason, a multiple nutrient approach may be preferred, in which different combinations of potentially limiting nutrients have different effects on phytoplankton community structure according to the "Resource Ratio Hypotesis" formulated by Tilman (1982). Tilman's theory predicts that the coexistence between two competing species is unstable if each species consumes relatively more of the resource of which it also tolerates the lowest levels.

Scheffer's model, where competition is for light and phosphorus, follows this theory. However, introducing additional limiting resources, such as nitrogen for instance, may better explain the existence of a relatively wide coexistence interval for green algae and cyanobacteria. This interval can be seen (Figure 4.2) when plotting the resource-dependent growth isoclines along which the reproductive rate of a population (species A & B) balances its mortality rate and the supply rate of each resource balances consumption (Tilman *et al.*, 1982).

We can assume these limiting resources to be nitrogen and phosphorus, or light and phosphorus as in the Scheffer's example. However, a tridimensional plot with light, nitrogen and phosphorus would predict competition dynamics closer to the reality. In this latter case phosphorus recycling from the sediments and species-specific light effect may generate the hysteretic pattern often observed in many eutrophic lakes. This mechanism of hysteresis offers an explanation for the resistance of cyanobacteria dominance in shallow lakes to restoration efforts by means of nutrient reduction.

**Figure 4.2:** Competitive coexistence and exclusion between species (Tilman *et al.*, 1982)



Green-algae vs. cyanobacteria competition models can be compared with Smith's (1986) study in which the factors determining the relative biomass of cyanobacteria were studied using multiple linear regression analyses from a dataset of 22 lakes worldwide. He presented two alternative equations where total nitrogen (*TN*, mg $m^{-3}$), total phosphorus (*TP*, mg $m^{-3}$), light estimated from Secchi disc transparency (*SD*, meters), and the depth of the mixed water layer ($Z_m$, meters) interact to determine the relative biomass of planktonic blue-green algae (*%BG*) as follows:

$$Logit(\%BG) = 2.358 - 1.297 \ln TN + 0.692 \ln TP - 2.058 \ln SD + 0.538 \ln Z_m \qquad [4.108]$$

$$Logit(\%BG) = 1.096 - 1.604 \ln\left(\frac{TN}{TP}\right) - 1.119 \ln\left(\frac{SD}{Z_m}\right) \qquad [4.109]$$

Smith (1986) found that at fixed *TN:TP* ratios cyanobacteria relative biomass increases as light availability decreases, while at a fixed light level their relative biomass increases as the *TN:TP* ratio decreases. This is consistent with the current knowledge of algal physiology in the framework of algal community structure.

## 4.6. Trophic state index as environmental variable

Since the 1960-70s several attempts have been made to quantitatively evaluate the trophic state of lakes using single-variable trophic state indexes (Vollenweider, 1968b; Rodhe, 1969; Beeton & Edmondson, 1972; Dobson *et al.*, 1974). However, the subjective judgements of single-variable trophic state criteria (see Table 1.1) and the multidimensional nature of the eutrophication phenomenon means that no single variable is representative of the eutrophication status of a given water body. More robust multiparameter indexes, based on a combination of a number of parameters, have been used (Brezonik & Shannon, 1971; Shannon & Brezonik, 1972). Nevertheless, a multiparameter index is limited in its usefulness because of the number of parameters that must be measured (Carlson, 1977). Furthermore, the use of descriptive classifications for lake trophic states such as oligotrophic, mesotrophic, eutrophic and so on, could create difficulties when attempting to describe continuous changes in a lake's trophic state or in studying quantitatively the eutrophication mechanism.

To overcome these problems Carlson (1977) introduced a trophic state index (TSI) in which a 0-100 scale provides continuous numerical classes of lake trophic states and all the parameters are related to each other by a series of predicted equations. In this case, measurements of any water quality parameter can be used to determine the trophic status of a lake. Carlson's TSI, which allows for a quantitative measure of the degree of lake eutrophication, was originally based on the $\log_2$-transformed Secchi disk[8] transparency (*SD*) such that doubling of transparency resulted in 10-fold increases in the index values, and the range of index values (0 to 100) corresponded generally to transparency extremes (6 cm to 64 m) observed in nature.

Indeed, *SD* is a common method for estimating the depth of light penetration in lake waters assuming that *SD* transparency is expressed by the Beer-Lambert law's [4.92] for vertical extinction of light in water. Under such assumption and according to [4.93], increased turbidity due to phytoplankton and other suspended materials would increase the value of the extinction coefficient and cause the Secchi disk to disappear at shallower depths. Then, given its relationship with the reciprocal of phytoplankton concentration, *SD*

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[8] The Secchi disk transparency method was introduced more then a century ago by the Italian physicist Angelo Secchi. It consists in a black-and-white disk lowered into the water, and the depth (measured in meters) at which it just disappears from view is noted as the Secchi-depth.

could be used as measure of lake's eutrophication. Following these assumptions the TSI as function of *SD* is obtained according to the following equation (Carlson, 1977):

$$TSI(SD) = 10 * (\log_2 64 - \log_2 (SD)) \qquad [4.110]$$

He uses a single trophic criterion, represented by Secchi depth transparency, as basis for an index in which other water quality variables could be used to determine the trophic status by means of established relationships. Carlson (1977) then estimates a series of predictive empirical equations among chlorophyll-a (*Chla*), total phosphorus (*TP*) and Secchi depth (*SD*) as follows:

$$\ln(SD) = 2.04 - 0.68 \ln(Chla) \qquad [4.111]$$

$$\ln(Chla) = -2.442 + 1.449 \ln(TP) \qquad [4.112]$$

and, from the combination of [4.111] and [4.112],

$$SD = \frac{48}{TP} \qquad [4.113]$$

where both *Chla* and *TP* are expressed in $\mu g\ l^{-1}$ and *SD* in meters. Given these relationships the TSI can be measured by either chlorophyll-a, total phosphorus concentration or Secchi depth as follows:

$$TSI(SD) = 10 * \left(6 - \frac{\ln(SD)}{\ln 2}\right) = 60 - 14.427 \ln(SD) \qquad [4.114]$$

$$TSI(Chla) = 10 * \left(6 - \frac{2.04 - 0.68 \ln(Chla)}{\ln 2}\right) = 30.6 + 9.81 \ln(Chla) \qquad [4.115]$$

$$TSI(TP) = 10 * \left(6 - \frac{\ln\left(\frac{48}{TP}\right)}{\ln 2}\right) = 4.15 + 14.427 \ln(TP) \qquad [4.116]$$

Since Carlson's (1977) first introduction, other authors have further developed the trophic state index concept adding more interrelated trophic criteria. Kratzer & Brezonik

(1981) introduced a TSI based on total nitrogen (*TN*), obtained from a chlorophyll-nitrogen regression relationship for 40 nitrogen limited Florida lakes:

$$TSI(TN) = 10 * \left( 6 - \frac{\ln\left(\frac{1.47}{TN}\right)}{\ln 2} \right) \qquad [4.117]$$

Porcella *et al.* (1980) proposed not only a TSI for *TN*, but also for net dissolved oxygen (*NetDO*) and percent of available lake area covered by macrophytes (*MAC*):

$$TSI(TN) = -23.8 + 14.427 \ln(TN) \qquad [4.118]$$

$$TSI(netDO) = 10 \frac{\sum_{i=0}^{Z_{max}} \left|(EDO - CDO)_i\right| \Delta V_i}{V} \qquad [4.119]$$

$$TSI(MAC) = \%MAC \qquad [4.120]$$

where the lake equilibrium instantaneous total dissolved oxygen (*EDO*), obtained from atmospheric pressure and the temperature-depth profiles, is defined as the reference value for a clean-water lake; *CDO* is the instantaneous total lake dissolved oxygen; *V* represents the lake water volume; $Z_{max}$ is the maximum depth; $\Delta V$ is the volume at a selected and convenient depth increment; *i* is the depth increment; and *%MAC* is the percent of lake area subject to growth of macrophytes that is actually covered by such vascular plants.

This area for potential growth can be defined as the area encompassed by the lake margin and whichever is shallower: either the 10 m line hydrostatic limit or the photic zone, the depth at which light becomes limiting to vascular plant distribution and growth, i.e. 2**SD*, (Porcella *et al.*, 1980). The TSI(*MAC*) index follows the general assumption that high quality lakes have few macrophytes and lower quality lakes have more macrophytes in the littoral zones. Indeed, Rigler (1964) reported that return of phosphorus from the littoral zone during the summer season was about 20% higher than the loss. Furthermore, as macrophytes are capable of assimilating phosphorus from the sediment, it is clear that littoral vegetation could play a major role in the dynamics of eutrophication by supplying nutrients to the free water in periods where the water level is low (Jørgensen, 1980).

Following Carlson (1977) and later developments by (Porcella *et al.*, 1980), the continuous trophic state index may alternatively be estimated by any of the water quality variables considered (Table 4.4). Generally, the trophic state can be identified from the ranges of the 0-100 scale in which the index is divided: 0-30 representing oligotrophic; 30-40 oligo-mesotrophic; 40-50 mesotrophic; 50-60 meso-eutrophic; 60-70 eutrophic; and 70-100 hypereutrophic (Xu *et al.*, 2001).

**Table 4.4:** Trophic state index.

| **TSI** | **SD*** (m) | **TP*** ($\mu$g $l^{-1}$) | **TN**** ($\mu$g $l^{-1}$) | **Chla*** ($\mu$g $l^{-1}$) | **NetDO**** (mg $l^{-1}$) | **MAC**** (% of available lake area covered) |
|---|---|---|---|---|---|---|
| *0* | 64 | 0.75 | 5.2 | 0.04 | 0 | 0 |
| *10* | 32 | 1.5 | 10.4 | 0.12 | 1 | 10 |
| *20* | 16 | 3 | 20.8 | 0.34 | 2 | 20 |
| *30* | 8 | 6 | 41.6 | 0.94 | 3 | 30 |
| *40* | 4 | 12 | 83.2 | 2.61 | 4 | 40 |
| *50* | 2 | 24 | 167 | 7.23 | 5 | 50 |
| *60* | 1 | 48 | 333 | 20 | 6 | 60 |
| *70* | 0.5 | 96 | 666 | 55.5 | 7 | 70 |
| *80* | 0.25 | 192 | 1330 | 154 | 8 | 80 |
| *90* | 0.125 | 384 | 2670 | 426 | 9 | 90 |
| *100* | ≤0.0625 | ≥768 | ≥5330 | ≥1180 | ≥10 | 100 |

***Source:*** *Carlson (1977); **Porcella *et al.* (1980).

It must be noticed that, according to Carlson (1977), all the water quality parameters whenever transformed to the trophic scale should give approximately the same value. However, this may not happen when the assumptions about the relationships among the various components of the lake ecosystem do not hold. In this case, the TSI scale not only may be used to classify the lake but can also serve as internal check of these relationships (Carlson, 1977). Secchi disk transparency, for instance, may give erroneous values of TSI(*SD*) in lakes containing high amounts of non-algal particulate matter, in highly colored lakes, and in extremely clear lakes. Divergences between TSI(*TP*) and TSI(*Chla*) may indicate whether or not phosphorus is limiting to algal production. Phosphorus limitation is indicated where TSI(*Chla*) ≥ TSI(*TP*); but when TSI(*Chla*) < TSI(*TP*), limitation by some other factor, i.e. light or another nutrient, is indicated (Carlson, 1991). Other authors (Kratzer & Brezonik, 1981) demonstrated that instances of TSI(*TN*) > TSI(*TP*) corresponded precisely with conditions of phosphorus limitation, while instances of TSI(*TN*) < TSI(*TP*) were common in nitrogen-limited environments.

Porcella *et al.* (1980) proposed a 0-100 scale Lake Evaluation Index (LEI) which overcomes the problem of diverging values among trophic state indexes by averaging specific target variables as follows:

$$LEI = 0.25 * \left[ \frac{TSI(Chla) + TSI(MAC)}{2} + TSI(SD) + TSI(TP) \right] \qquad [4.121]$$

Averaging the TSI values is an approach which is not new in the literature as other authors have used it (i.e. Xu *et al.*, 2001). This common practice, however, is criticized by Carlson (1977), as he suggested that, where the biological TSI values diverge from that predicted by the abiotic components, priority should be given to the firsts, especially the TSI(*Chla*).

## 4.7. Discussion

Watershed degradation and land use change affect water quality, having profound implications for the freshwater ecosystem and fisheries of Lake Victoria. In this context, the theory of ecosystem and biomass dynamics models presented in this chapter provides the basis for our watershed approach to fishery and wetland management.

In chapter 5, dynamic simulations through an ecosystem model using the Ecopath approach will show how eutrophication may explain the sudden upsurge of the introduced Nile perch during the 1980s. In chapter 6, to quantify the combined effect of eutrophication and fishing effort on the aggregate fish stock and fishery, we specify and estimate a modified Gordon-Schaefer model, which embeds a measure of phytoplankton biomass, in terms of chlorophyll-a concentration, obtaining the general expression for the environmental variable which influences fish growth and fishery yields. In the current chapter we also present other environmental variables, such as the trophic state index, or models (i.e. phytoplankton growth models; inter-specific algae competition models) that could be embedded in the Gordon-Schaefer fishery model as an alternative to chlorophyll-a concentration.

In chapter 7 we will use the interaction between land use and the fishery to identify catchment-based indicators, in particular price indicators and wetland extent, which may anticipate changes in environmental variables driving stock assessment models. Our model first links nutrient loading and phytoplankton density (i.e. chlorophyll-a concentration) using functional forms drawn from the literature. In doing so we convert nutrients flows in concentrations, drawing on the Vollenweider's limnological model, which can be used to evaluate or predict changes in the nutrient balance of the lake when the input changes from one constant level to another load level. Then, we combine these components with the estimated biomass dynamics model of chapter 5.

# PART II:

## *The case of Lake Victoria*

# Chapter 5

# ECOSYSTEM MODEL OF LAKE VICTORIA

## 5.1. Introduction

As indicated in Chapter 1, a number of studies have shown that Lake Victoria fisheries are overexploited. However, the declining stocks of commercial fish in the lake over the last decades may also be related to other factors, such as eutrophication (§ 1.3.1.2) and changes in the structure of the food web (§ 1.3.1.3). Ecosystem models (§ 4.3.2) are an appropriate tool for specifying dynamics and for constructing future scenarios of change in a complex trophic web. These models may be used to estimate the impact of nutrient enrichment on freshwater ecosystems and to trace the effects of changes in primary productivity.

## 5.2. The Ecopath model of Lake Victoria

There is some experience with ecosystem models based on Ecopath in Lake Victoria (Moreau *et al.*, 1993; Villanueva & Moreau, 2002). Moreau *et al.* (1993) estimated the steady-state biomasses in two different periods, 1971-1972 and 1985-1986, to study the effect of the growth of the Nile perch. Their study traces the effects of a shift in dominance from haplochromines species to *Lates niloticus* (Table 5.1). They found that it did not affect the trophic level at which fisheries operate – which only increased from 3.98 to 4.10. Nevertheless, overall catches from the system increased considerably as did the gross efficiency of the fisheries (catch/net primary productivity), which increased from 0.0016 to 0.0082. Villanueva & Moreau (2002) used the same approach to study the decline in Nile perch in the mid 1990s, comparing the potential (steady state) biomass in the 1985-1986 and 1994-1995 periods (Table 5.1). They showed that the overfishing was associated with lower biomass values not only for Nile perch but for Nile tilapia and *Rastrineobola* as well.

**Table 5.1**: Biomass estimates (t km$^{-2}$ yr$^{-1}$) from previous Ecopath models for Lake Victoria.

| | 1971a | 1985a | 1985b | 1995b |
|---|---|---|---|---|
| Adults Nile Perch | 0.0 | 17.2 | 12.02 | 7.65 |
| Juveniles Nile Perch | - | - | 5.06 | 3.83 |
| Catfishes | 2.8 | 1.7 | 1.91 | 1.62 |
| Lungfish | 4.5 | 2.3 | 1.61 | 1.37 |
| Snoutfishes/squeekers | 1.1 | 0.8 | 0.54 | 0.45 |
| Haplochr. Predatory | 3.0 | 0.2 | 0.20 | 0.19 |
| Haplochr. Planktiv. | 1.9 | 0.1 | 0.20 | 0.21 |
| Haplochr. Benthiv. | 3.6 | 0.2 | 0.42 | 0.43 |
| *Rastrineobola* | 7.6 | 7.9 | 10.30 | 6.00 |
| Nile Tilapia | 0.0 | 12.1 | 12.72 | 10.10 |
| Other Tilapias | 2.5 | 0.9 | 0.68 | 0.43 |
| Zooplankton | 10.3 | 10.5 | 10.72 | 10.7 |
| Lake prawn | 2.6 | 5.9 | 5.96 | 5.7 |
| Insects/Molluscs | 28.0 | 28.7 | 25.13 | 23.1 |
| Phytoplankton | 4.3 | 4.5 | 5.13 | 5.0 |
| Benthic Producers | 12.4 | 12.9 | 12.86 | 13.1 |

***Note:*** a=Moreau *et al.* (1993); b=Villanueva & Moreau (2002).

These models track the structure of trophic interactions occurring in the ecosystem before the introduction of Nile perch, at the time of its maximum expansion and in the period of decreasing catches respectively. However, they do not attempt to explain other driving forces behind the changes in ecosystem states. They identify only the increase in Nile perch predation (Moreau *et al.*, 1993) and the impact of overfishing (Villanueva & Moreau, 2002) as the main causes of change in the ecosystem dynamics.

The Nile perch was officially introduced into Lake Victoria in 1962. However, catches became significant only in the late 1970s. In the Kenyan sector of the lake, the year 1980 marks the starting point of the exponential increase in Nile perch landings which rose in few years from virtually zero to almost 60% of total yield (see Figure 1.3). The slow growth of Nile perch in the first two decades has not been explored, but one potentially important factor may have been eutrophication. Nutrient enrichment of the lake was well underway by 1950. Analysis of sediment cores shows that changes in phytoplankton composition, following increased deposition rates of nitrogen (starting in the 1920s) and phosphorus (starting in the 1950s), were clearly underway before the introduction of Nile perch (Hecky, 1993). This process accelerates in the 1980s, when severe algal blooms were observed (Ocumba & Kibaara, 1989; Mugidde, 1993) along with a shift in predominance from diatoms to nutritionally less valued cyanophytes in the phytoplankton community (Gophen *et al.*, 1993, 1995; Hecky, 1993; Mugidde, 1993) that was followed by a gradual shift from large-bodied calanoids and cladocerans to smallbodied cyclopoids occurred in the zooplankton community of Lake Victoria (Wanink *et al.*, 2002).

In this thesis the Ecopath approach is used to construct a steady-state trophic model for the aquatic ecosystem of Lake Victoria in year 1979, just before the Nile perch upsurge. The model focuses on the Kenyan part of the lake only, providing estimates of the biomass of each of $n$ functional groups in the system in that year. These are the same functional groups used in the previous ecopath models (Moreau *et al.*, 1993; Villanueva & Moreau, 2002), with the exception of a different classification for the haplochromine subgroups. In our analysis, these sub-groups include predatory, phytoplanktivores-detritivores and zooplanktivores, following the subaggregation suggested by Witte *et al.* (2000). In addition to fishery data, the model requires information on production/biomass ratios ($P_i/B_i$), consumption biomass ratios ($Q_i/B_i$), ecotrophic efficiencies ($EE_i$), diet composition ($DC_{ji}$) and food assimilation rates for each group, assuming there is no migration ($E_i$) and biomass accumulation ($BA_i$).

Under quite general conditions, the $P/B$ ratio can be approximated by the instantaneous rate of total mortality ($Z$) used by fisheries biologists (Allen, 1971). Different studies in Lake Victoria provide total mortality estimates for adult Nile perch (Acere, 1985), *Rastrineobola argentea* (Manyala, 1993) and indigenous tilapia (Palomares, 1991). The total mortality rates for the other groups are given by Moreau *et al.* (1993), while the rate for juvenile Nile perch was estimated from the 32% growth-efficiency value, which approximates the production/consuption rate in Ecopath, reported by Kitchell *et al.* (1997) for Nile perch up to 2 years of age.

The $Q/B$ ratio is estimated from the empirical relationship proposed by Palomares & Pauly (1989):

$$\log(Q/B) = 7.964 - 0.204\log(W_{inf}) - 1.965T + 0.083A + 0.532h + 0.398d \qquad [5.1]$$

where $W_{inf}$ (or asymptotic weight) is the mean weight that a population would reach if it were to grow indefinitely; $T$ is the mean water temperature expressed as 1000/ (C° + 273.15); $A$ is the aspect ratio of the caudal fin indicative of metabolic activity and expressed as the ratio of the square of the height of the caudal fin and its surface area; '$h$'

and '*d*' are dummy variables indicating herbivores ($h = 1, d = 0$), detritivores ($h = 0, d = 1$) and carnivores ($h = 0, d = 0$). The $W_{inf}$ value is taken from observed asymptotic length ($L_{inf}$) and empirical length-weight relationships for each species. Following Palomares & Pauly's (1999) empirical equation, *Q/B* ratios are estimated from observed biological parameters for adult Nile perch (Ligtvoet & Mkumbo, 1991), juvenile Nile perch (Kitchell *et al.*, 1997), *Rastrineobola argentea* (Manyala, 1993), Nile tilapia (Getabu, 1992), *Oreochromis leucostictus* (Fryer & Iles, 1972) as a proxy for the "Other tilapia" group, and *H. phyrrhocephalus* (Witte & Witte-Maas, 1987) as a proxy for zooplanktivorous haplochromines. Aspect ratio paramaters were obtained from Fishbase (www.fishbase.org) and mean water temperature is assumed at 26 C°. Remaining *Q/B* ratios were taken from Moreau *et al.* (1993).

The ecotrophic efficiency value (*EE*) is defined as the fraction of total production that is used in the system either consumed by predators and fishery, used for biomass accumulation, migration or export. Ecotrophic efficiency is difficult to measure directly. It varies between 0 and 1 but is often assumed to range from 0.65 to 0.95 (Ricker, 1968), approaching 1 for groups with considerable predation/fishing pressure like adult Nile perch in Lake Victoria. Ecotrophic efficiency values for all groups were taken from Moreau *et al.* (1993).

Fish landing data by species are provided by KMFRI and published in Othina (1999) and Othina & Tweddle (1999). Disaggregation of the catch among haplochromine species is obtained from the catch size composition in experimental botton trawls in 1978 (reported by Witte *et al.*, 2000). Catches of juvenile Nile perch, up to 2 years of age, are assumed to be 20% of total Nile perch landings, following Villanueva & Moreau (2002).

The diet composition of each species is defined as the fraction of each prey species consumed to the total consumption. The diet matrix is obtained from results of stomach contents analyses available in the literature for the two groups of Nile perch (Ogari & Dadzie, 1988), catfishes (Greenwood, 1966), lungfish (Ssentongo & Welcomme, 1985), snoutfishes/squeekers (Ssentongo & Welcomme, 1985), haplochromine species (Fryer & Iles, 1972; Greenwood, 1974), *Rastrineobola argentea* (Ssentongo & Welcomme, 1985), Nile tilapia (Trewavas, 1983), other tilapiines (Payne, 1986), zooplankton (Leveque, 1979), lake prawn (Hart & Allanson, 1981), and insects/molluscs (Leveque *et al.*, 1983).

The percentage of ingestion that is not assimilated is assumed at 20% for all groups (Winberg, 1971). The rest of the parameters, e.g. biomass, gross conversion efficiency (equivalent to production/consumption *P/Q*) and mean trophic level are estimated by Ecopath (Table 5.2).

**Table 5.2:** Key features of the Ecopath model for Kenyan waters of Lake Victoria in 1979.

| | **Trophic level** | **P/B ($year^{-1}$)** | **Q/B ($year^{-1}$)** | **Landings ($t\ km^{-2}\ yr^{-1}$)** | **Ecotrophic efficiency** | **P/Q ($year^{-1}$)** | **Biomass ($t\ km^{-2}\ yr^{-1}$)** |
|---|---|---|---|---|---|---|---|
| Nile perch (adults) | 3.68 | 0.85 | 2.95 | 0.83 | 0.75 | 0.29 | 1.37 |
| Nile perch (juveniles) | 3.43 | 2.11 | 6.60 | 0.21 | 0.95 | 0.32 | 0.56 |
| Catfishes | 3.41 | 0.79 | 5.47 | 1.17 | 0.95 | 0.14 | 3.65 |
| Lungfish | 3.23 | 0.30 | 4.84 | 0.12 | 0.95 | 0.06 | 3.60 |
| Snoutfishes/Squeekers | 3.38 | 0.90 | 11.00 | 0.20 | 0.95 | 0.08 | 1.30 |
| Haplochr. Predatory | 3.60 | 1.4 | 8.50 | 0.15 | 0.95 | 0.16 | 2.95 |
| Haplochr. Zooplanktiv. | 3.02 | 2.50 | 19.09 | 0.48 | 0.95 | 0.13 | 2.08 |
| Haplochr. Phyto/Detritiv. | 2.00 | 2.50 | 41.00 | 0.98 | 0.95 | 0.06 | 3.93 |
| Rastrinebola | 2.61 | 3.12 | 26.89 | 2.27 | 0.90 | 0.12 | 5.37 |
| Nile tilapia | 2.07 | 0.65 | 18.28 | 0.23 | 0.95 | 0.04 | 1.35 |
| Other Tilapias | 2.06 | 1.00 | 30.20 | 0.43 | 0.70 | 0.03 | 2.49 |
| Zooplankton | 2.05 | 33.50 | 140.00 | - | 0.70 | 0.24 | 15.17 |
| Lake prawn | 2.48 | 14.00 | 64.00 | - | 0.95 | 0.22 | 3.95 |
| Insects/Molluscs | 2.24 | 5.00 | 25.00 | - | 0.95 | 0.20 | 19.73 |
| Phytoplankton | 1.00 | 365.00 | - | - | 0.95 | - | 6.53 |
| Benthic Producers | 1.00 | 25.00 | - | - | 0.85 | - | 16.52 |
| Detritus | 1.00 | - | - | - | 0.25 | - | 10.00 |

Our estimates of 1.93 t $km^{-2}$ for Nile perch stock biomass are between the 0.022 t $km^{-2}$ baseline value for the period 1969-1971 observed by Kudhongania & Cordone (1974) and the 17.2-17.8 t $km^{-2}$ estimated for 1985 (Moreau *et al.*, 1993; Villanueva & Moreau, 2002), thus representing the conditions before Nile perch upsurge. Haplochromine biomass is also close to its baseline value, suggesting that the model may describe a system still close to its pre-Nile perch steady-state. The estimated haplochromine biomass of 8.96 t $km^{-2}$ does not differ significantly from the 8.5 t $km^{-2}$ estimated by Moreau *et al.* (1993) for year 1971 and is very close to the baseline of 8.81 t $km^{-2}$ observed in 1969-1971 (Kudhongania & Cordone, 1974). Looking at the subgroups, the biomass for predatory haplochromines (32.9%) seems overestimated, while those for zooplkanktivores (23.2%) and especially phyto/detritivores (43.9%) are underestimated compared to the percentages of 18.3%, 27.0 % and 54.7% respectively observed in 1978 (Goldschmidt *et al.*, 1993), and 9.3% 29.6% and 61.1% in 1979 (Witte *et al.*, 2000). This is probably influenced by discrepancies in the diet composition inputs in these two papers, both of which refer to field observations from the late 1960s and early 1970s (Fryer & Iles, 1972; Greenwood, 1974).

The estimated value of 0.002527 for the gross efficiency of the fishery is consistent with the value of 0.001 observed during the 1960s and early 1970s (Gophen *et al.*, 1995) and the values of 0.0016 and 0.0082 observed in 1971 and 1985 respectively (Moreau *et al.*, 1993). The mean trophic level of the catch is 2.84, which is lower than the value of 3.98 estimated in 1971 (Moreau *et al.*, 1993). This is probably due to the catch of zooplanktivorous *Rastrineobola,* which increased from 7.3% of total landings in 1971 to 32.1% in 1979. It may also be evidence of early overfishing of this stock during the late 1970s, which declined from 7.6 t $\text{km}^{-2}$ in 1971 (Moreau *et al.*, 1993) to 5.37 t $\text{km}^{-2}$ in 1979, even though phytoplankton biomass, and therefore the productivity of the system, increased from 4.30 t $\text{km}^{-2}$ to 6.53 t $\text{km}^{-2}$, and even though the biomass of *Rastrineobola*'s main predators remained almost constant.

## 5.3. Biomass dynamics simulation using Ecosim

In order to test the impact of eutrophication on the freshwater ecosystem in Lake Victoria an Ecosim (Walters *et al.*, 1997, 2000) dynamic simulation was run for the period 1980-1998 using estimates from the Ecopath model of 1979. The model simulation depends on the trend in relative fishing effort. This trend was obtained from the available dataset covering the period 1989-1998 (Othina & Tweddle, 1999). We estimated the output price elasticity of fishing effort at time *t* from a log-log regression using prices in nominal terms rather than real deflated values ($R^2$=.95 against $R^2$=.44):

$$\underset{(23.253)}{\ln E_t = 4.888} + \underset{(12.246)}{0.263 \ln p_t} \qquad [5.2]$$

The assumption here is that prices at time *t* are a measure of the expectations of fishers at the beginning of the year, according to which they choose the amount of fishing effort at time *t*. Using this equation fishing effort may be estimated for the periods for which there are no data (Table 5.3).

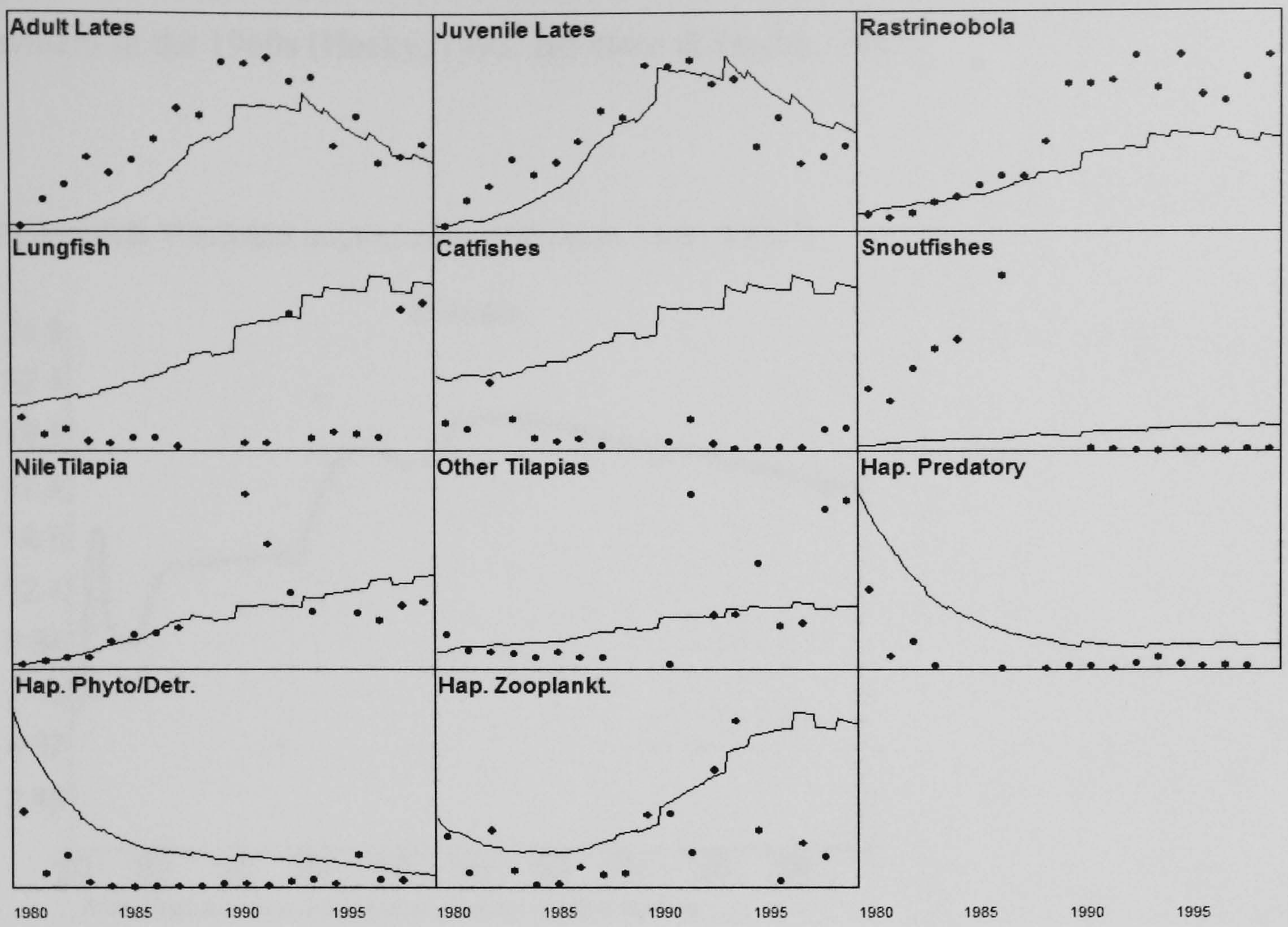
**Table 5.3:** Estimated fishing effort from the price elasticity equation.

| Year | Observed fishing effort ('000 boat days)[b] | Estimated fishing effort ('000 boat days)[a] | Constant=1987 ex-vessel prices (Ksh tonn$^{-1}$)[c] | Nominal ex-vessel prices (Ksh tonn$^{-1}$)[c] |
|---|---|---|---|---|
| 1974 | | 929 | | 1,632 |
| 1975 | | 942 | | 1,723 |
| 1976 | | 941 | | 1,716 |
| 1977 | | 948 | | 1,768 |
| 1978 | | 988 | | 2,071 |
| 1979 | | 1037 | | 2,488 |
| 1980 | | 1082 | 5966 | 2,918 |
| 1981 | | 1088 | 5460 | 2,987 |
| 1982 | | 1060 | 4102 | 2,703 |
| 1983 | | 990 | 2832 | 2,079 |
| 1984 | | 1036 | 3064 | 2,479 |
| 1985 | | 1077 | 3139 | 2,872 |
| 1986 | | 1111 | 3394 | 3,228 |
| 1987 | | 1152 | 3701 | 3,701 |
| 1988 | | 1248 | 4481 | 5,032 |
| 1989 | 1202 | 1192 | 3314 | 4,225 |
| 1990 | 1387 | 1487 | 6629 | 9,778 |
| 1991 | 1496 | 1503 | 5778 | 10,192 |
| 1992 | 1606 | 1533 | 4889 | 10,976 |
| 1993 | 1862 | 1799 | 6158 | 20,185 |
| 1994 | 1862 | 1919 | 6116 | 25,822 |
| 1995 | 2007 | 1917 | 6001 | 25,720 |
| 1996 | 1971 | 2107 | 7884 | 36,825 |
| 1997 | 2008 | 1950 | 5284 | 27,443 |
| 1998 | 2190 | 2146 | 7114 | 39,457 |
| 1999 | | 2094 | 6131 | 35,947 |
| 2000 | | 2136 | 6008 | 38,752 |

**Sources:** a) from estimated equation $E_t = \exp(4.888)*p_t^{0.263}$ b) Othina & Tweddle (1999), c) adapted from Republic of Kenya: Statistical Abstracts (1980-1993), and Republic of Kenya: Fisheries Bulletins (1997-2000).

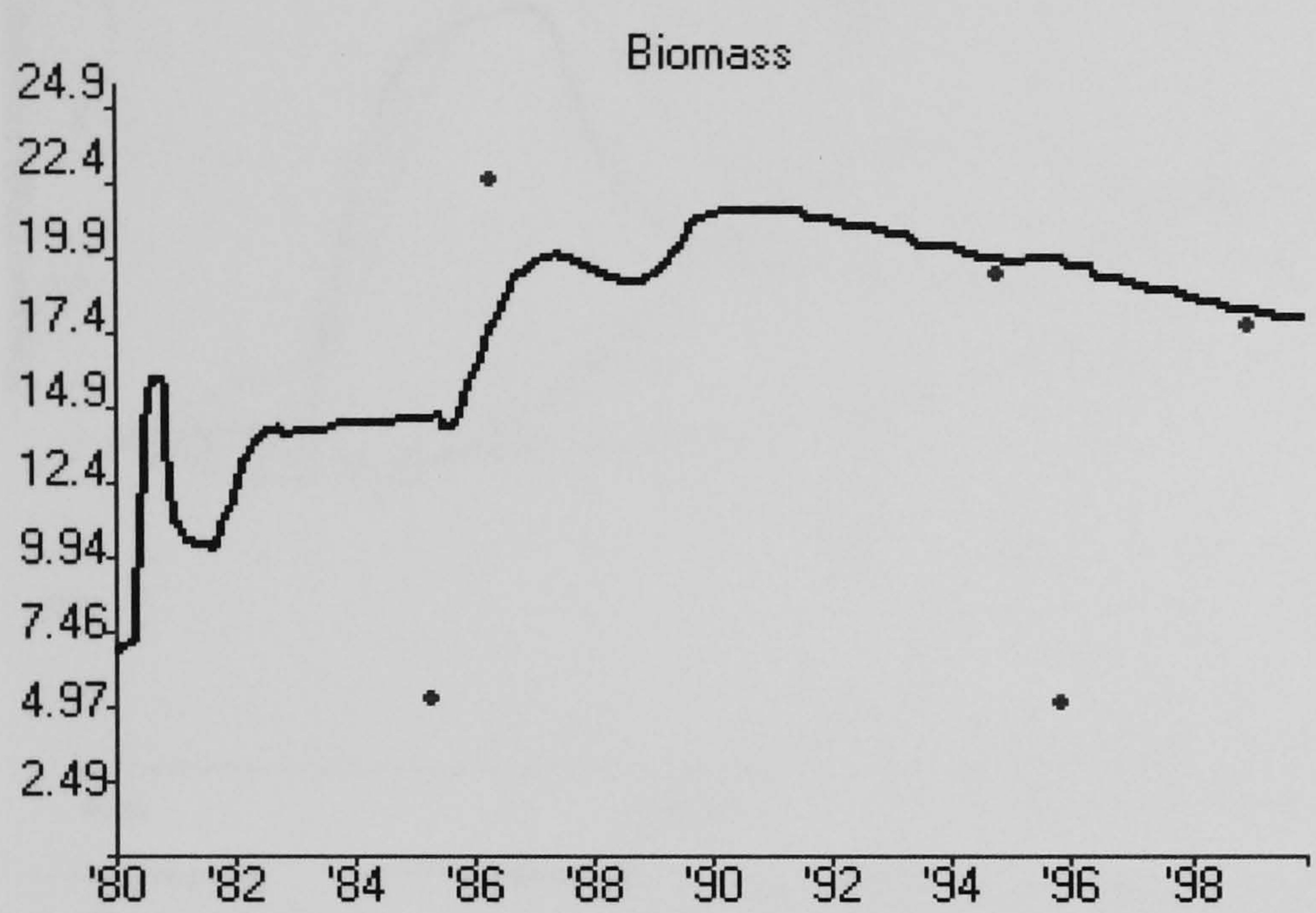
Using the Runge-Kutta 4th order integration method, the Ecosim model is fitted to fishery time series. This yields estimates for both vulnerability and annual relative primary productivity that may represent historical productivity "regime shifts" impacting biomasses throughout the ecosystem. The trend in fishery yields predicted by the fitted model (SS = 277.35) are shown in Figure 5.1 for comparison with observed landings. The fitted values for adult Nile perch are in general lower than the observed values. This trend is partially compensated by higher values for juvenile Nile perch during the 1990s, suggesting that the initial assumption of 20% Nile perch yields represented by juveniles has changed over the time. Our model predicts a progressively higher proportion of juveniles on overall Nile perch landings which may be an evidence of overfishing.

**Figure 5.1:** Model fit comparison between predicted and observed (dots) yields.



The model predicts an abrupt rise of phytoplankton biomass in 1980 (Figure 5.2). Our hypothesis is that the shift in primary productivity was caused by an exeptional climatic event, accelerating the eutrophication process which started in the 1920s (Hecky, 1993; Bootsma & Hecky, 1993). In 1977-1980 heavy rains, often associated with "El Niño" events, increased the lake level by 1.5 m (Kite, 1981). During this (and other similar) events almost all the rooted vegetation was permanently submerged and eventually rotted away. Floating vegetation, such as papyrus swamps, was either washed ashore or disintegrated through wave action compromising the nutrient retention function of wetlands during that period. Land vegetation in newly submerged areas also died. As result of the inundation of land, large loads of sediments and nutrients where pumped into the lake. Moreover, these water level perturbations could have modified near-shore habitats and changed breeding area available to many fish species (Lung'ayia *et al.*, 2001). Also Fryer & Iles (1972) consider that changes in water levels may have important implications for fish speciation and possibly fish production. In the absence of this environmental shock the sudden upsurge of Nile perch during the 1980s is difficult to explain. It may also clarify why the biomass of this newly introduced predator did not increase during the previous two decades, even if the eutrophication process was already evident in the 1960s (Hecky, 1993; Bootsma & Hecky, 1993).

**Figure 5.2:** Predicted phytoplankton biomass trend (t km$^{-2}$).

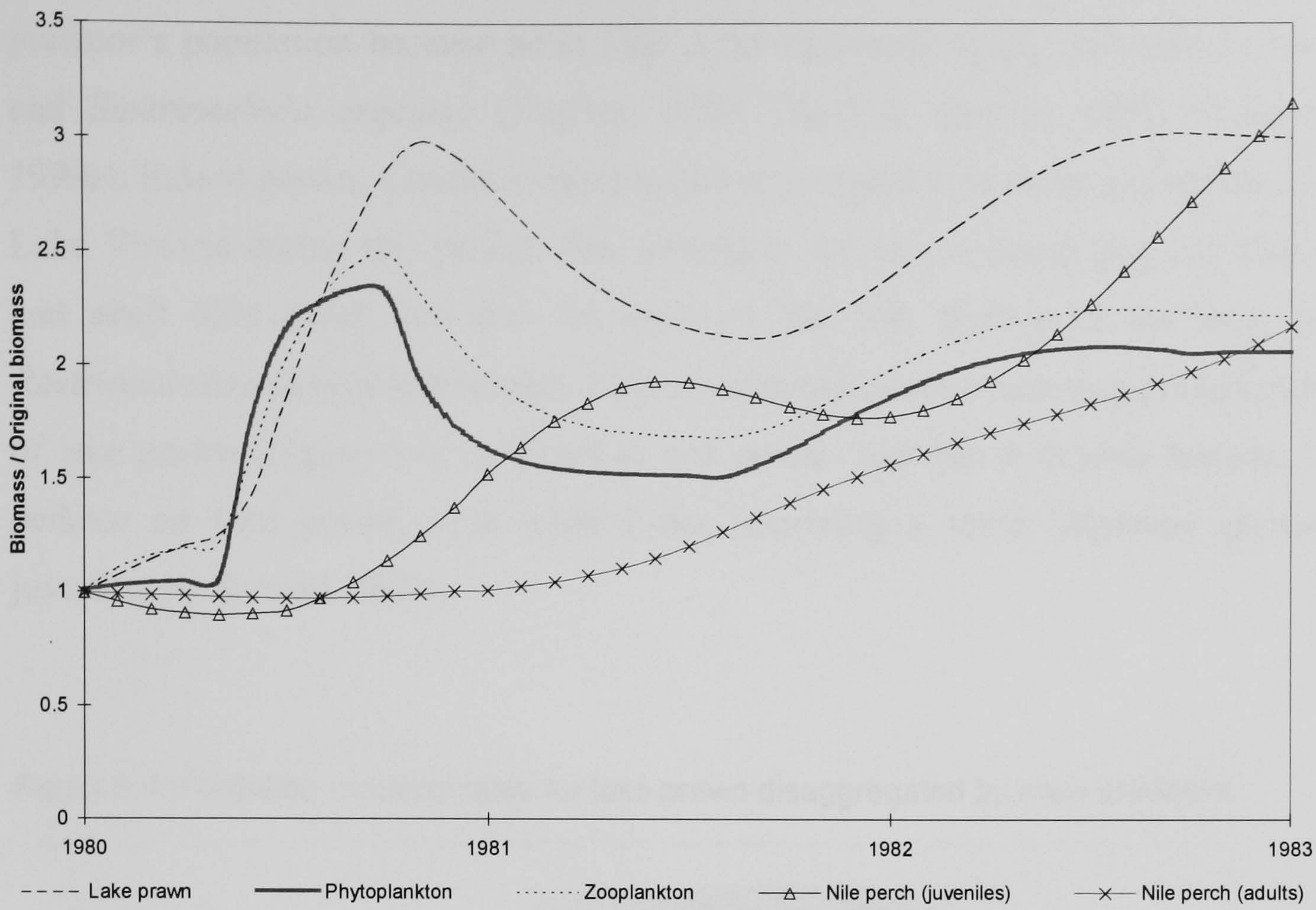


*Note:* Dots are values estimated or observed by other authors.

As shown in Figure 5.2, our estimates are considerably higher than phytoplankton biomass values estimated by other authors in 1985 (Moreau *et al.*, 1993) and 1996 (Villanueva & Moreau, 2002) using the Ecopath approach. Our findings however, closely approximate values observed in Kenyan inshore waters in 1986 (Ochumba & Kibaara, 1989), 1995 (Kenyanya, 1999) and 1998 (Lung'ayia *et al.*, 2000) respectively, suggesting that previous Ecopath models for Lake Victoria underestimated the impact of nutrient enrichment on the freshwater ecosystem.

In our simulation the phytoplankton biomass peak in 1980 produces a cascading effect through the food web leading to Nile perch upsurge and a general increase in fish biomass with upwards trends for *Rastrineobola* and Nile tilapia as well (Figure 5.3 and 5.6). On the other hand, haplochromine species decrease sharply (Figure 5.7a). The abrupt rise in phytoplankton biomass also produced an upwards shift in lake praw biomass through the zooplankton community (Figure 5.3).

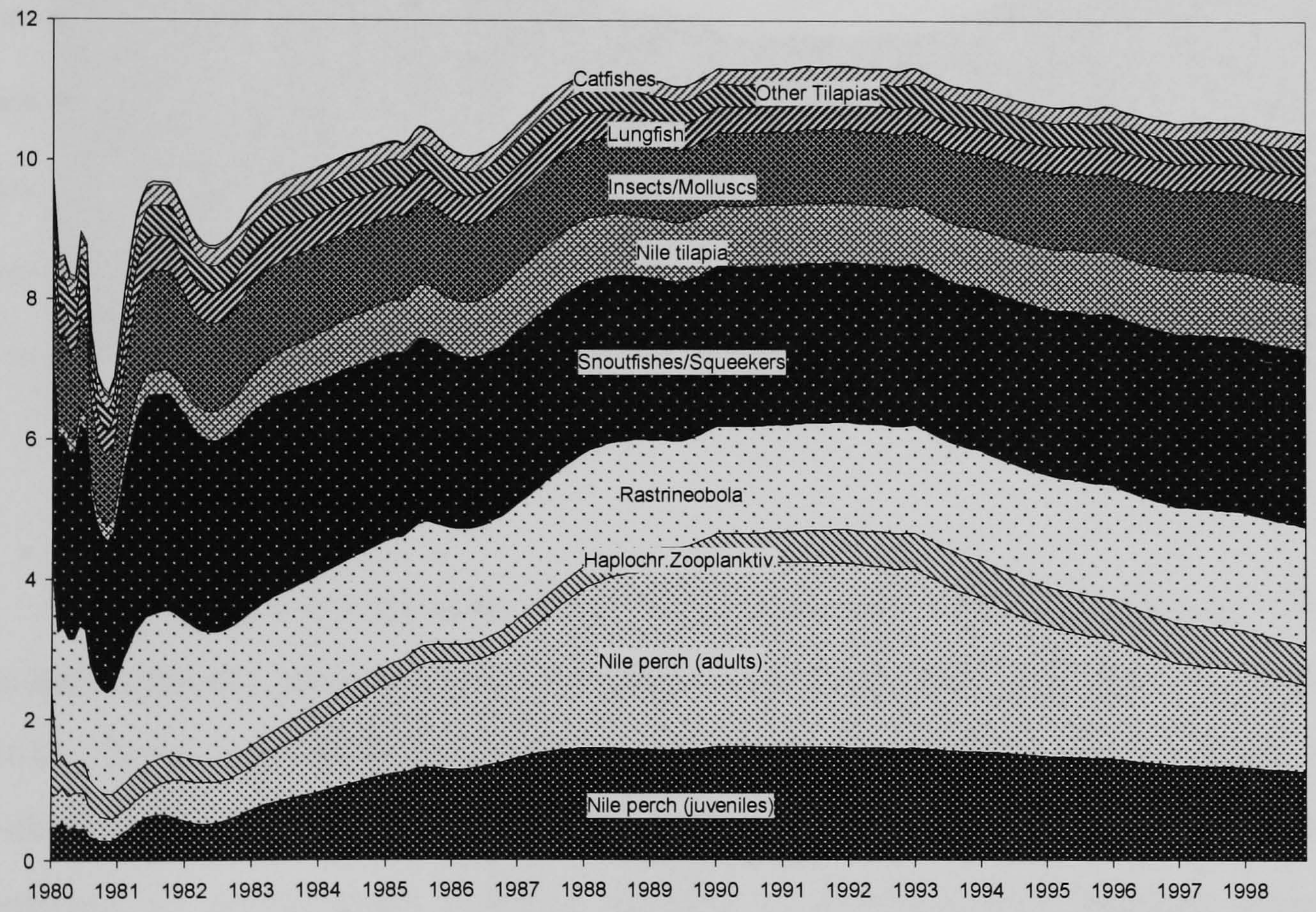
**Figure 5.3:** Estimated bottom-up ecosystem dynamics in the early 1980s.



Several field experiments have yielded data that support the model projections for prawn, *Cardinia nilotica,* during the 1980s (Witte *et al.*, 1992; Goldschmidt *et al.*, 1993; Kaufman & Ochumba, 1993). Increased abundance of lake prawn generated a food surplus. This favored the survival of juvenile Nile perch, feeding largely on invertebrates (Ogutu-Ohwayo, 1990b) and which were until that time outcompeted for this primary food source by the high density of haplochromines. Although the haplochromine community was trophically diverse, it had a limited ability to respond to the rapidly increasing food supply induced by increased nutrient loading. Haplochromines have a K-selected life history (Witte *et al.*, 1992). This life history contrasts strongly with the r-selected invertebrates and Nile perch. According to their life history, the ability of species to respond to a rapid increase in food supply is quite different and rapid eutrophication will favor the r-selected species (Odum, 1985). This picture is consistent with the hypothesis formulated by Hecky (1993), in which increasing turbidity and deoxygenation, caused by eutrophication, broke down the food web through a combination of demersal haplochromines displacement and increased invertebrate abundance.

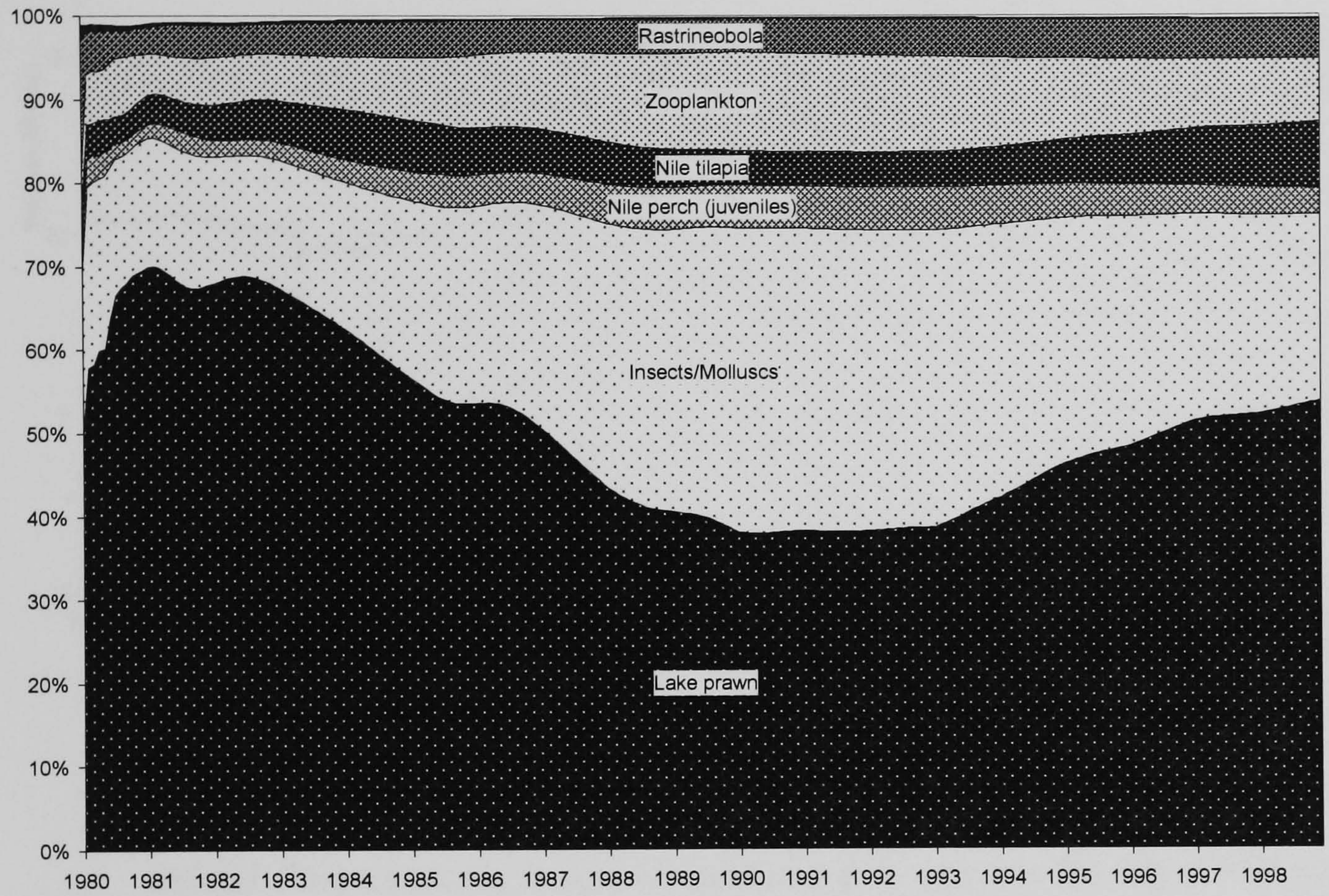
Increased survivorship of juvenile Nile perch led to explosive growth of the adult population which then decimated the haplochromines. The relative disappearance of haplochromines in the 1980s did not induce a corresponding decrease in their main predator's population because adult Nile perch opportunistically switched to lake prawn and *Rastrineobola argentea* (Hughes, 1986; Ogari & Dadzie, 1988; Ogutu-Ohwayo, 1990a). Indeed prawn, *Cardinia nilotica*, played a crucial role in the ecological changes of Lake Victoria during this period. This shrimp is not only a major prey for both juvenile and adult Nile perch but also for most of the fish species in the lake, including *Rastrineobola* and haplochromines. This is reflected by the increasing predation mortality of lake prawn (Figure 5.4) predicted in our model. Nile perch quickly became the main pedator on lake prawn, with adult *Lates* becoming a more important predator than juveniles by the mid 1980s.

Figure 5.4: Predation mortality rates for lake prawn disaggregated by main predators.



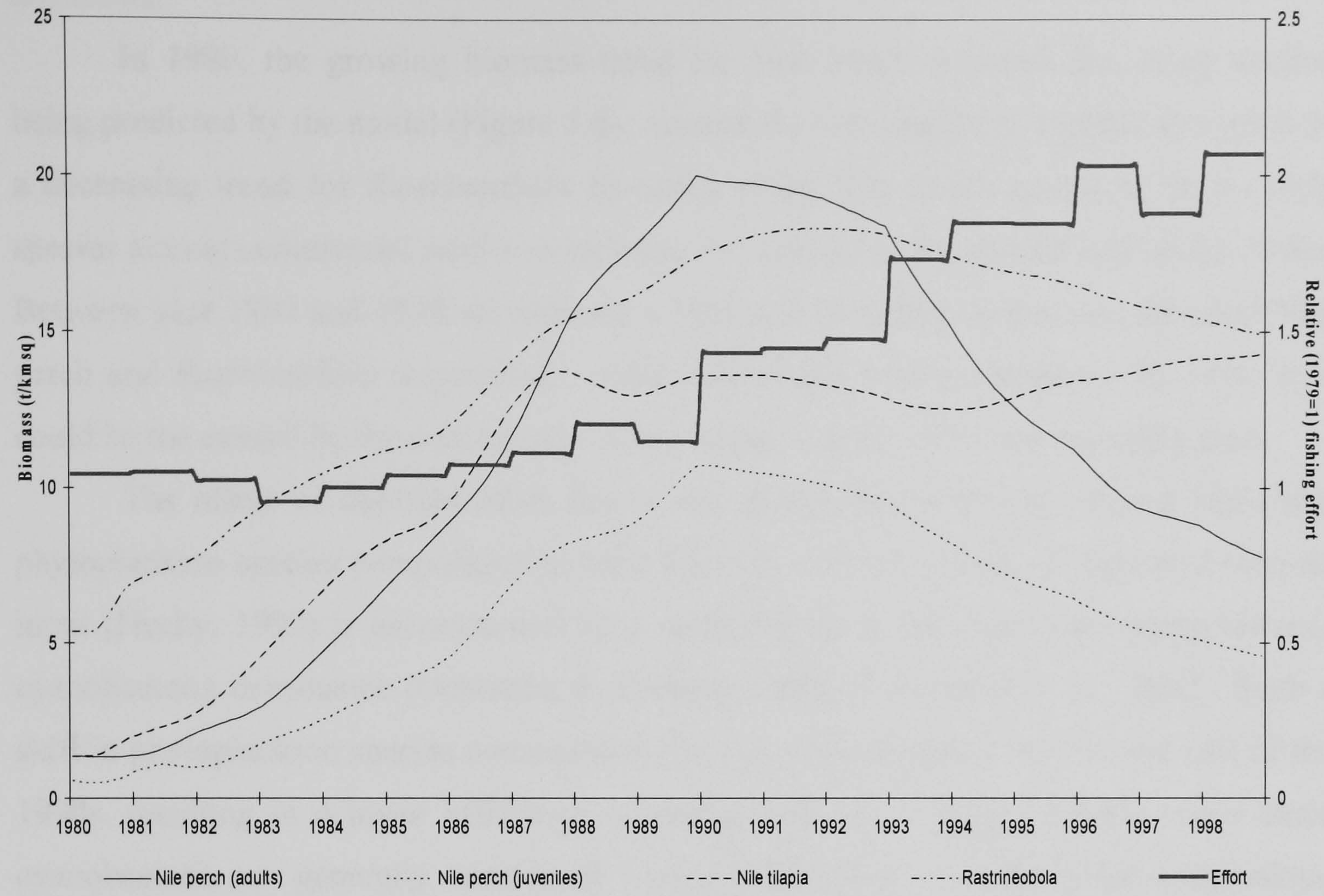
At this point juvenile Nile perch then increased the proportion of insects and molluscs in their diet (Figure 5.5) to compensate for the increased predation pressure of adult *Lates* on their main prey. It may be noticed that, despite heavy predation, the density of lake prawn increased after the Nile perch boom. The eutrophic condition of Lake Victoria may explain this phenomenon. It has been hypothesized that the increased hypoxia in the lower part of the water column for prolonged periods (Hecky, 1993) provided a refugium for *Cardinia nilotica* to avoid Nile perch predation (Kaufman & Ochumba, 1993; Branstrator & Mwebaza-Ndawula, 1998). Under these conditions the stocks of both lake prawn and Nile perch were able to coexist and grow.

**Figure 5.5:** Estimated diet composition trends for juvenile Nile perch.



Looking at the biomass trend for the main commercial fish stocks (Figure 5.6), the model correctly predicts both the increase of Nile perch during the 1980s, and its peaking at the beginning of the 1990s. Both *Rastrineobola* and Nile tilapia follow a similar path which is also consistent with other studies referring to that period. Trawl survey data indicate an approximate fourfold increase in *Rastrineobola* biomass during the 1980s (Wanink, 1991; Witte *et al.*, 1992b), while trophic mass-balance modeling, comparing situations at pre- and post-Nile perch expansion, also indicates a biomass increase for both *Rastrineobola* (Villanueva & Moreau, 2002) and Nile Tilapia (Moreau *et al.*, 1993). The biomass of both increased during the Nile perch upsurge even if these species occupied a lower tropic level (2.07 and 2.61 for Nile tilapia and *Rastrineobola* respectively against 3.68 for adult Nile perch) and were amongst the species preyed upon by *Lates niloticus*.

**Figure 5.6:** Estimated stock biomass trend and observed relative (1979=1) fishing effort for main commercial fisheries in Kenian waters of Lake Victoria.



Indeed *Rastrineobola* is one of the main prey species for adult Nile perch (Ogutu-Ohwayo, 1990b; Mkumbo & Ligtvoet, 1992), although is not the favorite prey and does not itself favor the preferred habitat of Nile perch (Kaufman & Ochumba, 1993). In the case of Nile tilapia, the explanation for its growth along with the Nile perch may lie in its rapid growth rate and large adult size (more than 30 cm.) (Kitchell *et al.*, 1997). Moreover, Nile tilapia has a comparative advantage over the other tilapiine species in utilizing the increased productivity of the system during the 1980s. This has been attributed to several factors including its ability to exploit a wider range of food materials, faster growth rates and extended spawning periods (Fryer & Iles, 1972).

*Rastrineobola*, on the other hand, reacted to increased predation pressure by Nile perch by increasing its growth rate, reducing its age at first maturity and reaching a smaller adult size (Wanink, 1996). According to Wanink *et al.* (2001), this was probably facilitated by the displacement of demersal haplochromines, allowing *Rastrineobola* to exploit the bottom area, switching from zooplankton to energetically richer macrobenthic prey (insects larvae and lake prawn), which increased strongly in abundance after Nile perch boom (Witte *et al.*, 1995). Moreover, a shift from juvenile to adult mortality has probably increased the number of fish recruiting to the reproducing population (Wanink *et al.*, 1999).

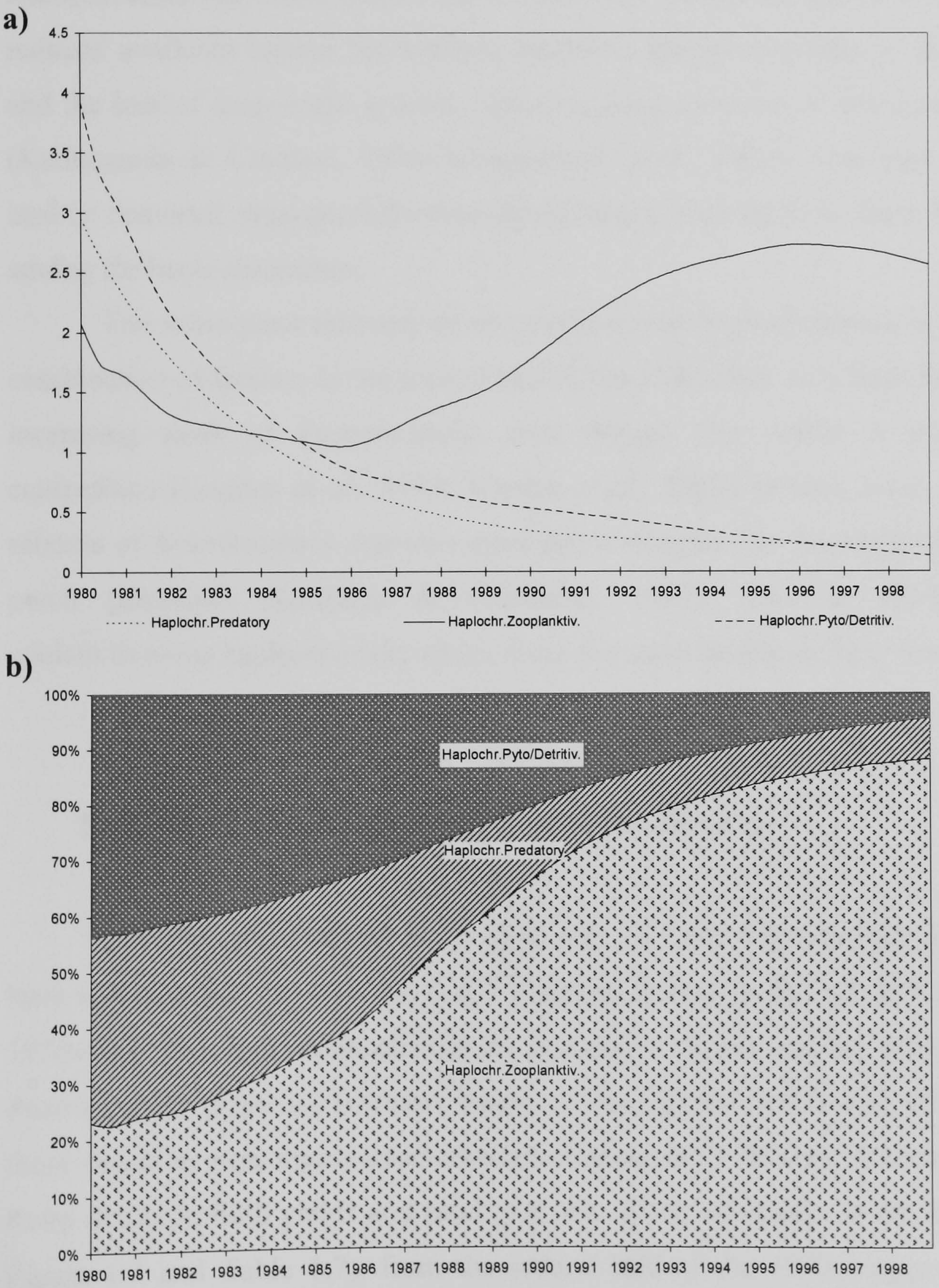
In 1990, the growing biomass trend for Nile perch reversed, the sharp decline being predicted by the model (Figure 5.6). Around the same period the model also predicts a decreasing trend for *Rastrineobola* biomass, while Nile tilapia seems to be the only species among commercial stocks to increase its biomass in the second half of the 1990s. Between year 1992 and 1998 we estimate a 58% and 18% drop in biomass for adult Nile perch and *Rastrineobola* respectively, while Nile tilapia biomass increases by 11%. This could be the caused by the concurrence of increasing fishing effort and eutrophication.

The effect of eutrophication lies in the relationship between nutrient input and phytoplankton species composition in Lake Victoria, where evidence of increased nutrient input (Hecky, 1993) is accompanied by a restructuring of the algal community towards cyanobacteria dominance (Ochumba & Kibaara, 1989; Verschuren *et al.*, 2002). Such a shift in phytoplankton species composition may have accelerated in the second half of the 1990s, resulting in a lower efficiency of energy transfer to higher trophic levels since cyanobacteria are generally considered a poor food source, especially for zooplankton (Lampert, 1981; Heerkloss *et al.*, 1984; Haney, 1987). Moreover, in an eutrophic environment with strongly linked food web interactions, biomass may accumulate at weakly coupled trophic links (Pitcher *et al.*, 1996). In these environments a significant proportion of phytoplankton production may be diverted to pathways, including sediments, which contribute little to fish production (Riemann & Christoffersen, 1993).

In addition to direct losses of algal production to anoxic sediments, a substantial amount of zooplankton production in Lake Victoria is probably diverted into a non-fish pathway: insect larvae tolerant of anoxia. These insects move into the upper layers of the lake at night and consume large quantities of zooplankton, but escape being eaten by fish through daytime vertical migration into the anoxic layers (Pitcher *et al.*, 1996). Such an environment would not be favorable for zooplanktivorous fish such as *Rastrineobola* and zooplanktivorous haplochromines (Pitcher *et al.*, 1996). On the other hand, the ability of Nile tilapia to digest cyanobacteria (Moriarty, 1973; McDonald, 1987) suggests that eutrophication may result in a more productive fishery dominated by this specie.

The model correctly predicts the sharp deline of haplocromine biomass during the 1980s (Figure 5.7a). However, haplochromine zooplanktivores, which were the least abundant during the 1980s, subsequently started to increase their biomass eventually becaming the most abundant among the cichlids (Figure 5.7b). This trend is consistent with the partial recovery of some haplochromine species observed during the 1990s (Seehausen *et al.*, 1997). The resurgence of zooplanktivorous cichlids predicted by the model has also been observed in the field. In the late 1970s, for instance, about 60% of the haplochromine biomass consisted of phytoplanktivores/detritivores against 30% of zooplanktivores (Goldschmidt *et al.*, 1993; Witte *et al.*, 2000). By 1997 this proportion had changed to 15% and 84% for phytoplanktivores/detritivores and zooplanktivores respectively (Witte *et al.*, 2000).

**Figure 5.7:** Estimated stock biomass trend (a) and species composition (b) for haplochromines.



Since the Nile perch population boom and haplochromine crash correlate in time, it has been argued that the decrease in haplochromine biomass during the 1980s was mainly due to Nile perch predation (Ogutu-Ohwayo, 1990a). It is now clear, however, that Nile perch is but one of many causes of haplochromine decline, most or all of which are anthropogenic (Kaufman, 1992). Indeed, decimation of native species is coincident with the combined effect of alien introduction (Ogutu-Ohwayo, 1990a), eutrophication (Ochumba, 1990), unusually rapid and extreme fluctuations in lake level (Welcome, 1970; Kite, 1981) and other profound alterations in the food web and ecosystem dynamics of the lake (Kaufman, 1992; Witte *et al.*, 1992b; Gophen *et al.*, 1993). Amongst these causes, eutrophication may have played the crucial role. Increasing anoxia in the deeper waters reduced available habitat for hypoxia intolerant species resulting in distributional shifts and the loss of deep-water species, including large numbers of demersal haplochromines (Kudhogania & Cordone, 1974; Hoogerhoud *et al.*, 1983). This may explain why the mostly demersal phytoplanktivorous/detritivorous cichlids have been the most affected among the haplochromines.

The subsequent recovery of zooplanktivorous haplochromines is probably due to a combination of factors. In the second half of the 1980s they may have benefitted from the increasing stock of *Rastrineobola*, even though this cichlid is one of their main competitors (Gophen *et al.*, 1995; Wanink *et al.*, 2002). In fact, it has been noticed that schools of *Rastrineobola argentea* represent a refugium for haplochromines against Nile perch predation (Kaufman & Ochumba, 1993). This is particularly true for zoolanktivorous haplocromines which share the same habitat as *Rastrineobola*.

## 5.4. The equilibrium analysis of the fishery

The dramatic changes in the food web of Lake Victoria during the last 40 years have also radically changed the lake's fisheries from a focus on native species before the 1970s to a focus on introduced Nile perch and Nile tilapia, and the native pelagic cyprinid *Rastrineobola argentea* (Oguto-Ohwayo 1990a) in the 1980s and 1990s. Even though these fisheries were highly productive throughout the 1980s and first half of the 1990s, a sharp deline in both yields and catch per unit effort, especially in the Kenyan sector (see Figure 1.2 and Table 1.7), from the second half of the 1990s suggests that maximum sustainable yields were exceeded and the fishery was moving to a state of stock depletion (Ogutu-Ohwayo *et al.*, 1998). As things stand, there is a serious risk that commercially important stocks will collapse unless urgent action is taken to improve their management.

Using fishery statistics and published trends in fishing effort for the four main fishing technologies (Othina & Tweddle, 1999), we analyze a number of fishery scenarios in the Kenyan waters of Lake Victoria during the 1990s. In doing so, we perform an equilibrium analysis given the base values of a new Ecopath model for year 1989. This uses updated input values reflecting the change in biological parameters during the 1990s. These parameters better describe the change in the Lake Victoria freshwater ecosystem to a new state of high productivity.

Diet composition as well as consumption/biomass (*Q/B*) and ecotrophic efficiency (*EE*) inputs are taken from (Villanueva & Moreau, 2002), with the exeption of the phytoplankton ecotrophic efficiency, which we set to a value of 0.6 given the high level of eutrophication during the 1990s. We set production/biomass input values of *P/B* = 1.1 for adult Nile perch (Ligtvoet & Mkumbo, 1990), *P/B* = 3.5 for juvenile Nile perch (Villanueva & Moreau, 2002), *P/B* = 2.47 for Nile tilapia (Getabu, 1994) and *P/B* = 3.59 for *Rastrineobola* (Wanink, 1989), keeping the production/consumption (*P/Q*) values from our previous Ecopath model for all the other groups.

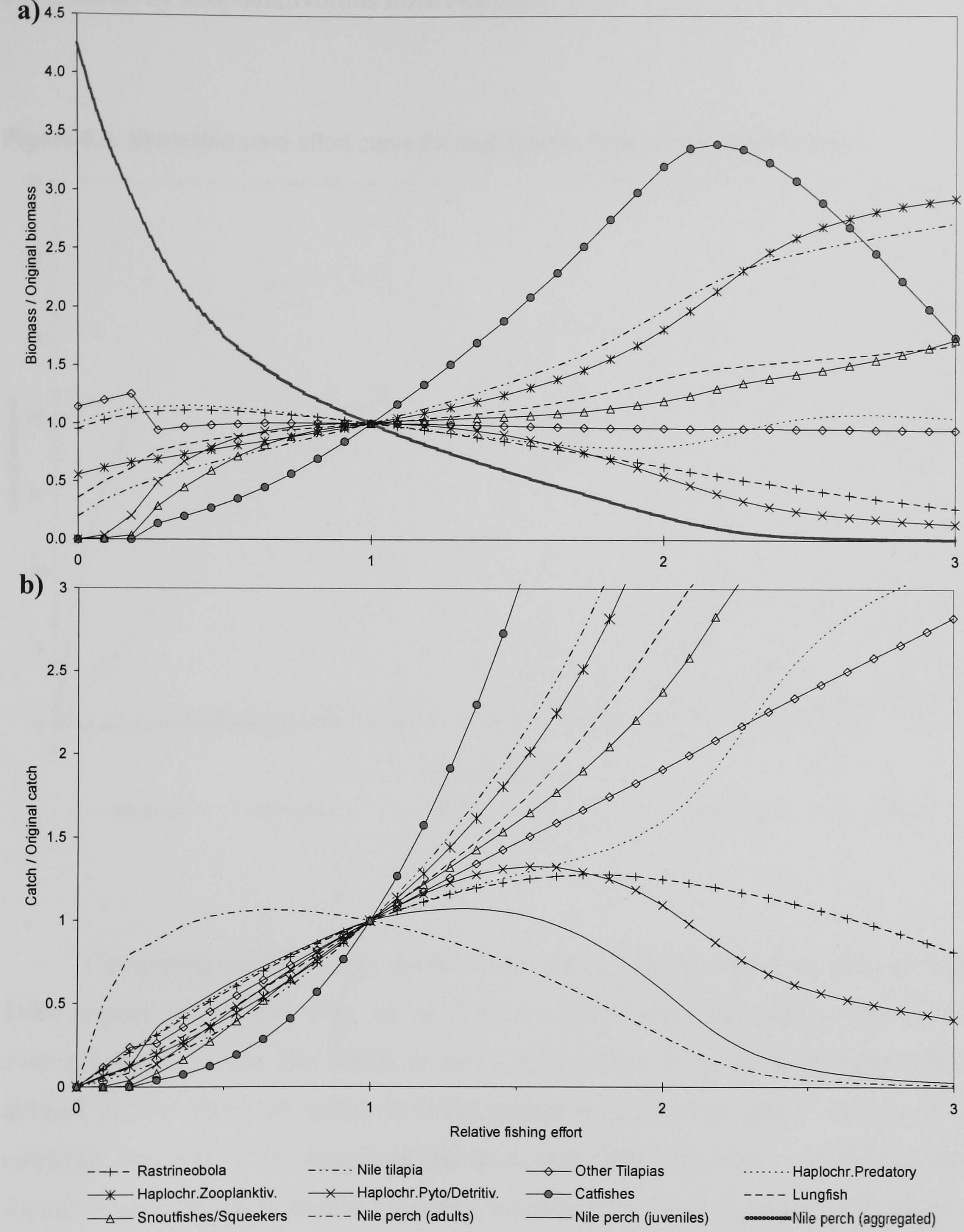
Fish landing data for year 1989 were disaggregated among the four gears considered, i.e. gillnets, beach seines, mosquito seines and longlines (Table 5.4). For Nile perch we use data on percentage contributions of fishing gears to Nile perch landings in the Kenyan waters of Lake Victoria during the period 1987-1989 (Asila *et al.*, 1990): 27% gillnets, 26% beach seines, 25% mosquito seines and 22% longlines. Catches of juvenile Nile perch are obtained from data on relative selectivity and percentage of juvenile harvesting for each gear reported by Ikiara (1999). Nile tilapia landings are disaggregated according to the percentage distribution reported by Asila *et al.* (1990). The other groups are assigned to a specific fishing gear according to information on main target species reported by Witte & van Densen (1995).

**Table 5.4:** Fish landings (t km$^{-2}$) by species and fishing gear for Kenyan waters of Lake Victoria in year 1989.

| | Longlines | Beach seines | Gillnets | Mosquito seines | TOTAL |
|---|---|---|---|---|---|
| Nile perch (adults) | 5.920 | 5.700 | 7.380 | | 19.00 |
| Nile perch (juveniles) | 0.459 | 1.840 | 0.446 | 7.250 | 10.00 |
| Catfishes | 1.040 | | | | 1.04 |
| Lungfish | 0.200 | | | | 0.20 |
| Snoutfishes/Squeekers | 0.197 | | | | 0.20 |
| Haplochr. Predatory | | | | 0.005 | 0.01 |
| Haplochr. Zooplanktiv. | | | | 0.524 | 0.52 |
| Haplochr. Phyto/Detritiv. | | | | 0.016 | 0.02 |
| Rastrinebola | | | | 20.120 | 20.12 |
| Nile tilapia | | 0.393 | 0.321 | 0.476 | 1.19 |
| Other Tilapias | | 0.219 | | | 0.22 |
| TOTAL | 7.82 | 8.15 | 8.15 | 28.39 | 52.51 |

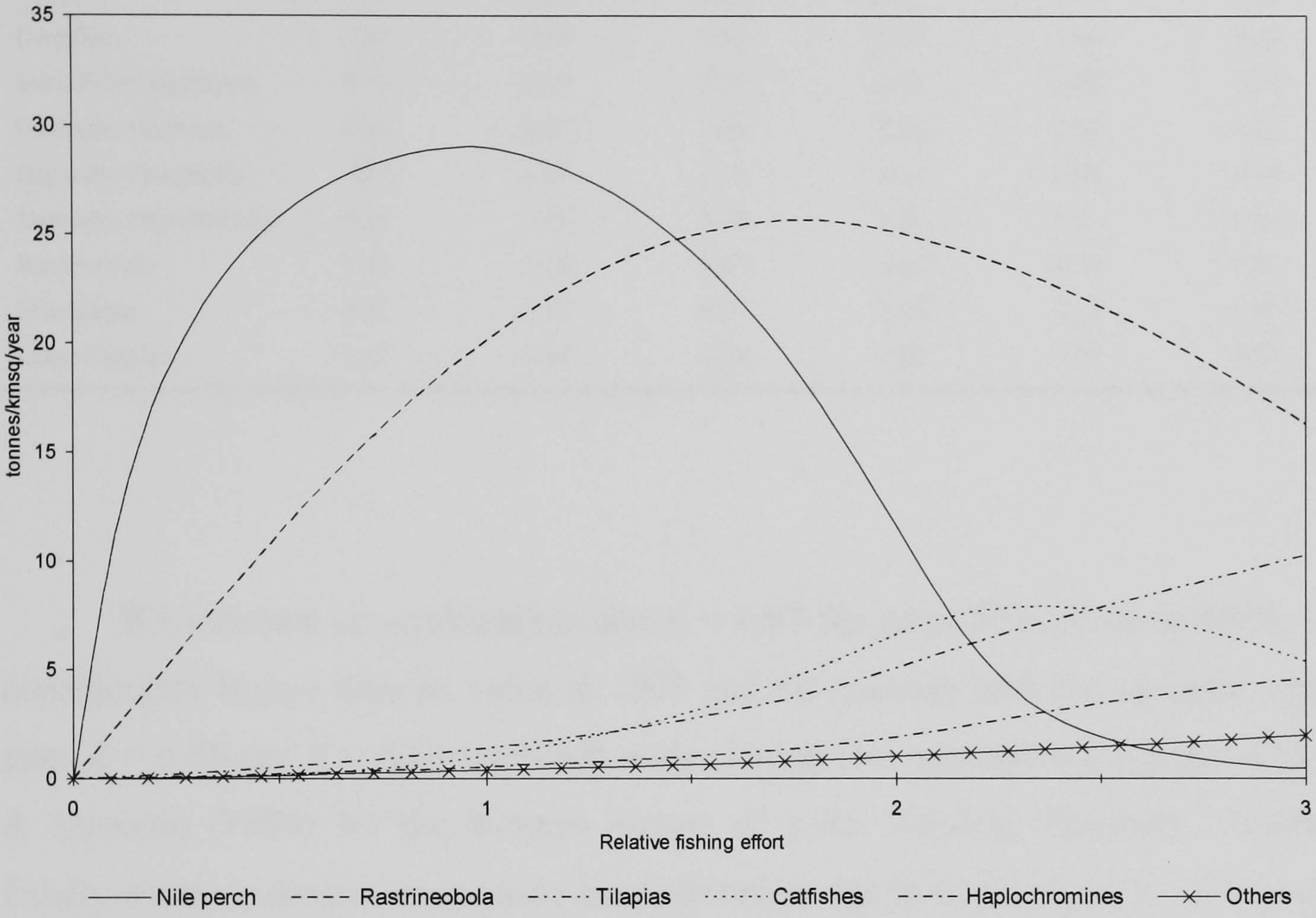
The equilibrium analysis takes the partial derivatives of the differential equations which define Ecosim ($dB/dt = f$(fishing rate, predation rate, etc.)) with regard to fishing mortality ($\delta B/\delta F$), and sets these equal zero to identify the biomass values that would result from the continued application of the different levels of fishing mortality ($F$). The results of the equilibrium analysis are shown as a plot, centered about the Ecopath baseline value of relative fishing effort ($F = 1$) referring to year 1989 (Figure 5.8a). Fishing effort higher than its equilibrium value would be reflected by a marked decrease in Nile perch biomass, which approaches zero at around $F = 2.5$. *Rastrineobola* is also affected by higher relative fishing effort, even though its biomass eventually stabilizes at a lower level (showing some resilience to high fishing pressure). All the other species benefit from increasing fishing effort due to lower Nile perch predation, exept for haplochomine phytoplanktivores/detritivores affected by the increased biomass of other predators (i.e. predatory haplochromines). Nile perch yields are relative stable up to a 50% increase in fishing effort, with juvenile Nile perch catches making up for lower yields of adults (Figure 5.8b). Under high fishing pressure *Rastrineobola* yields are not affected until fishing effort is doubled, while the Nile tilapia fishery replaces the collapsing Nile perch fishery as the high value commercial fishery.

**Figure 5.8:** Equilibrium analysis for relative biomass (a) and relative yields (b) of multispecies fisheries in Kenyan waters of Lake Victoria.



Given the equilibrium aggregate fishing effort in 1989 at 1,192 thousand/boat/day, the estimated MSY for Nile perch is attained at the equilibrium level $F = 1$ with 29 t km$^{-2}$ yr$^{-1}$ (Figure 5.9). *Rastrineobola* reaches its MSY of 25.7 t km$^{-2}$ yr$^{-1}$ at $F = 1.72$. All other species are predicted to attain their MSY at a much higher level of relative aggregate fishing effort, benefiting from the reduced stock of the predator Nile perch and/or lower competition by zooplanktivorous *Rastrineobola*.

**Figure 5.9:** Estimated yield-effort curve for multispecies fisheries of Lake Victoria.



Comparing the estimated exploitation rates of our Ecopath models in 1979 and 1989 respectively (Table 5.5), our analysis confirms that Nile perch stocks have been overexploited since the late 1980s. In terms of mortality rates, the exploitation rate ($E$) is defined as: $E = F/(F+M)$; where $M$ is the natural mortality rate and $F$ the rate of fishing mortality. Gulland (1971) suggested that in an optimally exploited stock, fishing mortality should be about equal to natural mortality, resulting in a fixed $E_{opt} = 0.5$. This value is still used widely but has been shown to overestimate potential yields in many stocks by a factor of 3-4 (Beddington & Cooke 1983). For small tropical fishes with high natural mortality the exploitation rates at maximum sustainable yield ($E_{msy}$) may be unrealistically high.

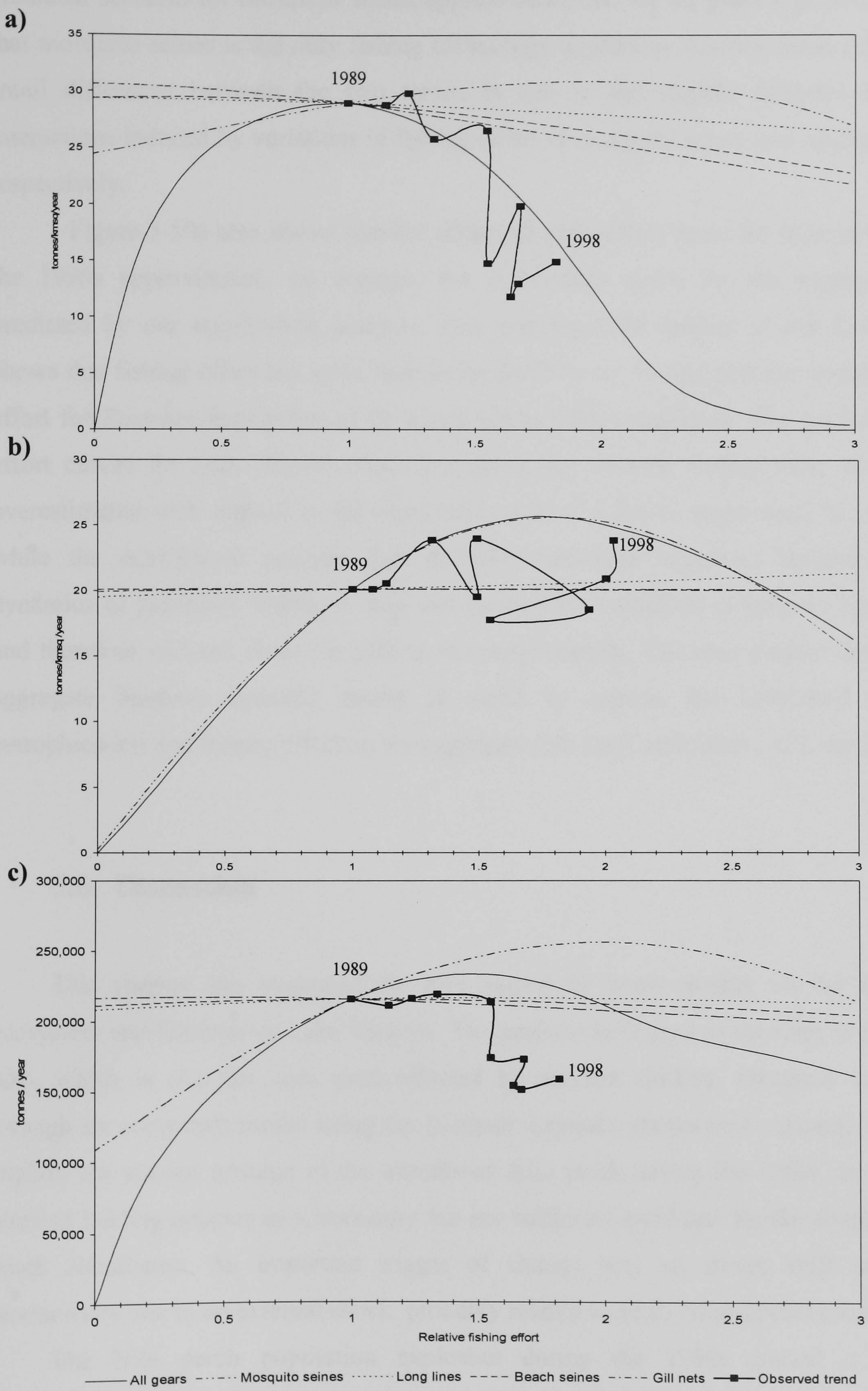
**Table 5.5**: Estimated mortality and exploitation rates.

| | Fishing mortality rate | | Predation mortality rate | | Exploitation rate | |
|---|---|---|---|---|---|---|
| | 1979 | 1989 | 1979 | 1989 | 1979 | 1989 |
| Nile perch (adults) | 0.59 | 1.07 | 0.01 | 0.01 | 0.69 | 0.97 |
| Nile perch (juveniles) | 0.43 | 1.03 | 1.52 | 2.29 | 0.20 | 0.29 |
| Catfishes | 0.32 | 0.28 | 0.43 | 0.59 | 0.40 | 0.31 |
| Lungfish | 0.03 | 0.05 | 0.25 | 0.19 | 0.10 | 0.19 |
| Snoutfishes/Squeekers | 0.16 | 0.19 | 0.70 | 0.68 | 0.18 | 0.21 |
| Haplochr. Predatory | 0.05 | 0.01 | 1.28 | 1.28 | 0.04 | 0.01 |
| Haplochr. Zooplanktiv. | 0.23 | 0.49 | 2.15 | 2.11 | 0.09 | 0.18 |
| Haplochr. Phyto/Detritiv. | 0.25 | 0.03 | 2.12 | 2.30 | 0.1 | 0.01 |
| Rastrinebola | 0.44 | 1.18 | 2.53 | 2.23 | 0.14 | 0.33 |
| Nile tilapia | 0.22 | 0.17 | 0.37 | 2.18 | 0.34 | 0.07 |
| Other Tilapias | 0.17 | 0.19 | 0.78 | 0.72 | 0.17 | 0.20 |

We estimate an exploitation rate $E = 0.97$ for adult Nile perch in 1989, which is considerably higher than its value in 1979 and far exceeds both the optimal exploitation rates $E = 0.66$ and $E = 0.84$ for beach seines and gillnet respectively estimated by Rabuor & Manyala (1994) for the Kenyan waters of Lake Victoria. Similarly, *Rastrineobola* fishery shows a consistent increase in exploitation rate from $E = 0.14$ in 1979 to $E = 0.33$ in 1989, though this is still below the optimal exploitation rate of 0.48 estimated by Njiru *et al.* (1995) for the Kenya inshore waters of Lake Victoria. However, in the mid 1990s, a study by Manyala (1996) of length frequency data collected from five stations in the Winam Gulf, indicates that *Rastrineobola* is overexploited in this area at least, with a mean exploitation rate $E = 0.65$.

Looking at the scenario for each fishing gear separately, a doubling in fishing effort ($F = 2$) by mosquito seines only, keeping all the other gears at the initial level of fishing effort, produces the highest aggregate yield at 30.9 t $km^{-2}$ $yr^{-1}$ for Nile perch fishery (Figure 5.10). This catch, however, would be composed exclusively by juveniles and would not be sustainable if there is a further increase in fishing effort (as shown by a steep downwards trend beyond the MSY). On the other hand, longlines would sustain a stable Nile perch yield even at high level of fishing effort, since it mosly targets adult *Lates*.

**Figure 5.10:** Predicted and observed fish landings for Nile perch (a), *Rastrineobola* (b), and aggregated fish stocks (c) in the Kenyan waters of Lake Victoria.



The *Rastrineobola* fishery, by constrast, is not a multigear fishery. In fact the predicted scenario for mosquito seines approximates that for all gears together, implying that mosquito seines is the only fishing technology applied to *Rastrineobola* fishery. The small difference between the two trends is due to the slightly different food web interactions induced by variations in fishing effort of mosquito seines and aggregate gears respectively.

Figure 5.10a also shows that the observed yield-effort trend for Nile perch during the 1990s approximates, on average, the yield-effort curve for the aggregate gears predicted by our equilibrium analysis. This confirms the validity of our findings and shows that fishing effort has gone beyond the MSY level. Noting that the observed yield-effort for *Rastrineobola* refers to mosquito seines fishery statistics. The predicted yield-effort curves for both *Rastrineobola* and the Lake Victoria fishery as a whole seem overestimated with respect to the observed trends. It must be mentioned, however, that while the equilibrium analysis just described provides important insights into the dynamics of particular stocks, it does not account for variations in primary productivity and therefore will not show the effects of eutrophication. The next chapter describes an aggregate biomass dynamic model in order to capture the combined effect of eutrophication and fishing effort on the aggregate fish stock and fishery of Lake Victoria.

## 5.6. Discussion

This chapter has evaluated the role played by water quality on the freshwater ecosystem and fisheries of Lake Victoria. The analysis is limited to the Kenyan part of the lake, which is also the area most affected by nutrient loading. Dynamic simulations through an ecosystem model using the Ecopath approach shows how eutrophication may explain the sudden upsurge of the introduced Nile perch during the 1980s. In that case, nutrient loading appears as a necessary but not sufficient condition for the change in Nile perch abundance. An important trigger of change was an abrupt shift in primary productivity due to an external shock, probably related to an El Nino-ENSO event.

The Nile perch population explosion during the 1980s caused a profound transformation of the fishing industry leading to a dramatic increase in fishing effort. Even though these fisheries were initially highly productive, a sharp decline in both yields and catch per unit effort from the second half of the 1990s suggests that maximum sustainable yields could have been exceeded and the fishery is moving to a state of stock depletion. An equilibrium analysis (§ 5.4) from the base values of a Ecopath model for year 1989 depicts an overfishing situation in which fishing effort goes beyond its MSY level.

# Chapter 6

# BIOMASS DYNAMIC MODEL OF LAKE VICTORIA FISHERY

## 6.1. Introduction

In Chapter 5 we use an ecosystem model to analyze the multispecies fishery adopting the equilibrium analysis approach. We study the change in the Kenyan fishery of Lake Victoria, assuming equilibrium in 1989. Given the equilibrium assumption, this analysis neglects changes in phytoplankton biomass since it predicts changes in yield due only to variation in relative fishing effort, while keeping primary productivity constant. To quantify the combined effect of eutrophication and fishing effort on the aggregate fish stock and fishery, we use a biomass dynamics model which embeds a measure of phytoplankton biomass.

The dynamic simulation applied to the ecosystem model using the Ecosim approach in Chapter 5 estimates the time series for this environmental variable, which is then used in the parameterization of the biomass dynamics model in this chapter. In this way, we account for the food web interactions and shifts in primary productivity, embedding the estimated chlorophyll-a concentration variable into the biomass dynamics model for the aggregate fish stock. Therefore, both the ecosystem and biomass dynamics models are complementary, since the dynamics of the species composition of the fishery provided by the former contribute to the estimation of the level of aggregate fishing effort and aggregate yields in the latter.

## 6.2. The Gordon-Schaefer model for the Lake Victoria fishery

We now consider a biomass dynamics model for the aggregate fish stocks of the Kenyan waters of Lake Victoria. Whilst any Lotka-Volterra model of the dynamics of fish stocks indirectly captures the constraints imposed by the environment via the carrying capacity parameter, to date there are relatively few bioeconomic studies of fisheries (§ 4.4) that explicitly model the effect of environmental variables (Ikeda & Yokoi, 1980; Fréon *et al.*, 1993; Kasulo & Perrings, 2002, 2006). Following Kasulo & Perrings (2002), we include a measure of nutrient enrichment in a Gordon-Schaefer fishery model (Gordon, 1954; Schaefer, 1954, 1957). This variable is provided by chlorophyll-a concentration, a proxy for phytoplankton density, that gives a measure of primary productivity and eutrophication. We allow for a delay in the impact of water quality change on the fish stock growth. This gives the following equation of motion for fish stocks (in discrete time):

$$X_{t+1} - X_t = rX_t W_{t-1}\left(1 - \frac{X_t}{KW_{t-1}} - eW_{t-1}\right) - qE_t X_t \qquad [6.1]$$

where $X_t$ is the aggregate fish stock biomass at time *t* (tonnes); $W_{t-1}$ is chlorophyll-a concentration at time *t-1* (mg $m^{-3}$ or µg $l^{-1}$); $E_t$ is fishing effort at time *t* ('000 boat days); *r* is the intrinsic growth rate of the stock; *q* is its catchability coefficient; and *e* is a coefficient which gives the amount by which a unit change in the environmental variable depresses the natural growth rate of fish biomass. The model implies that nutrient loading positively affects the growth of fish stocks up to a certain point, after which further increases in nutrient loading cause a decrease in the maximum sustainable yield (MSY), open access and profit maximising levels of effort, and stock size. Which stock measure is appropriate depends on the fishery management regime and the set of property rights.

The degree to which the Lake Victoria fisheries have been regulated or policed varies over time. Table 6.1 reports the steady state levels of fish stock (*X*), catch (*Y*) and fishing effort (*E*) for each management regime by comparison with those of the standard Gordon-Schaefer model. The three outcomes of MSY, profit maximisation and open access reported in Table 6.1 are derived in the usual manner. The MSY solutions are obtained through the sustained yield function estimated from the equilibrium condition of equation [6.1]:

$$Y_t = qE_t K\left(W_{t-1} - eW_{t-1}^2 - \frac{qE_t}{r}\right) \qquad [6.2]$$

**Table 6.1:** Maximum sustainable yield (MSY), open access (oa) and profit maximising (*) steady state solutions.

| **MODEL 1:** Standard Gordon-Schaefer model | **MODEL 2:** With environmental variable |
|---|---|
| $X_{msy} = \frac{K}{2}$ | $X_{msy}(W) = \frac{K}{2}\left[W(1-eW)\right]$ |
| $Y_{msy} = \frac{rK}{4}$ | $Y_{msy}(W) = \frac{rK}{4}\left[W^2(1-eW)^2\right]$ |
| $E_{msy} = \frac{r}{2q}$ | $E_{msy}(W) = \frac{r}{2q}\left[W(1-eW)\right]$ |
| $X_{oa} = \frac{c}{pq}$ | $X_{oa}(W) = \frac{c}{pq}$ |
| $Y_{oa} = \frac{cr(pqK-c)}{p^2q^2K}$ | $Y_{oa}(W) = \frac{cr(pqKW-pqKeW^2-c)}{p^2q^2K}$ |
| $E_{oa} = \frac{r(pqK-c)}{pq^2K}$ | $E_{oa}(W) = \frac{r(pqKW-pqKeW^2-c)}{pq^2K}$ |
| $X^* = \frac{K}{4}\left[\left(\frac{c}{pqK}+1-\frac{\delta}{r}\right)+\sqrt{\left(\frac{c}{pqK}+1-\frac{\delta}{r}\right)^2+\frac{8c\delta}{pqKr}}\right]$ | $X^*(W) = \frac{K}{4}\left[\left(\frac{c}{pqK}+W-eW^2-\frac{\delta}{r}\right)+\sqrt{\left(\frac{c}{pqK}+W-eW^2-\frac{\delta}{r}\right)^2+\frac{8c\delta}{pqKr}}\right]$ |
| $Y^* = rX^*\left(1-\frac{X^*}{K}\right)$ | $Y^*(W) = rX^*(W)W\left(1-\frac{X^*(W)}{KW}-eW\right)$ |
| $E^* = \frac{Y^*}{qX^*}$ | $E^*(W) = \frac{Y^*(W)}{qX^*(W)}$ |

***Note:*** $p$ = market price of fish stock; $c$ = cost of fishing effort; $\delta$ = discount rate.

The yield function is an equilibrium concept expressing sustainable harvest as a function of effort. The maximum sustainable level of effort ($E_{MSY}$) is obtained by differentiating equation [6.2] with respect to $E$ and then solving for $E$ (Table 6.1). The maximum sustainable yield level of fish stock ($X_{MSY}$) in Table 6.1 is obtained by substituting $E_{MSY}$ into the growth function [6.1], differentiating with respect to $X$ and then solving for $X$.

Gordon's theory of the open-access fishery predicts an equilibrium in which economic rent is dissipated as effort expands to a level at which revenues exactly equal opportunity costs. In terms of the logistic yield-effort model, this equilibrium is determined by solving the following conditions:

$$\begin{cases} X_{t+1} - X_t = rX_t W_{t-1}\left(1 - \dfrac{X_t}{KW_{t-1}} - eW_{t-1}\right) - qE_t X_t = 0 \\ \pi = pqE_t X_t - cE_t = 0 \end{cases} \qquad [6.3]$$

where $p$ is the market price of the harvested stock; c is the cost of fishing effort; and $\pi$ represents the fishery profit. The system can be readily solved for the equilibrium-effort level $E = E_\infty$ and for the corresponding stock level $X = X_\infty$, leading to the open access solutions shown in Table 6.1. On the other hand, the optimal level of fish stock ($X^*$) and fishing effort ($E^*$) is given by maximising the fishery profit subject to the fish growth function [6.1].

Our modified version of the Gordon-Schaefer model explains how the environment can affect the yield through its influence on fish biomass. Different MSY, open access and optimal solutions can therefore be obtained, each for a particular state of the environmental variable. This should explain situations where wide fluctuations in the fish catch occur without proportional changes in effort.

Equation [6.1] is estimated using a method that is analogous to that developed by Schnute (1977) to estimate continuous time fishery models. Specifically, we transform the Gordon-Schaefer model into a linear form and then fit by linear regression. We use annual data on fishing effort and catch per unit effort observed from 1989 to 1998 for the Kenyan fisheries of Lake Victoria (Othina & Tweddle, 1999), and a time series of water quality data (phytoplankton biomass) obtained from the dynamic simulation of the Ecopath model using Ecosim (Table 6.2). The values, expressing annual average phytoplankton biomass for 1989-1998, are converted into chlorophyll-a concentrations assuming an average mixing depth of 10 m (Ochumba, 1995) and a phytoplankton/chlorophyll ratio of 70 (Scheffer, 1998), although the chlorophyll-a content of phytoplankton may vary within the range 0.5-2% of the dry weight depending on nutrient status, light and temperature (Ahlgren *et al.*, 1988).

**Table 6.2:** Observed fishing effort and estimated Chl-a concentration in Kenyan waters of Lake Victoria.

| Year | CPUE (kg boat$^{-1}$ day$^{-1}$) | Fishing effort ('000 boat days) | Phytoplankton concentration (t km$^{-2}$) | Chla ($W_t$) concentration (mg m$^{-3}$) | ($W_{t-1} + W_t$)/2 |
|---|---|---|---|---|---|
| 1989 | 180 | 1202 | 11.579 | 16.54 | 16.54* |
| 1990 | 152 | 1387 | 12.342 | 17.63 | 17.09 |
| 1991 | 145 | 1496 | 8.782 | 12.55 | 15.09 |
| 1992 | 137 | 1606 | 10.673 | 15.25 | 13.90 |
| 1993 | 115 | 1862 | 9.190 | 13.13 | 14.19 |
| 1994 | 93 | 1862 | 11.217 | 16.02 | 14.58 |
| 1995 | 86 | 2007 | 13.478 | 19.25 | 17.64 |
| 1996 | 78 | 1971 | 8.401 | 12.00 | 15.63 |
| 1997 | 93 | 2008 | 17.050 | 24.36 | 18.18 |
| 1998 | 86 | 2190 | 16.105 | 23.01 | 18.22 |

*assuming $W_{1988} = W_{1989}$

Given the basic Schaefer (1954) assumption that catch per unit effort (*U*) is proportional to stock abundance, or *U* = *qX*, the model is transformed as follows:

$$\frac{U_{t+1} - U_t}{q} = r\frac{U_t}{q}W_{t-1}\left(1 - \frac{U_t}{qKW_{t-1}} - eW_{t-1}\right) - E_tU_t \qquad [6.4]$$

implying that,

$$U_{t+1} - U_t = rU_tW_{t-1}\left(1 - \frac{U_t}{qKW_{t-1}} - eW_{t-1}\right) - qE_tU_t \qquad [6.5]$$

and hence that,

$$\frac{U_{t+1} - U_t}{U_t} = rW_{t-1}\left(1 - \frac{U}{qKW_{t-1}} - eW_{t-1}\right) - qE_t \qquad [6.6]$$

The equation of motion to be estimated where water quality is included is:

$$\frac{U_{t+1} - U_t}{U_t} = rW_{t-1} - \frac{r}{qK}U_t - reW_{t-1}{}^2 - qE_t \qquad [6.7]$$

in which the parameters to be estimated are: *r, r/qK*, *q* and *re*. The equation of motion to be estimated where no account is taken of water quality is:

$$\frac{U_{t+1}-U_t}{U_t}=r-\frac{r}{qK}U_t-qE_t \qquad [6.8]$$

in which the parameters to be estimated are: *r, r/qK* and *q*. The continuous time equation, analogous to expression [6.7], is obtained through time integration from *t* to *t*+1. Since integrating over some time period involves time averaging over that period we use a time averaged expression, $(W_{t-1}+W_t)/2$, for water quality at year *t*-1, while we assume that the values of *E* and *U* are the same at the beginning and end of each year:

$$\ln\left(\frac{U_{t+1}}{U_t}\right)=r\left(\frac{W_{t-1}+W_t}{2}\right)-\frac{r}{qK}U_t-re\left(\frac{W_{t-1}+W_t}{2}\right)^2-qE_t \qquad [6.9]$$

Following the parameterization according to expression [6.9], all estimated coefficients for both models (Table 6.3) were significant at the 5% level and of the expected sign, and the goodness of fit was high ($R^2 = 0.67$ and 0.76 for models 1 and 2, respectively) relative to results reported on other fisheries (cf. Hilborn & Walters, 1992). The model with the environmental variable fitted the data much better than the standard model, consistent with the findings of Kasulo & Perrings (2002).

**Table 6.3:** Parameter values for models 1 (standard Gordon–Schaefer) and 2 (with environmental variable) with t-ratio in parenthesis (** p<0.02; * p<0.05).

| **Model 1** | | **Model 2** | |
|---|---|---|---|
| *r* | 2.01 (3.0)* | *r* | 0.20 (3.1)* |
| *r/qK* | –0.0070 (3.3)** | *r/qK* | –0.0060 (3.7)** |
| *q* | –0.00075 (3.0)* | *q* | –0.00063 (3.3)* |
| | | *re* | –0.0059 (2.8)* |
| $R^2$ | 0.67 | $R^2$ | 0.76 |
| Adj $R^2$ | 0.56 | Adj $R^2$ | 0.62 |
| p-value | 0.04 | p-value | 0.05 |

The fitted equations of motion are:

$$X_{t+1} - X_t = 0.201033\,X_t W_{t-1}\left(1 - \frac{X_t}{53082\,W_{t-1}} - 0.029214\,W_{t-1}\right) - 0.000629\,E_t X_t \qquad [6.10]$$

$$X_{t+1} - X_t = 2.012948\,X_t\left(1 - \frac{X_t}{387523}\right) - 0.000746\,E_t X_t \qquad [6.11]$$

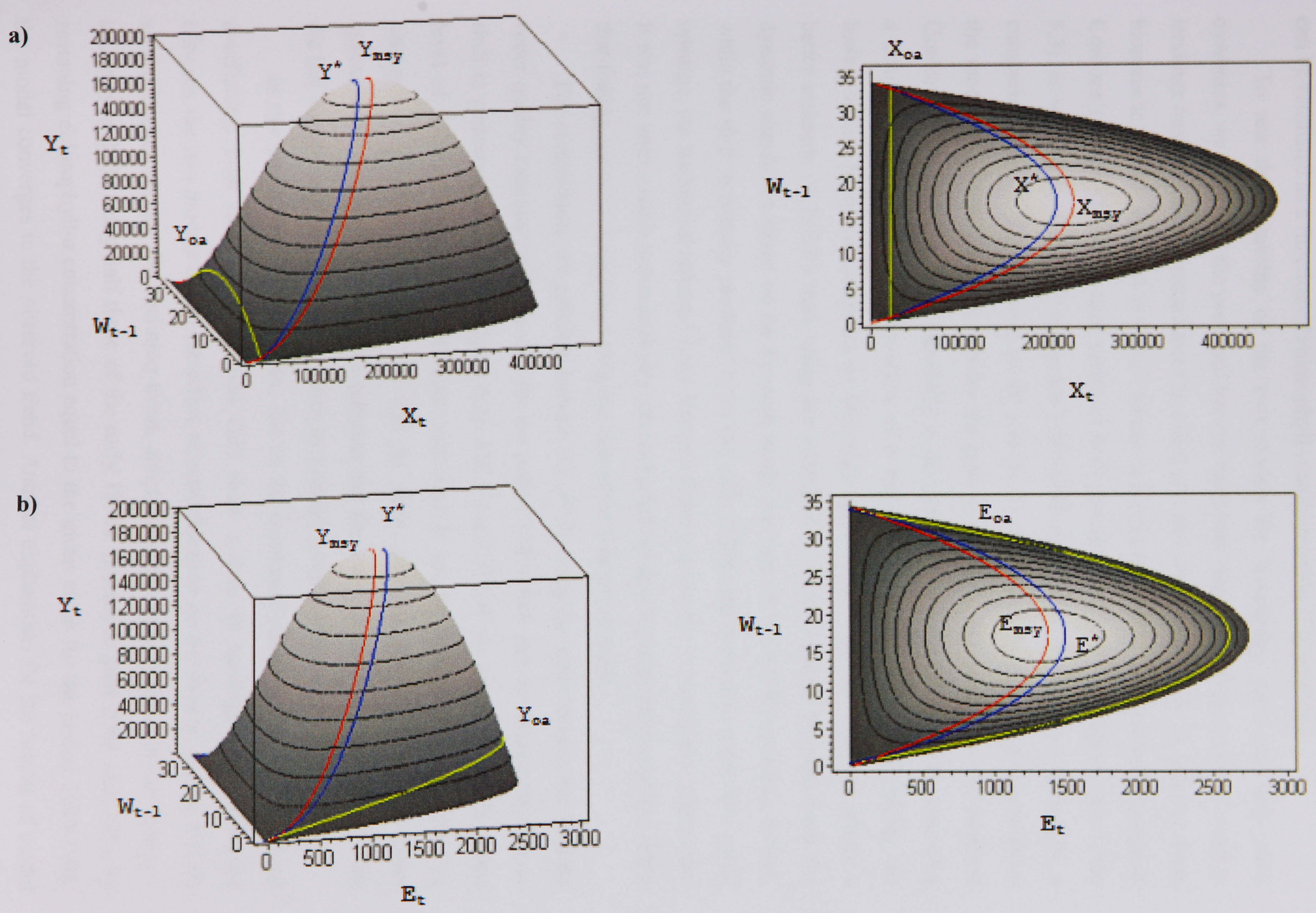
The first term of the right hand side of equation [6.10] says that the growth of the fish stock is positively affected by chlorophyll-a concentration at relatively low levels. Such growth reaches its maximum at the $W_{t-1}$ value of 17.11 mg $m^{-3}$ beyond which it starts to decline. At this point the maximum sustainable yield attainable is 195,368 tonnes and the fish stock 227,126 tonnes (Figure 6.1a). A further increase in nutrient load has a negative impact on growth and on harvests. The environmental variable influences both the growth rate at a given level of the stock ($rXW$), the maximum sustainable yield and the carrying capacity of the lake ($KW[1\text{-}eW]$). The values for MSY, given the increasing level of chlorophyll-*a* (Figure 6.1), are supported by previous MSY estimates, which increased from an average of 20–50,000 tonnes during the 1960s, 1970s, and 1980s (Geheb, 1997), to 150,000 tonnes in the late 1980s (Republic of Kenya, 1988), then declined to 107–135,000 tonnes (Brown *et al.*, 2005).

To identify the steady state conditions under both open access ($Y_{oa}$) and profit-maximizing regimes ($Y^*$), we assume constant values for ex-vessel prices ($p$ = 6198 Ksh $t^{-1}$ (US\$ 376) at 1987 values), cost of fishing effort ($c$ = 84188 Ksh (US\$ 5102) per thousand boat-days at 1987 values). These are average values during the period 1980–2000. The discount rate is taken to be $\delta = 0.255$. Applying the equations for optimal harvest under each property right regime, the results are graphed in Figure 6.1. Like MSY, they turn out to be sensitive to water quality.

Similarly, the related yield-effort function from equation [6.10] depends not only on fishing effort but also on water quality:

$$Y_t = qE_t K\left(W_{t-1} - eW_{t-1}^2 - \frac{qE_t}{r}\right) = 33.3886\,E_t(W_{t-1} - 0.0292\,W_{t-1}^2 - 0.00313\,E_t) \qquad [6.12]$$

**Figure 6.1:** Logistic growth and yield-effort curves for the aggregate fish stock as function of water quality measures.



Assuming the same values for $p$, $c$, and $\delta$, a path for fishing effort at MSY ($E_{msy}$), as well as for steady-state conditions under open access ($E_{oa}$) and profit-maximizing ($E^*$) regimes, can be estimated as a function of chlorophyll-*a* concentration (Figure 6.1b).

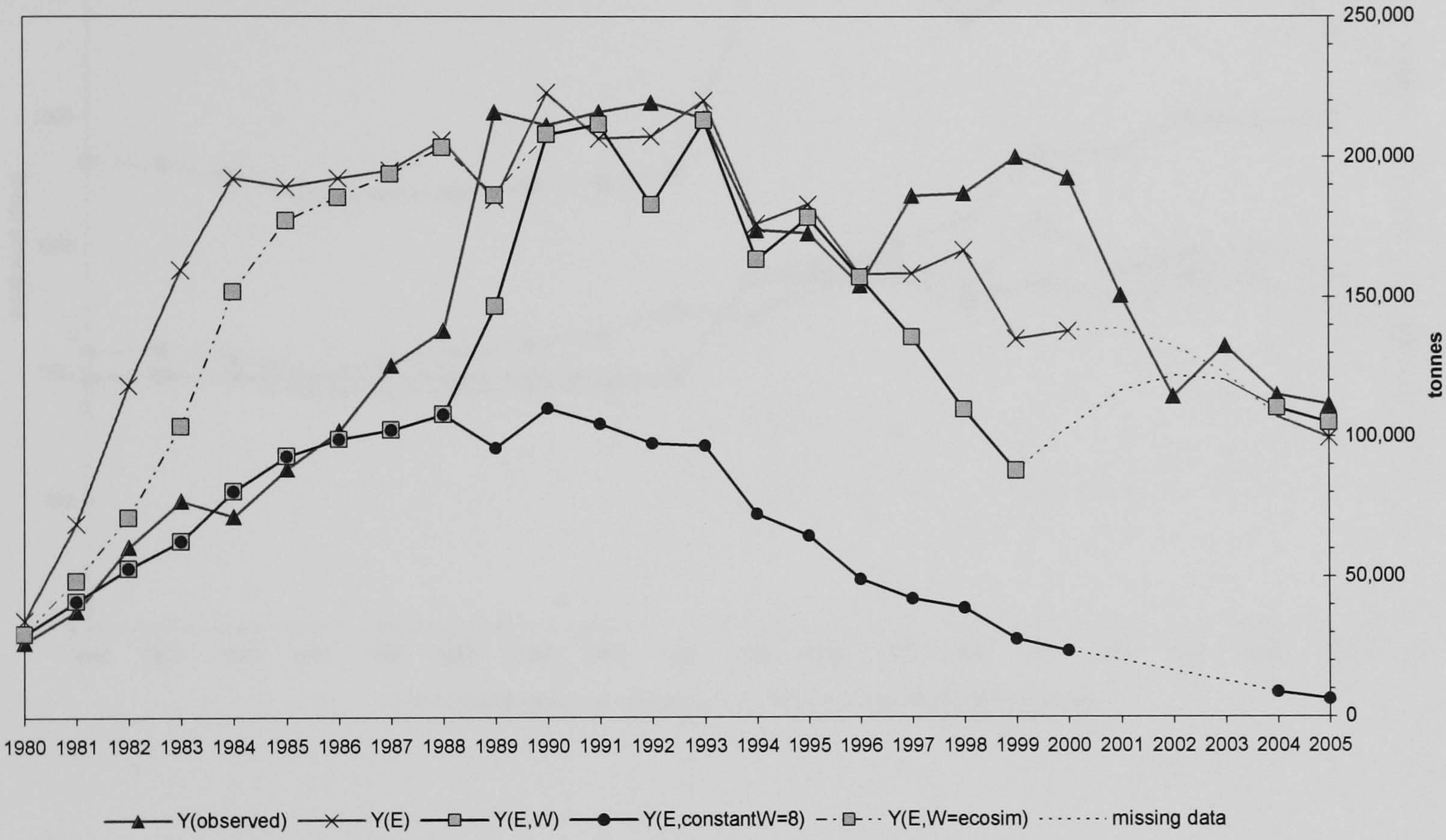
To test the suitability of the two models for forecasting yield and fish stock dynamics, we compared the predicted harvest from both models with the observed trend in landings based on the estimated trend in effort ($E_t$) and water quality ($W_{t-1}$). Starting stock biomass in 1980 is assumed at 43,627 tonnes as the baseline estimated by Kudhongania & Cordone (1974). Given the time series of both observed and estimated effort (see Table 5.3), as well as water quality values for 1989-1998 estimated with Ecosym (Table 6.2), a comparison (Figure 6.2) between [6.10] and [6.11] shows that the predicted harvest from the model $Y(E,W)$ better approximates the path of the observed catch than the standard Gordon-Schaefer model, $Y(E)$, especially in the period 1980-1988. Note that by assuming a constant chlorophyll-a concentration of 8 mg $m^{-3}$ for period 1980-1988, taking the lowest values from measurements by Ochumba & Kibaara (1989) in 1986, we obtain a better estimate of $Y(E,W)$ than using our estimates of phytoplankton biomass from the dynamic simulation based on the Ecopath model for year 1979 ($Y(E, W{=}ecosim)$). In fact, while the shift in primary productivity in the early 1980s may have driven the Nile perch upsurge, the traditional artisanal Lake Victoria fishery could not exploit *Lates* population. It was not until a more technologically advanced fishery developed at the end of the 1980s that fishing pressure started impacting the size of the Nile perch stock.

For comparison, we estimate harvest $Y(E,W{=}8)$ using the same model, but holding water quality constant (at 8 mg $m^{-3}$) for the period 1989-1998 and therefore assuming no shift in primary productivity (Figure 6.2). $Y(E,W)$ and $Y(E,W{=}8)$ are subject to the same level of fishing effort, so the different values of $W$ are an indication of the effect of changes in primary production. It can be seen that the predicted $Y(E,W)$ better approximates observed harvest. This suggests that the sharp increase in fish landings over the last decade may be explained as much by primary productivity as by fishing effort.

In the second half of the 1990s, due to the combined effect of eutrophication and overfishing, both models $Y(E,W)$ and $Y(E)$ show a drop in landings. The difference between the two models lies in the effect of eutrophication on the fishery. Model $Y(E,W)$ seems to overestimate the declining trend, anticipating the near collapse of the fishery - landings in 2005 being half those of the early 1990s. Indeed, in year 2004 and 2005, by assuming chlorophyll-a concentration equal to the mean value for the period 1989-1998, the model converges to the observed trend. Another explanation for the results of model

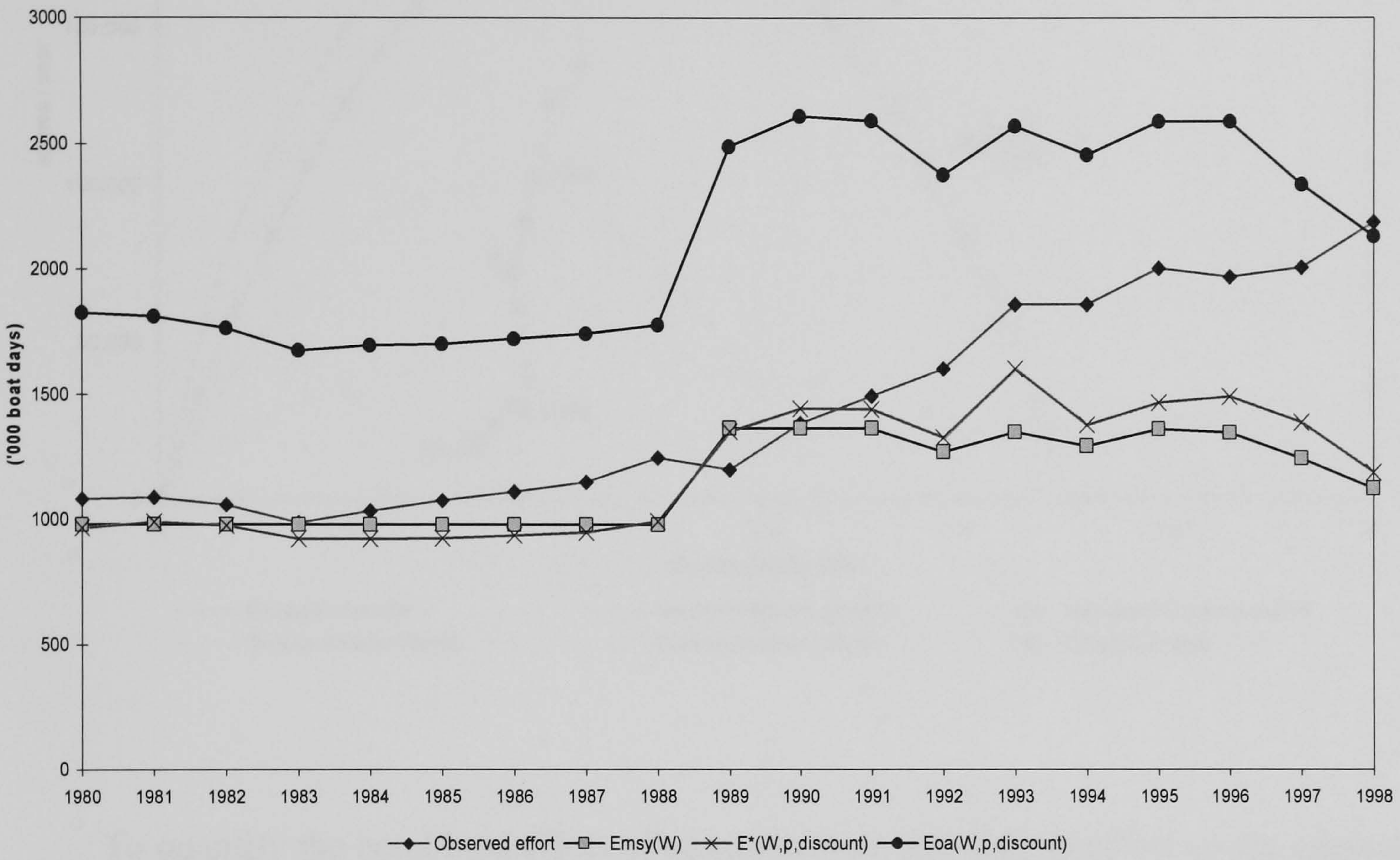
*Y*(*E*) may be due to the behaviour of fishers, since the fishery started targetting species less affected by eutrophication such as the Nile tilapia. In this case model *Y*(*E*,*W*) tends to underestimate predicted yields, since parameterisation was obtained with a data set that refers to a period in which Nile perch landings were accounting for about 60% of the fishery.

**Figure 6.2:** Predicted yield as function of fishing effort from MODEL 1 and as function of effort and water quality for MODEL 2.



One other implication of the model results is worth noting. After 1990 the observed level of effort departs from the predicted level of effort (Figure 6.3). Comparing observed effort with the paths of the steady state solutions, $E_{oa}$ and $E^*$, we see the industry moving from an almost-profit-maximising level of effort in the 1980s towards an open access rent-dissipating level of effort in the 1990s. The increased productivity of the fishery attracted new and largely unregulated investment in the sector during the early 1990s. In those years, the success of the Nile perch fishery was recognised not only in East Africa but also internationally where a market for the species was quickly established (Crean *et al.*, 2002). This prompted an unprecedented inflow of national and international capital, transforming the once locally-based fishery. Within a few years fish processing capacity in Kenya grew from nil in the early 1980s to 15 registered factories with a processing capacity far in excess of the sustainable level (Bokea & Ikiara, 2000).

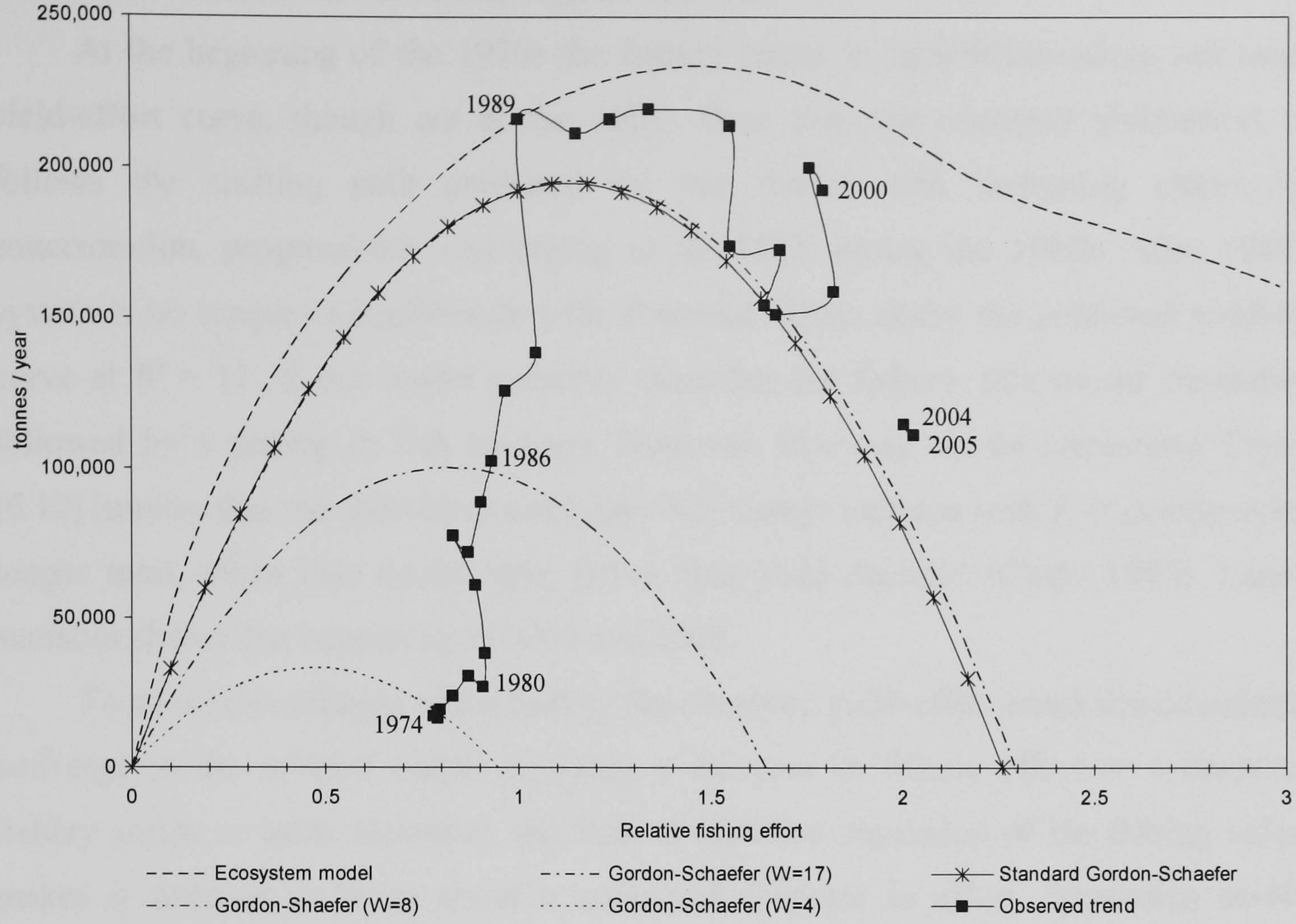
**Figure 6.3:** Trend in the steady-state equilibrium conditions at MSY ($E_{msy}$), under open access ($E_{oa}$) and bioeconomic optimum ($E^*$) for fishing effort ('000 boat days) with varying stock price (*p*), cost of effort (*c*), discount rate, and chlorophyll-a concentration (*W*).



## 6.3. Discussion

In the previous chapter (§ 5.4) we perform a fishery equilibrium analysis for the multispecies fisheries of Lake Victoria using the ecosystem approach, Ecopath. We study the change in the Kenyan fishery of Lake Victoria assuming equilibrium in 1989. During the 1980s fishing effort did not increase subsantially (Figure 6.4). The model predicts a MSY of 233,656 tonnes at 1.728 million boat days (45% increase from the fishing effort equilibrium level in 1989). The estimated yield-effort curve in this case is significantly above the observed yield-effort trend. We hypothesise that this is due to the fact that ecopath neglects changes in phytoplankton biomass. In fact, the equilibrium analysis predicts changes in yield due only to variation in relative fishing effort, while keeping primary productivity constant.

**Figure 6.4:** Comparison of the observed yield-effort trend against the estimated yield-effort curves from the ecosystem model and Gordon-Schaefer models.



To quantify the combined effect of eutrophication and fishing effort on the aggregate fish stock and fishery, we specify and estimate a biomass dynamics model, which embeds a measure of phytoplankton biomass as an environmental variable. The MSY is attained at 195,368 tonnes, with a fishing effort of 1.368 million boat days and chlorophyll-a concentration $W_{t-1}$ value of 17.11 mg m$^{-3}$. Figure 6.4 compares the predictions from the ecosystem and Gordon-Schaefer models. We consider three different Gordon-Schaefer models that include the environmental variable ($W$), chlorophyll-a concentration. The first yield-effort curve refers to the period after 1989, assuming costant 16.97 mg m$^{-3}$ chlorophyll-a concentration, according to the long term 1989-1998 average. The second curve may describe the situation over the 1980s assuming constant value of 8 mg m$^{-3}$. The third yield-effort curve is meant to describe the baseline situation (in both fishing effort and water quality) considering a chlorophyll-a concentration of 4 mg m$^{-3}$, which is in line with the baseline value of 1.2-5.5 mg m$^{-3}$ measured by Talling (1965) in 1960-1961. For comparison we consider also the standard Gordon-Schaefer model. It can be seen that the Gordon-Schaefer model with the environmental variable approximates the standard model when chlorophyll-a concentration approaches the MSY value at 17.11 mg $m^{-3}$. Below or above this threshold the curve will shift downwards.

At the beginning of the 1970s the fishery seems in equilibrium along our baseline yield-effort curve, though not at the MSY. Over time the observed yield-effort trend follows the shifting path predicted by our model with increasing chlorophyll-a concentration, progressively converging to the MSY during the 1980s. After 1988 the system is no longer in equilibrium with observed values above the predicted yield-effort curve at $W = 17$. If our model correctly describes the fishery, this would inevitably be followed by a decline in fish landings. However, this may not be immediate. Equation [6.12] implies that the short-term catch rate will always increase with $E$. It is only over the longer term, when fish stocks have fallen, that yield declines (Clark, 1990). Landings statistics shows this happening in 2004 and 2005.

To avoid the collapse of the fishery the observed yield-effort trend should eventually converge to the optimal curve, implying a decrease in fishing effort or a decrease in fishery yields or both. However, the lack of effective regulation of the fishing industry, makes it difficult to bring about a measured decrease in effort. Increasing levels of eutrophication may further complicate the situation, since average chlorophyll-a concentrations above 17.11 mg $m^{-3}$ may be expected to produce a downward shift of the yield-effort curve, requiring a greater cut in fishing effort to avoid the collapse of the fishery.

# Chapter 7

# THE WATERSHED APPROACH TO FISHERY AND WETLAND MANAGEMENT

## 7.1. Introduction

Freshwater ecosystems and fisheries must be studied within the context of their watershed and its related human activities (§ 1.2). During the last decades, increased nutrient loading, due to land cover and land use changes, is intensifying the eutrophication process in Lake Victoria. Eutrophication is thought to be an important factor in explaining the recent decline of fish stocks in the lake (§ 1.3.1.2). We show how changes in phytoplankton biomass impact the freshwater ecosystem through a cascading effect (§ 5.3). This finally affects fish productivity and fisheries. We estimate the impact of water quality on fishery through parameterization of a modified Gordon-Schaefer model (§ 6.2). We found that in an eutrophic state, which for the Kenyan waters of Lake Victoria was above an estimated chlorophyll-a concentration threshold of 17.11 mg $m^{-3}$, a further increase in phytoplankton biomass would cause a decrease in aggregate fish stock biomass and landings respectively.

The primary focus of this chapter is the relation between the fishery and changes in the wetlands at the margins of Lake Victoria. We model the relation between land use change and water quality, and hence between wetlands and fish stock biomass. In this context we are concerned with the nutrient retention function of wetlands and the relationship between land use and nutrient loading. In the case of Lake Victoria the latter relation has long been argued in the literature (Hecky & Bugenyi, 1992). However, lack of long-term monitoring data and the complexity of ecosystem changes made it difficult to estimate a land use/water quality function (Verschuren *et al.*, 2002). Nevertheless, our working hypothesis in this chapter is that the observed change in fish productivity is at least partly the result of excess nutrient loading from the drainage basin.

## 7.2. The land allocation problem

Given our working hypothesis, the economic problem to be investigated stems from two facts. One is that downstream economic activities, such as fishing, depend on the quality of water (i.e. eutrophication) through its impact on ecosystem health and fish biomass productivity. The other is that water quality in the lake is influenced by both the extent and intensity of agricultural activities in the upstream basin and the buffering function between the agricultural and fishery sectors exerted by wetlands. Since the effects of upstream activities are not reflected in market prices they are ignored by those responsible. That is, any costs imposed or benefits conferred on other economic sectors or members of society are not taken into account. If we can identify the nature and significance of the interdependence between activities that either depend on or affect the characteristics of the wetlands, we can also identify ways in which the provincial or national government can intervene to assure the sustainable use of wetland resources.

To address this problem we model the impact of nutrient loading on fisheries assuming the watershed drains into wetlands before entering the lake. The stylized facts are as follows. The watershed extent, which includes the sum of drainage land and wetlands, is fixed ($L_{TOT}$). This fixed amount of land is allocated between two types of land use only: agricultural land ($L_A$) and wetland ($L_W$). Wetland and agricultural land are spatially homogeneous. Each household is assumed to control a spatial unit of 1 ha. Within the agricultural land, fallow land ($L_F$) is spatially patchy and it is assumed that households produce under rotation system where, within each spatial unit of agricultural land, a fixed portion is covered by fallow-pasture for managing soil productivity so that at each point in time this spatial unit fraction is switched into tillage ($L_C$) and out of fallow-pasture ($L_F$), keeping constant the chosen level of agricultural intensity ($\beta$). These assumptions are summarized by the following equation:

$$L_{TOT} = L_A + L_W = [\alpha(1-\beta) + \alpha\beta + (1-\alpha)] * L_{TOT} \qquad [7.1]$$

with,

$$L_A = L_C + L_F$$
$$0 \le \alpha \le 1$$
$$0 \le \beta \le 1$$

where $L_{TOT}$, $L_A$, $L_C$, $L_F$ and $L_W$ are the extent of total land in the watershed, agricultural land, land under crop production, land under fallow and wetlands respectively expressed in ha; $\alpha$ is the percent of total land ($L_{TOT}$) which is agricultural land ($L_A$), such that $L_A = \alpha L_{TOT}$ and $L_W = (1\text{-}\alpha)L_{TOT}$; and $\beta$ is is the percent of agricultural land ($L_A$) which is under fallow ($L_F$), such that $L_F = \beta L_A$ and $L_C = (1\text{-}\beta)L_A$. In other words the choice between agricultural land and wetland extent is expressed by $\alpha$, while the choice of agricultural intensity by $\beta$.

First, we consider our fish stock dynamics model [6.1] in terms of a damage function directly related to nutrient loading from the watershed, $D(z_t)$. Observed changes in primary productivity and phytoplankton composition in Lake Victoria are assumed to be the result of excess nutrient loading from the drainage basin. The equation for fish stock dynamics in presence of harvesting is:

$$\Delta X = b(X_t, D(z_t)) - h(X_t, E_t) \qquad [7.2]$$

or,

$$\Delta X = rX_t\left(D(z_t) - \frac{X_t}{K}\right) - qE_tX_t \qquad [7.3]$$

where $X$ represents the fish stock biomass; $D(z) \leq 1$ is the damage function in terms of nutrient loading $z$; $b(\bullet)$ and $h(\bullet)$ are the growth and harvesting function respectively; and $r$, $K$, $q$, and $E$ are the usual Gordon-Schaefer parameters. The damage function is represented by the negative quadratic function of chlorophyll-a concentration:

$$D(z_t) = W(z_t) - eW^2(z_t) \qquad [7.4]$$

where $W$ is chlorophyll-a concentration (mg $m^{-3}$) and $e$ is the estimated coefficient in equation [6.10].

The capacity of wetlands to purify watersheds reducing nitrogen, phosphorus and other pollutant levels has been analysed in a number of studies (Nichols, 1983; Mitsch & Jorgensen, 1989; Reckhow & Quian, 1994; Richardson *et al.*, 1997; Wang & Mitsch, 2000), and has also been evaluated at catchment scale (Mitsch & Wang, 2000; Arheimer

& Wittgren, 2002; Trepel & Palmeri, 2002). At the catchment scale, predictive models have been developed to characterise the ability of wetlands to reduce point and non-point pollution from the upstream watershed.

Several studies analysed nutrient retention by wetlands as a function of wetland area and nutrient loading (§ 3.2). Following Arheimer & Wittgren (2002) as in equation [3.28], we use the general mass balance model presented by Kadlec & Knight (1996) without considering the background concentration values. Assuming that hydraulic flow through and out of the wetland is equivalent to the water inflow and framing the equation in terms of $0 \leq \alpha \leq 1$, we obtain the following expression which follows the generalized equation [3.34]:

$$z(g_t,\alpha_t) = g_t(\bullet) * w(\alpha_t) = g_t * \exp\left(\frac{-k_{TP}(1-\alpha_t)L_{TOT} * 10^4}{v}\right) \qquad [7.5]$$

where $z_t$ is the nutrient outflow from the wetland to the main lake (t $yr^{-1}$); $g_t$ is nutrient inflow to the wetland (t $yr^{-1}$) which is represented by the nutrient runoff function $g(\bullet)$; $w(\bullet)$ is the wetland function dependent on wetland area; $v$ is water inflow to the wetland ($m^3$); $k_{TP}$ is the areal removal rate constant (m $yr^{-1}$) for total phosphorus (or $k_{TN}$ if we consider nitrogen); $(1-\alpha_t)L_{TOT}*10^4$ represents wetland area ($m^2$); and $L_{TOT}$ is total watershed area including wetlands (ha).

We consider a nutrient runoff function which models the non-point nutrient pollution and related flow from the watershed into the wetland. Phosphorus, unlike nitrogen, does not have a leaching component. The general form of the the function is as follows:

$$g_t = g(Q_t, P_t, \Omega_t, \alpha_t, \beta_t) \qquad [7.6]$$

and the specific form is:

$$g_t = (\alpha_t L_{TOT} S_{DR} A(\beta_t, \Omega_t)\theta Q_t(\bullet)) * 10^{-3} \qquad [7.7]$$

where nutrient loading ($g_t$) into the wetland is expressed in t $yr^{-1}$; $Q_t$ is a measure of soil nutrient content, considering the 20 cm topsoil and espressed in kg $ha^{-1}$; $P_t$ is fertilizer application (in this case phosphorus) in kg $ha^{-1}$ $yr^{-1}$on tillage land ($L_C = (1-\beta)*\alpha L_{TOT}$); $\Omega_t$ is rainfall (mm $yr^{-1}$); $0 \leq S_{DR} \leq 1$ is the sediment delivery ratio; $\theta$ is the soil bulk density factor; $A_t$ is a measure of soil erosion (t $ha^{-1}$ $yr^{-1}$) and $\alpha_t$ and $\beta_t$ are the already defined indexes of agricultural extent and production intensity respectively.

The nutrient runoff function is subject to the soil dynamics. Indeed soil management is a dynamic process that has to be adjusted continually to changes in the stock, when decisions made in one time period have cumulative effects over time (§ 2.2). Soil dynamics are affected by the original level of stock, in this case nutrient content in the soil, land use and a set of parameters mostly related to precipitation patterns. Soil dynamics vary according to the nutrient considered. Nitrogen (N) dynamics for instance are particularly difficult to assess, since it would be necessary to account not only for nutrient runoff of soil's nitrogen through soil erosion, but also leaching from nitrogen fertilizer applications. In addition, the processes of denitrification through volatilization and nitrogen fixation by plants have to be considered. Phosporous (P) dynamics on the other hand are easier to model. In tropical soils phosphorus is tightly bound by soil particles and runoff is represented almost entirely by soil erosion (Roy & Misra, 2003).

In general terms soil dynamics, when accounting for phosphorus, are represented by the following general and specific functional forms:

$$\Delta Q = q(Q_t, P_t, \beta_t, \Omega_t) \qquad [7.8]$$

and

$$\Delta Q = P_t(1-\beta_t) + Dep(\Omega_t) + \chi\beta_t - \eta y_A(P_t, \beta_t, Q_t) - S_{DR}A_t(\beta_t)\theta Q_t \qquad [7.9]$$

where the variation $\Delta Q$ is expressed in kg P $ha^{-1}$; $y_A$ is crop productivity (t $ha^{-1}$ $yr^{-1}$); $\chi$ represents the nutrient build-up factor due to fallow rotation (kg P $ha^{-1}$ $yr^{-1}$); $\eta$ expresses the phosphorous content in the harvested crops (kg $t^{-1}$); $Dep_t$ is atmospheric deposition of phosphorous (kg $ha^{-1}$ $yr^{-1}$); and the other parameters are defined as above. Soil dynamics in equation [7.9] are identified by a simplified nutrient balance approach (§ 2.2.2) where the initial natural stock of phosporous concentration in the soil is increased by fertilizer application on cropland ($L_c$ = (1-$\beta$)), nutrient buid up due to fallow rotation, and atmospheric deposition. This stock however is diminished by loss of phosphorous through its content in the harvested crop and soil erosion. It must be noticed that we are also including the sediment delivery ratio ($S_{DR}$) to account for sediment deposition within the basin, which in other words diminishes the net soil loss due to erosion.

The productivity of the agriculture sector is influenced not only by fertilizer applications, but also by topsoil quality among other factors. Indeed, soil is an essential input in agricultural production. Agricultural economists recognize that soil management is a dynamic process which has to be adjusted continually to changes in the stock, i.e. the depth or the average nutrient concentration in the topsoil layer. In this context several economic models of onsite externalities have been developed to estimate the optimal agricultural utilization of the soil (§ 2.2.1). In our model both soil quality and fertilizer applications are embedded in the crop production function, which estimates the average per-hectare productivity of agricultural land ($L_A$) in the watershed:

$$y_{At} = (1-\beta_t) * f(P_t, N_t, Q_t) \qquad [7.10]$$

with the total crop production in the watershed given by:

$$Y_{At} = L_{TOT} * \alpha_t * y_{At} \qquad [7.11]$$

where $y_{At}$ is crop productivity (t ha$^{-1}$ yr$^{-1}$); $Q_t$ is a measure of soil quality (per-hectare nutrient content considering the 20 cm topsoil); $P_t$ is phosphorous fertilizer application (kg ha$^{-1}$) on tillage land ($L_C =$ (1-$\beta$)); and $N_t$ is nitrogen fertilizer application (kg ha$^{-1}$).

### 7.2.1. The sectoral and social decision problem

The decision problem can be modelled at several different levels. It is important to distinguish between private, sectoral and social decisions wherever there are potentially serious externalities. The agricultural sector and its private agents influence the downstream fishery through their decisions on land allocation ($\alpha$), cropping intensity ($\beta$) and phosphorous fertilizer application ($P$), subject to soil nutrient dynamics. The fishery sector on the other hand takes these upstream externalities as given in its choice of fishing effort. If fishers ignore the effect of nutrient loading on fish stock dynamics, fishery management could lead to an excess of fishing effort which may drive the fishery to collapse.

The fishery sector chooses the optimal amount of fishing effort (*E*) to maximise its profit, taking into account the fish stock externality caused by nutrient loading (*z*). The sectoral decision problem is given by the sum of the private decision problems among all the fishermen. Considering the $j^{th}$ representative agent, the maximisation problem for the fishery sector is:

$$Max_E \sum_{t=0}^{\infty}\sum_{j=1}^{n} \rho^t \pi_F^j\left(E_t^j, X_t\right) = \sum_{t=0}^{\infty}\sum_{j=1}^{n} \rho^t \left(pqE_t^j X_t - cE_t^j\right) \qquad [7.12]$$

subject to:

$$\Delta X_t = b\left(X_t, D(z_t)\right) - \sum_{j=1}^{n} h^j\left(X_t, E_t^j\right) = rX_t\left(D(z_t) - \frac{X_t}{K}\right) - \sum_{j=1}^{n} qE_t^j X_t \qquad [7.13]$$

$$X_0 = a$$

where $p$ is the price of fish; $c$ is the cost of fishing effort; $\rho = 1/(1+\delta)$ is the discount factor; $\delta$ is the discount rate; and the other parameters are as defined in the Gordon-Schaefer model. The current value Hamiltonian is:

$$\widetilde{H}\left(X_t, E_t^j, \lambda_{t+1}\right) = \sum_{j=1}^{n}\left(\left(pqE_t^j X_t - cE_t^j\right) + \rho\lambda_{t+1}\left(rX_t\left(D(z_t) - \frac{X_t}{K}\right) - qE_t^j X_t\right)\right) \qquad [7.14]$$

which leads to the first order conditions:

$$\frac{\partial \widetilde{H}(\bullet)}{\partial E_t} = \sum_{j=1}^{n}\left(\frac{\partial \pi_F^j(\bullet)}{\partial E_t^j} - \rho\lambda_{t+1}\frac{\partial h^j(\bullet)}{\partial E_t^j}\right) = 0 \qquad [7.15]$$

$$\rho\lambda_{t+1} - \lambda_t = -\frac{\partial \widetilde{H}(\bullet)}{\partial X_t} = \sum_{j=1}^{n}\left(-\frac{\partial \pi_F^j(\bullet)}{\partial X_t} - \rho\lambda_{t+1}\left(\frac{\partial b(\bullet)}{\partial X_t} - \frac{\partial h^j(\bullet)}{\partial X_t}\right)\right) = 0 \qquad [7.16]$$

$$X_{t+1} - X_t = \left(rX_t\left(D(z_t) - \frac{X_t}{K}\right) - \sum_{j=1}^{n} qE_t^j X_t\right) = 0 \qquad [7.17]$$

By solving the system with [7.17] evaluated at the steady state we obtain the profit maximising solution for fish stock $X^*$, yield $Y^*$, and fishing effort $E^*$ respectively as shown in Table 6.1.

Agricultural households, on the other hand, maximize farm profits when each $i^{th}$ household, which represents 1 ha unit, allocates the optimal amount of phosphorous (or nitrogen) fertilizer (*P*), and chooses the optimal fallow extent ($\beta$), subject to soil nutrient dynamics ($\Delta Q$) and an initial concentration of nutrient in the soil ($Q_0$ = a) Thus the decision problem for the agricultural sector considering the $i^{th}$ representative agent is:

$$Max_{P,\beta}\sum_{t=0}^{\infty}\sum_{i=1}^{\alpha L_{TOT}}\rho^t\pi_A^i\left(P_t^i,\beta_t^i,Q_t^i\right)=\sum_{t=0}^{\infty}\sum_{i=1}^{\alpha L_{TOT}}\rho^t\left(p_A(1-\beta_t^i)f\left(P_t^i,Q_t^i\right)-c_A P_t^i\right) \qquad [7.18]$$

subject to:

$$\Delta Q^i = P_t^i\left(1-\beta_t^i\right)+Dep_t^i(\Omega_t^i)+\chi\beta_t^i-\eta y_{At}^i(P_t^i,\beta_t^i,Q_t^i)-S_{DR}^i A_t^i(\beta_t^i,\Omega_t^i)\theta Q_t^i \qquad [7.19]$$

$$Q_0^i = a$$

where $p_A$ is the price of harvested crop; $c_A$ is the cost of mineral fertilizer; and the other parameters and variables have already been defined. The current value Hamiltonian for this problem is:

$$\widetilde{H}=\sum_{i=1}^{\alpha L_{TOT}}\left(p_A(1-\beta_t^i)f^i(\bullet)-c_A P_t^i+\rho\lambda_{t+1}\left(P_t^i\left(1-\beta_t^i\right)+Dep_t^i+\chi\beta_t^i-\eta(1-\beta_t^i)f^i(\bullet)-S_{DR}^i A_t^i(\bullet)\theta Q_t^i\right)\right) \qquad [7.20]$$

which leads to the following first order conditions:

$$\frac{\partial\widetilde{H}}{dP_t}=\sum_{i=1}^{\alpha L_{TOT}}\left(p_A\left(1-\beta_t^i\right)\frac{\partial f^i(\bullet)}{\partial P_t^i}-c_A+\rho\lambda_{t+1}\left(1-\beta_t^i-\eta\left(1-\beta_t^i\right)\frac{\partial f^i(\bullet)}{\partial P_t^i}\right)\right)=0 \qquad [7.21]$$

$$\frac{\partial\widetilde{H}}{d\beta_t}=\sum_{i=1}^{\alpha L_{TOT}}\left(-p_A f^i(\bullet)+\rho\lambda_{t+1}\left(-P_t^i+\chi+\eta f^i(\bullet)-S_{DR}^i\frac{\partial A_t^i(\bullet)}{\partial\beta_t^i}\theta Q_t^i\right)\right)=0 \qquad [7.22]$$

$$\rho\lambda_{t+1}-\lambda_t=-\frac{\partial\widetilde{H}}{dQ_t}=\sum_{i=1}^{\alpha L_{TOT}}\left(-p_A\left(1-\beta_t^i\right)\frac{\partial f^i(\bullet)}{\partial Q_t^i}-\rho\lambda_{t+1}\left(-\eta\left(1-\beta_t^i\right)\frac{\partial f^i(\bullet)}{\partial Q_t^i}-S_{DR}^i A_t^i(\bullet)\theta\right)\right)=0 \qquad [7.23]$$

$$Q_{t+1}^i-Q_t^i=P_t^i\left(1-\beta_t^i\right)+Dep_t^i+\chi\beta_t^i-\eta y_{At}^i-S_{DR}^i A_t^i\theta Q_t^i=0 \qquad [7.24]$$

The social decision problem considers both sectors together in a joint profit maximisation problem. In other words the net social value is given by maximising the sum of the agricultural and fishery sector profits with respect to level of fertilizer application $P$, wetland extent $\alpha$, agricultural intensity $\beta$, and fishing effort $E$:

$$Max_{P,\beta,\alpha,E} \sum_{t=0}^{\infty} \sum_{i=1}^{\alpha L_{TOT}} \sum_{j=1}^{n} \rho^t \left( \pi_A^i \left( P_t^i, \alpha_t, \beta_t^i, Q_t^i \right) + \pi_F^j \left( E_t^j, X_t \right) \right) \qquad [7.25]$$

subject to equation [7.13] and [7.19]. The current value Hamiltonian for the social problem is:

$$\widetilde{H} = \sum_{i=1}^{\alpha L_{TOT}} \sum_{j=1}^{n} \left( \pi_A^i(\bullet) + \pi_F^j(\bullet) + \rho\lambda_{t+1} q\left( P_t^i, \beta_t^i, Q_t^i \right) + \rho\mu_{t+1} \left[ b\left( E_t^j, X_t, D(\alpha_t, \beta_t^i) \right) - h\left( E_t^j, X_t \right) \right] \right) \qquad [7.26]$$

which gives the following first order conditions:

$$\frac{\partial \widetilde{H}}{\partial \alpha_t} = \sum_{i=1}^{\alpha L_{TOT}} \left( \frac{\partial \pi_A^i(\bullet)}{\partial \alpha_t} + \rho\mu_{t+1} \left[ \frac{\partial b(\bullet)}{\partial z} \frac{\partial z}{\partial w} \frac{\partial w}{\partial \alpha_t} g^i(\bullet) + w(\bullet) \frac{\partial b(\bullet)}{\partial z} \frac{\partial z}{\partial g} \frac{\partial g}{\partial \alpha_t} \right] \right) \qquad [7.27]$$

$$\frac{\partial \widetilde{H}}{\partial \beta_t} = \sum_{i=1}^{\alpha L_{TOT}} \left( \frac{\partial \pi_A^i(\bullet)}{\partial \beta_t^i} + \rho\lambda_{t+1} \frac{\partial q^i(\bullet)}{\partial \beta_t^i} + \rho\mu_{t+1} w(\bullet) \frac{\partial b(\bullet)}{\partial z} \frac{\partial z}{\partial g^i} \frac{\partial g^i}{\partial \beta_t^i} \right) \qquad [7.28]$$

$$\frac{\partial \widetilde{H}}{\partial P_t} = \sum_{i=1}^{\alpha L_{TOT}} \left( \frac{\partial \pi_A^i(\bullet)}{\partial P_t^i} + \rho\lambda_{t+1} \frac{\partial q^i(\bullet)}{\partial P_t^i} + \rho\mu_{t+1} w(\bullet) \frac{\partial b(\bullet)}{\partial z} \frac{\partial z}{\partial g^i} \frac{\partial g^i}{\partial P_t^i} \right) \qquad [7.29]$$

$$\frac{\partial \widetilde{H}}{\partial E_t} = \sum_{j=1}^{n} \left( \frac{\partial \pi_F^j(\bullet)}{\partial E_t^j} - \rho\mu_{t+1} \frac{\partial h^j(\bullet)}{\partial E_t^j} \right) \qquad [7.30]$$

$$\rho\lambda_{t+1} - \lambda_t = -\frac{\partial \widetilde{H}}{dQ_t} = \sum_{i=1}^{\alpha L_{TOT}} \left( -\frac{\partial \pi_A^i(\bullet)}{\partial Q_t^i} - \rho\lambda_{t+1} \frac{\partial q^i(\bullet)}{Q_t^i} - \rho\mu_{t+1} w(\bullet) \frac{\partial b(\bullet)}{\partial z} \frac{\partial z}{\partial g^i} \frac{\partial g^i}{\partial Q_t^i} \right) \qquad [7.31]$$

$$\rho\mu_{t+1} - \mu_t = -\frac{\partial \widetilde{H}}{\partial X_t} = \sum_{j=1}^{n} \left( -\frac{\partial \pi_F^j(\bullet)}{\partial X_t} - \rho\mu_{t+1} \left[ \frac{\partial b(\bullet)}{\partial X_t} - \frac{\partial h^j(\bullet)}{\partial X_t} \right] \right) \qquad [7.32]$$

This system does not give singular solutions but can provide a set of solutions for given couples of values of $P$ and $\beta$, and assuming an average value for $Q$. The right hand terms in equations [7.27], [7.28] and [7.29] account for the external effects on the fishery sector due to marginal changes in $\alpha$, $\beta$ and $P$ respectively. The externalities are estimated at the current value co-state variable $\mu$, representing the shadow value of the forgone fish production. According to equation [7.27] the policy maker should support wetlands reclamation for agricultural use up to the extent where the marginal revenue to the agricultural sector equal the marginal cost to the downstream fishery sector. Equation [7.28] says that the maximum social benefit is attained at a percent of area under fallow where the agricultural marginal revenue equal the sum of both the marginal cost to the agricultural sector due to soil nutrient loss (on-site externality) and the marginal cost to the fishery (off-site externality). Equation [7.29] shows how the social optimum amount of fertilizer application requires that the sum of both the marginal revenue and the marginal benefit to the agricultural sector of improved soil nutrient balance would be equal to the marginal cost to the fishery sector.

## 7.3. Fish stock indicators for the Kenyan side of Lake Victoria: wetlands extent and fertilizers price

Traditional stock assessment involves estimating the current status of a stock using one of the classic single-stock models (Ricker, 1954; Schaefer, 1954; Beverton & Holt, 1957). In most assessments, environmental conditions are taken to be exogenous to the problem. However, some studies consider these conditions as endogenous to fishery models.

Environmental conditions in many fisheries are directly affected by other, linked economic activities. Land-use change, for example, has had major effects on fisheries through the role of coastal systems in providing habitat for exploited species, and in regulating productivity (IWCO, 1998). The nutrient-retention function of coastal wetlands is an important determinant of stock biomass of exploited species. The disposal of inorganic and organic waste from agriculture, industrial, and domestic sources can also have significant effects. In many estuaries, the oxygen demand of organic waste leads to anaerobic conditions. Sewage and fertilizer run-off enhance algal growth, which gives rise to toxic tides and associated mass mortality.

We use the interaction between land use and the fishery to identify catchment-based indicators, in particular price indicators and wetland extent, which may anticipate changes in environmental variables driving stock assessment models. We then model the interactions between fertilizer applications, phytoplankton growth, and fish production to identify the functional relationships between fertilizer prices and fish stocks and yields. Because of lags within the system, this allows using both current prices and wetland extent to predict future changes in stock size.

### 7.3.1. Wetland area as fish stock indicator

We consider the role played by wetlands in moderating the effect of land based pollution and hence on eutrophication and fishery yields. Several studies have considered the linkage between wetlands and fisheries in coastal and estuarine systems (Turner, 1982; Swallow, 1994; Barbier & Strand, 1997; Barbier & Sathirathai, 2002). All these studies focus on the ecological function of wetlands in supporting fisheries by serving as both spawning ground and nursery for fry. However, to our knowledge, there is no study of the value of wetlands in terms of their impact on water quality. Although the nutrient retention function of coastal and freshwater wetlands is known to be an important determinant of stock biomass in capture fisheries, the economic significance of this has not been evaluated.

The changes in aquatic systems that follow from changes in nutrient loading due to land-based economic activities are said to be an externality of those activities. That is, those whose activities on land damage aquatic systems are not confronted by the cost of that damage. While it is recognised that internalisation of such externalities requires an understanding of the interactions between terrestrial and aquatic activities, there are few attempts to model the problem. Early approaches either focused on simple correlations between changes in watersheds and changes in fisheries, or else identified the consequences for fisheries if the linkages were of varying strength (Ruitenbeek, 1989; Hodgson & Dickson, 1998). Limnologists have investigated the consequences for freshwater aquatic systems of changes in land use, vegetative cover, and fertilizer regimes within the watershed (Postel & Carpenter, 1997; Carpenter & Pace, 1997; Carpenter & Cottingham, 1997). More recently, economists have considered the theoretical problems posed by the interaction between users of lake and catchment resources (Mäler *et al.*, 2003; Carpenter *et al.*, 1999a). Nevertheless, there are few attempts to model the interactions in real systems, or to estimate the value of land-water externalities of this kind.

Our model first links nutrient loading and phytoplankton density (i.e. chlorophyll-a concentration) using functional forms drawn from empirical studies. The dependence of the overall phytoplankton biomass on nutrients (total phosphorus *TP* and total nitrogen *TN*) is a general feature in freshwater ecosystem and many chlorophyll/nutrient relationships have been published (§ 4.5.1). In Lake Victoria there is evidence of *TN* being the limiting nutrient in offshore waters while *TP* concentration seems to be the key element influencing phytoplankton growth in inshore waters (Mugidde, 2001). Since our analysis is limited to the Kenyan side, which is mostly represented by inshore waters, it would be appropriate to consider the chlorophyll-*TP* functional relationship. Without the support of specific studies for Lake Victoria, we use the Dillon & Rigler (1974b) equation:

$$W_t = 0.0731 TP_t^{1.449} \qquad [7.33]$$

where $W_t$ is chlorophyll-a concentration (mg $m^{-3}$) and $TP_t$ is total phosphorus concentration (mg $m^{-3}$) in the lake.

To convert phosphorus flows in concentrations we draw on the substantial number of limnological models developed for lakes (§ 4.2). In general, lakes are open flow-through systems in which the mass of a given element is determined by the dynamic balance between supply through the inflows, loss through the outflows, and internal sources and sinks (Andersen, 1997). Prediction is commonly based on a family of phosphorus mass balance models directly descending from the pioneering work of Vollenweider (1968b, 1969). Under steady state conditions Vollenweider's model can be used to represent a description of overall nutrient processes in a lake and as such it can be used to evaluate or predict changes in the nutrient balance of a water body as a response to a step input, when the input changes from one constant level to another load level. By taking into account atmospheric deposition and scaling the unit of measurements, the limnological expression is:

$$TP(z_t) = \frac{(z_t + D_{TP}) * 10^9}{Q_{lake} + \frac{s_{TP}}{H_{lake}} V_{lake}} \qquad [7.34]$$

where *TP* is the concentration of phosphorus (mg $m^{-3}$); $z_t$ is the superficial loading (tonnes $yr^{-1}$); $D_{TP}$ is the long term average atmospheric deposition (t $yr^{-1}$); $V_{lake}$ is the lake volume

($m^3$); $Q_{lake}$ is the hydraulic superficial outflow from the lake ($m^3$ $yr^{-1}$); $H_{lake}$ is the mean depth of the water body (m); and $s_{TP}$ is the settling velocity rate for phosphorus (m $yr^{-1}$).

By combining equations [7.5], [7.33], and [7.34] we obtain the general expression for the environmental variable which influences fish growth and fishery yields. The environmental variable *W*, chlorophyll-a concentration, can therefore be represented as a function of nutrient loading (*g*), wetland area ($L_w$) in ha and a series of constants or long term averages:

$$W_t = .8034 * 10^{12} \left( \frac{g_t \exp\left( -\frac{k_{TP} L_W * 10^4}{v} \right) + D_{TP}}{Q_{lake} + \frac{s_{TP} V_{lake}}{H_{lake}}} \right)^{1.449} \qquad [7.35]$$

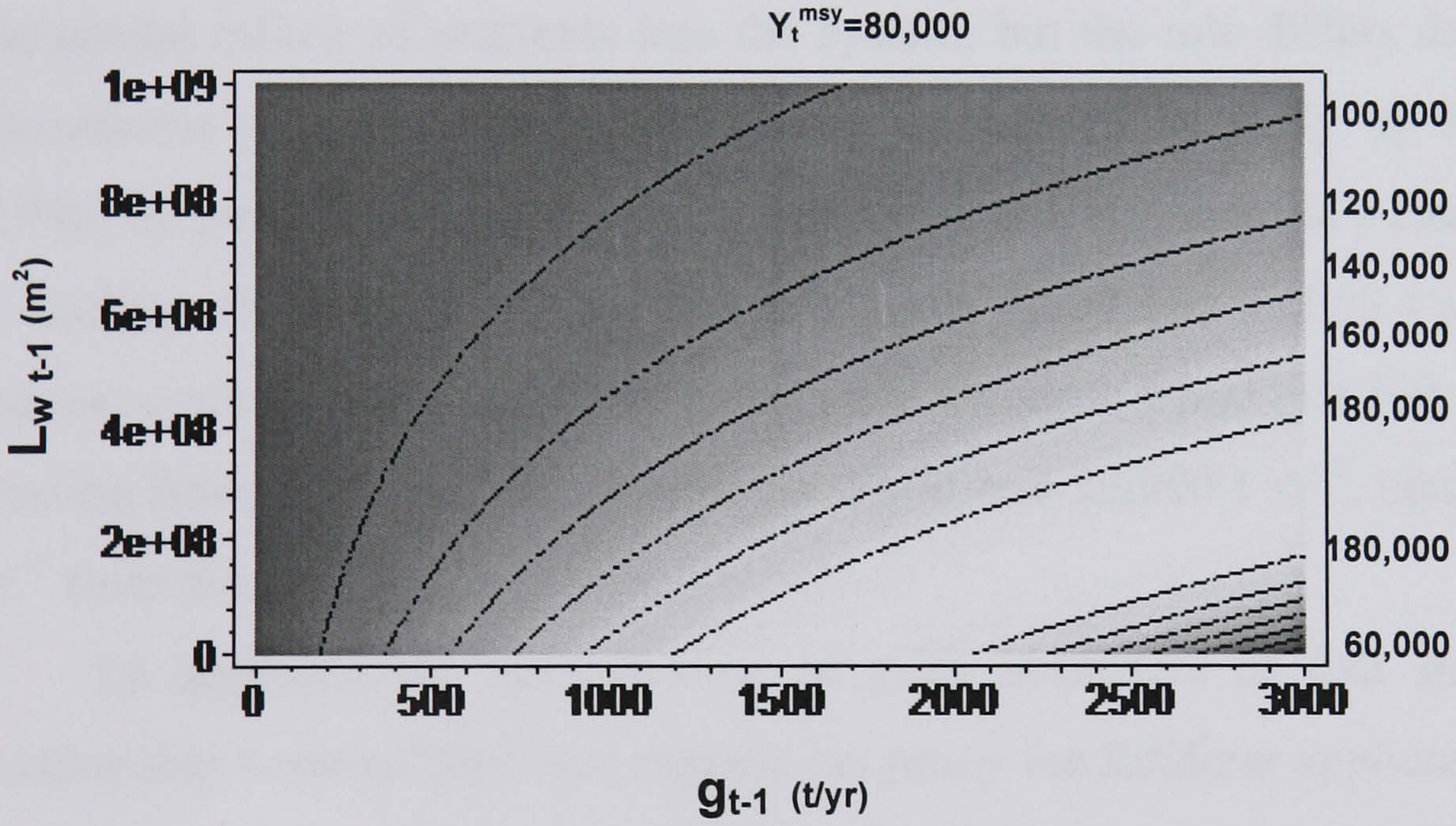
Notice that this expression assumes that all the phosphorus loading from the watershed is flowing through wetlands. In most of cases this will overestimate the nutrient retention function of wetlands. More complexity could be added to the model by considering hydraulic inflow (*v*) as variable, taking into account changes in precipitation patterns. But we do not add this here.

In applying the model to the Kenyan fishery, and watershed, of Lake Victoria we use the long term average of 1460 t P $yr^{-1}$ for atmospheric deposition (COWI, 2002), $5376*10^6$ $m^3$ $yr^{-1}$ for the hydraulic inflow (Balirwa & Bugenyi, 1988), the areal removal rate constant ($k_{TP}$) of 12 m $yr^{-1}$ (Kadlec & Knight, 1996), and a settling velocity for total phosphorus ($s_{TP}$) of 16 m $yr^{-1}$ (Chapra, 1975). We also assume an average depth of 12 m and a volume of $5.3664*10^{10}$ $m^3$ for the Kenyan sector of Lake Victoria, and that the hydraulic outflow is compensated ($Q_{lake} = 0$) by inflowing water from the Ugandan and Tanzanian side of the lake with the same concentration of phosphorus. This assumption is known in the engineering literature as the continuously stirred tank reactor (Reckhow & Chapra, 1983) and is equivalent to consider the lake's waters as completely mixed.

The predictive power of equation [7.35] can be tested by comparing predicted and observed values of chlorophyll-a concentrations. Given long term averages for both atmospheric deposition and superficial inflow of phosphorus (COWI, 2002) the predicted value of $W = 18.85$ mg $m^{-3}$ is close to the long term (1989-1998) average of 16.97 mg $m^{-3}$ from the values estimated using the simulation package in Ecopath, and well approximates the average value between inshore observations of 10.7-38.1 mg $m^{-3}$ (Kenyanya, 1999), 31.4 mg $m^{-3}$ (Ochumba & Kibaara, 1989) and offshore measurements of 6.3-14.1 mg $m^{-3}$ (Kenyanya, 1999) for the Kenyan waters of Lake Victoria.

Given [7.35] and allowing for a delay in the impact of water quality change on the fish stock growth as in equation [6.1], we can calculate the changes in the steady state level of fish yield under various management regimes ($Y_{msy}$ maximum sustainable yield, $Y_{oa}$ open access, $Y^*$ profit maximising) corresponding both to the level of phosphorus load at $t$-1 into the wetlands and the area under wetlands at $t$-1 for the entire Kenyan side of Lake Victoria. These steady state solutions can then be represented by a surface. An example is provided for the case of the steady state catch under maximum sustainable yield regime for the Kenyan fishery of Lake Victoria in Figure 7.1.

**Figure 7.1:** Fish yield (t/yr) under MSY regime as function of wetland area and superficial loading of phosphorus for the Kenyan fishery of Lake Victoria.



Without at the moment considering the impact of changes in the extent of the wetlands, the yield under maximum sustainable yield regime (and the profit-maximising regime) would peak at a nutrient loading of around 1,550 t $yr^{-1}$ (Figure 7.1). Since actual loading of phosphorus from the Kenyan basin is currently around 1,925 t $yr^{-1}$ (COWI, 2002), well above the turning point, we would expect the lake to move towards an eutrophic condition and fishery yields to decline. Any further increase in loading will accordingly reduce fishery yields in any of the three management regimes. It is in this context that we consider the importance of the wetlands at the margins of the lake.

### 7.3.2. Fertilizer price as fish stock indicator

The prices driving changes in land use may precede changes in the state of a fishery. Economic indicators of a fishery itself (e.g. prices, catch per unit effort, employment, investment, productivity, income distribution; Padilla *et al.*, 1995) may therefore act as a backdrop to studies of how the fishery responds to price-driven changes in those sectors with which it is physically linked, ultimately using those prices to predict changes in the fishery. For some time, economists have argued the importance of understanding the linkages between activities in watersheds (Ruitenbeek, 1989; Hodgson & Dickson, 1988) or coastal systems (World Bank, 1995), but little attention has been paid to the potential this offers for the development of new indicators, particularly price indicators.

In Lake Victoria, fertilizer application clearly plays an important role in determining the annual inflow of nutrients into the system, but the role differs depending on whether phosphorus (P) or nitrogen (N) is being considered. A study by Kirugara & Nevejan (1996) on pollution sources in the Kenyan part of Lake Victoria's catchment reported that P loadings from fertilizer use varied between 5,000 and 22,000 t $yr^{-1}$, whereas animal manure contributed roughly 3,000 t $yr^{-1}$, and domestic waste just 100 t $yr^{-1}$. By contrast, N loading from fertilizers varied between 4,000 and 22,000 t $yr^{-1}$, compared with 40,000 t $yr^{-1}$ from manure.

To approach the identification of price indicators of fish stocks, we estimate a relationship between fertilizer imports (as proxy for fertilizer application in the catchment area) and chlorophyll-*a* concentration for the Kenyan waters of Lake Victoria. We suggest a simple water quality ($W_t$ in mg Chl-*a* $m^{-3}$) "production function" of the Cobb–Douglas form, in which fertilizer imports ($F_t$ in thousand tonnes) are the sole argument. The best result was obtained using a one-year lag for the dependent variable ($R^2 = 0.55$; adj $R^2 = 0.50$; p-value = 0.01; t-ratio in parenthesis):

$$\ln W_t = -5.6568 + 0.6825 \ln F_{t-1} \qquad [7.36]$$

$$\qquad (2.11) \qquad (3.16)$$

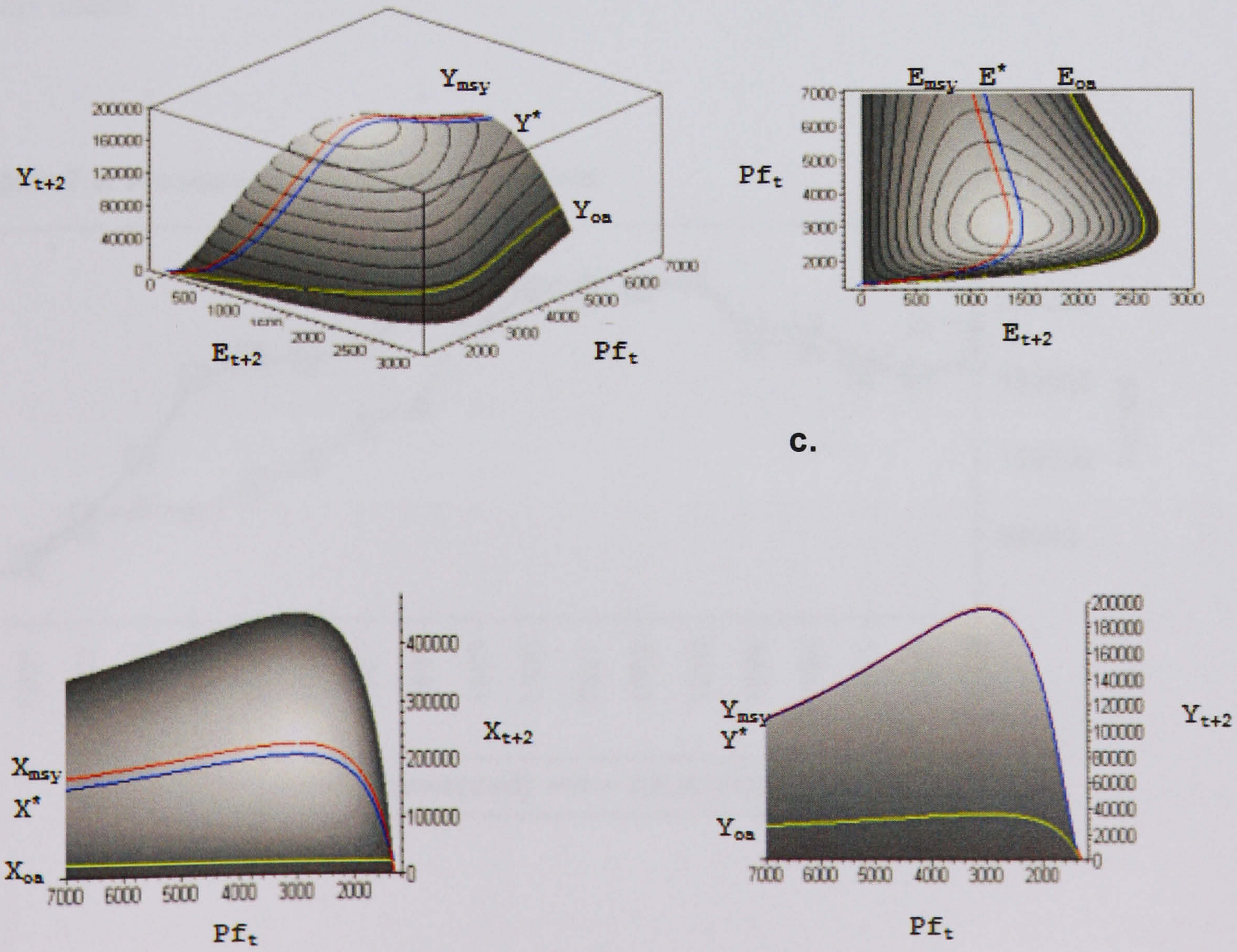
As fertilizer use depends on its prices (*Pf*, defined in 1987 constant Ksh per tonne), we estimate the short-run price elasticity of fertilizer demand ($R^2 = 0.38$; adj $R^2 = 0.35$; F-statistic p-value = 0.003):

$$\begin{array}{l} \ln F_t = 22.0741 + 1.2011 \ln Pf_t \\ \qquad\quad (7.74) \qquad (3.43) \end{array} \qquad [7.37]$$

Equations [7.36] and [7.37] allow explaining changes in fish stocks and yields at time $t$ as a function of $Pf_{t-2}$ via the effect of the latter on $W_{t-1}$. The yield-effort curve can then be explained as function of the two-year lagged price of fertilizer (Figure 7.2a). The relations between fertilizer prices and fish biomass and yields under each management regime are indicated in Figures 7.2b and c, respectively.

**Figure 7.2:** Fish growth and yield effort curves as function of fertilizer price.

**a.**



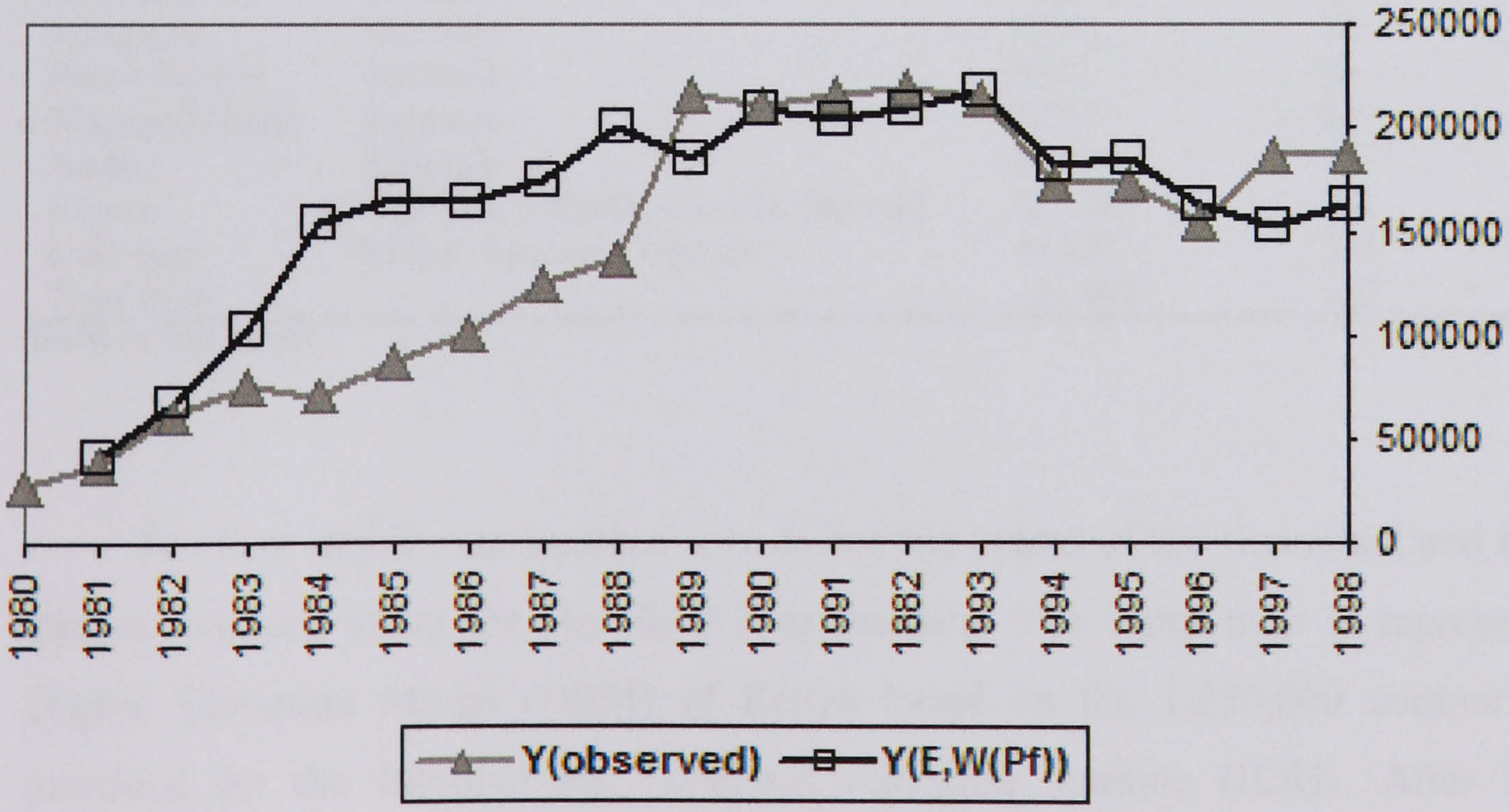
**b.**

**c.**

It is intuitive that as $Pf_t$ tends to zero, so will stocks and yields under all management regimes whenever the price elasticity of fertilizer demand is positive. It is less intuitive that as $Pf_t$ rises, stocks and yields will first increase and then decrease, but this follows directly from the non-linear relation between nutrient loading and biomass. If the lake is initially oligotrophic, a fall in $Pf_t$ will induce farmers to apply more, so increasing nutrient loading and hence fish production. However, as $Pf_t$ falls further and fertilizer applications increase, ultimately the lake will converge to an eutrophic state, with increasingly adverse effects on fish production. Figures 7.2b and c indicate that both biomass and yield peak at a little over 3,000 Ksh (US$ 182).

Given these relationships, fertilizer prices can be used as leading indicators of changes in fish biomass and yield. In other words, steady-state solutions for both can be predicted two years in advance. The predicted yields using the observed trend in real fertilizer prices $[Y(E,W_{Pf})]$ closely approximate (Figure 7.3) observed yields through the 1990s. Fertilizer prices overestimate the increase in yields during the 1980s, but since then have been a reasonable leading indicator of yield and stock changes in the fishery two years ahead.

**Figure 7.3:** Predicted against observed fish yield.



## 7.4. The Yala basin: watershed delineation and soil erosion estimation through GIS analysis

The Yala basin, on the Kenyan side of Lake Victoria, is one of the twelve main watersheds draining into the lake (Table 7.1). It is often associated with the Nzoia river basin representing the second watershed for size and population density. The peculiarity of this basin is that its main river, the Yala river, drains into a vast wetland, the Yala wetland, on the edge of Lake Victoria. The swamp is one of the most extended wetlands around the lake and is under severe threat by agricultural expansion. Given this characteristics, this watershed and the associated wetland represents the perfect area of study for the application of our model.

**Table 7.1:** River basins of Lake Victoria.

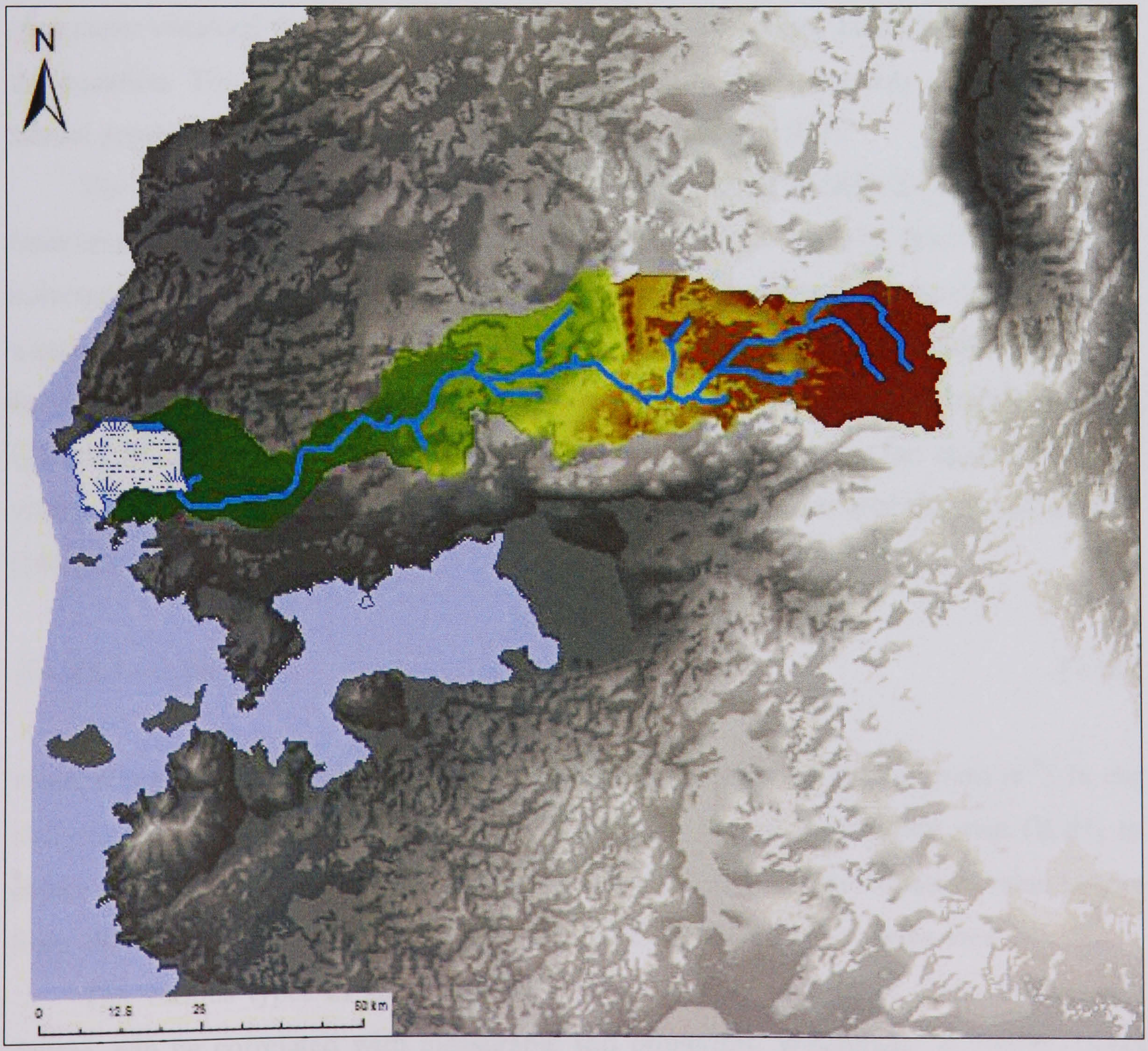
| River basin | Country | Basin size (km$^{-2}$) | Population density (people km$^{-2}$) |
|---|---|---|---|
| Nzoia/Yala | Kenya | 15,143 | 221 |
| Nyando | Kenya | 3,517 | 174 |
| Miriu Sondu | Kenya | 3,583 | 220 |
| Gucha | Kenya | 6,612 | 224 |
| Mara | Kenya, Tanzania | 13,915 | 46 |
| Gurumeti | Tanzania | 12,290 | 21 |
| Mbalageti | Tanzania | 5,702 | 37 |
| Duma/Simiyu | Tanzania | 9,702 | 50 |
| Magoga/Muame | Tanzania | 5,104 | 88 |
| Isonga | Tanzania | 8,972 | 48 |
| Kagera | Tanzania, Uganda, Rwanda, Burundi | 59,178 | 181 |
| Lake edge | Kenya, Tanzania, Uganda | 40,682 | 133 |
| *Total basin* | | *184,400* | *135* |

*Source:* ICRAF (2000).

The first step in our analisys is to define the extent of the watershed and identify its stream network using the ArcGis 9.2 application. The input map is represented by a Digital Elevation Model (DEM) of Kenya based on the 1:250,000 contour map and provided by the International Livestock Research Institute (ILRI). After filling the potential "sinks" the DEM is used to calculate the flow direction grid. The output in the flow direction grid is an integer grid whose values range from 1 to 255 representing a definite direction from the center of each cell. Flow direction is used to track the path of the sediment from the source grid cell to the stream, identifying the direction of the steepest drop in elevation to one of the surrounding eight cells.

Upstream area and stream network delineation can then be obtained by performing a flow accumulation procedure on the flow direction grid. Depending on the flow path, the number of upstream cells flowing into each downslope cell can be accumulated. Cells that have high accumulation are areas of concentrated flow and may be used to identify stream channels. When a threshold is set, the channel network can be delineated for the watershed. Any cell with a number of cells upstream less than that threshold is considered to be an overland cell and those with a number of cells upstream equal to or greater than the threshold are considered to be channel cells.

**Figure 7.4:** Delineated Yala watershed and its rivers network.



We use a digitalized map of the wetlands of Kenya and a shape file identifying the Lake Victoria edge, both produced by ILRI. These maps allow positioning the pour point, which identifies the basin upstream to that point from the flow accumulation grid using the "watershed function" in ArcGis. The analysis produces an output map identifying the Yala wetland, Yala river network and its watershed overlayed with the DEM of the region (Figure 7.4). According to the digitalized map of the wetlands of Kenya (ILRI) the area of Yala wetland is 22,386 ha. Our estimation of the extent of the Yala watershed, excluding the wetland, is 286,641 ha, occupying parts of the Siaya, Vihiga, Kakamega, Nandi, and Uasin Gishu districts.

After watershed delineation, we focus on the estimation of soil erosion and the associated sediment delivery that are key factors in determining nutrient loading. We apply the Universal Soil Loss Equation (Wischmeier & Smith, 1978) through a GIS approach. In this context, the product of the USLE equation [2.50] would be identified in a map raster obtained from the overlay of the spatial distributions maps for each factor in the equation. The inputs of our analysis are a soil characteristics map of Kenya, mean annual precipitation map and land cover map, all produced by ILRI.

The rain erosivity index ($R$) equals the kinetic energy of rainfall ($E$) multiplied by $I_{30}$ (maximum intensity of rain in 30 minutes expressed in cm per hour). This index corresponds to the potential erosion risk in a given region where sheet erosion appears on a bare plot with 9% slope. Rain erosivity can be determined from rainfall data. We adopt a functional relationship estimated by Lufafa *et al.* (2003) for a microcatchment study in Uganda which uses coefficients derived by regressing long-term rainfall against erosivity values determined by Moore (1979) for the Lake Victoria region applying the $EI_{30}$ index (Morgan, 1986):

$$R = 47.5 + 0.38\Omega \qquad [7.38]$$

where $R$ expresses rain erosivity (joules $m^{-2}$); and $\Omega$ is the annual rainfall (mm $yr^{-1}$). In our analysis we apply function [7.38] using the Kenya rainfall distribution map (ILRI) to obtain the layer for the rain erosivity factor (Figure 7.5a). The estimated $R$ value ranges from 389 to 883 (joules $m^{-2}$), with a mean value of 631 for the entire watershed.

The various types of soils have different degrees of erodibility. This, to a certain degree, can be correlated with measurable soil properties. Wischmeier & Smith (1978) report the following general $K$ values according to soil characteristics: 0.1 for sand and loamy sand, which is low because of high infiltration; 0.2 for clay, which is low because of strong cohesion; 0.25 for sandy loam and silt loam; 0.3 for clay loam; and 0.4 for loam, which is high because of low infiltration and weak cohesion. In our study soil erodibility factor (*K*) is obtained from a general soil classification map for Kenya (ILRI). Soil erodibility classes are then derived from generic soil properties applied to each soil category identified by the map. The resulting map layer (Figure 7.5b) for our area of study shows *K* value ranging from 0.2 to 0.4 with a mean value of 0.28.

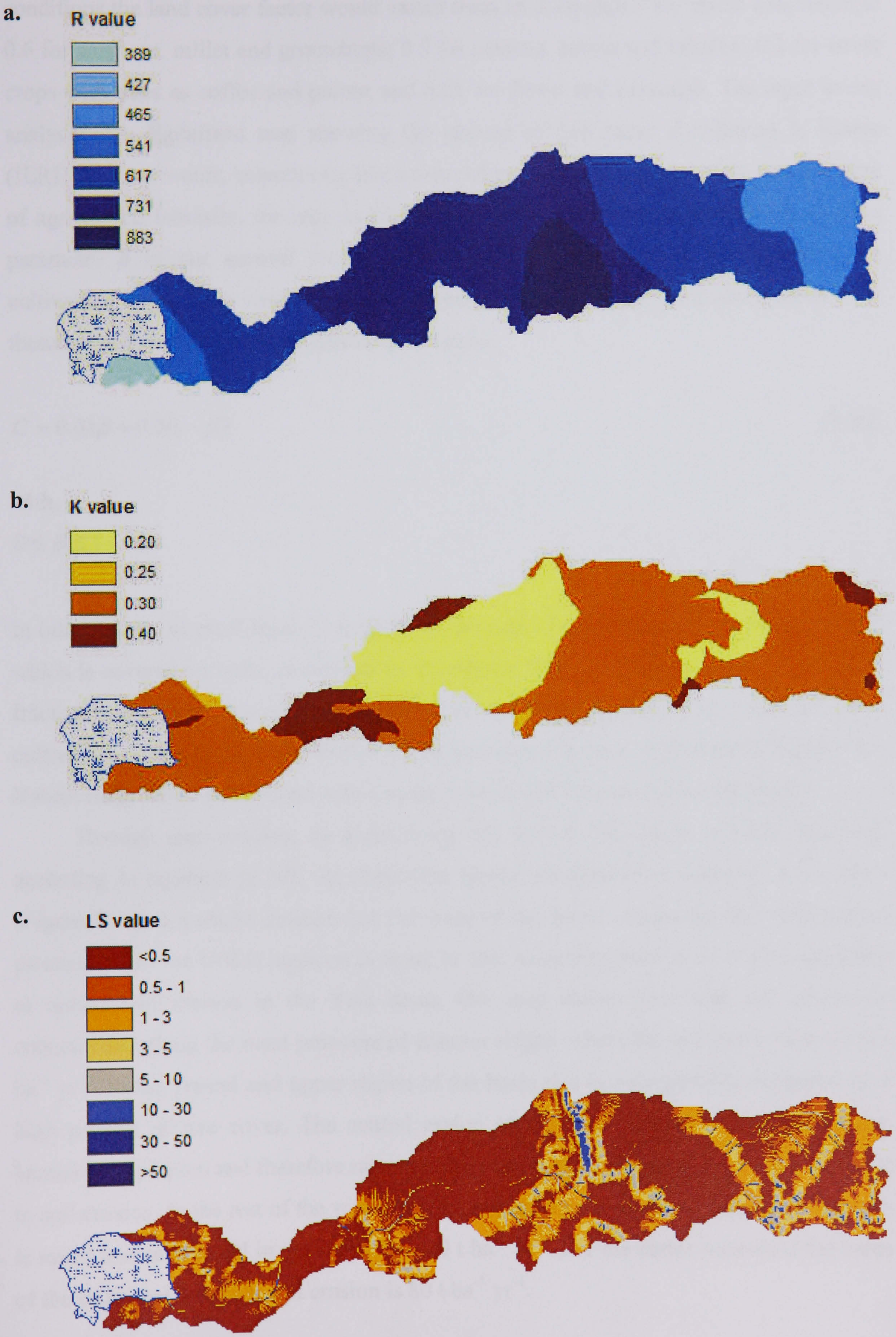
In the USLE the influence of terrain on erosion is represented by the length-slope (*LS*) factor which indicates that soil erosion increases with increase in slope angle and slope length. The *L* factor and *S* factor are usually considered together to combine the effect of slope and slope-length, which basically reflects the terrain on a given site. To compute the *LS* factor we use the approach developed by Moore & Burch (1985):

$$LS = \left(\frac{As}{22.13}\right)^{0.4} * \left(\frac{\sin\sigma}{0.0896}\right)^{1.3} \qquad [7.39]$$

where *As* is the specific catchment area, i.e. the upslope contributing area per unit width of contour (or rill), in $m^2$ $m^{-1}$; and $\sigma$ is the slope angle in degrees. The upslope contributing area is estimated in ArcGis using the "flowaccumulation" function. The resulting layer is then multiplied by the cell size expressed in meters (in our case 225 m), obtaining a layer which expresses the *As* value for each pixel. The slope angle in degrees $\sigma$ is calculated in ArcGis using the function called "slope" with option "percentrise" which is 100 times Tan$\sigma$. Then $\sigma$ is calculated using "Atan" function in ArcGis. The two map layers expressing *As* and sin$\sigma$ are then multiplied according to equation [7.39], obtaining a map showing the spatial distribution of the *LS* factor (Figure 7.5c). The estimated LS factor varies from about 0 to an extreme of 89, with a mean value of 1.13 for the entire watershed.

The land cover factor (*C*) is a simple relation between erosion on bare soil and erosion observed under a specific land use class. The *C* factor combines plant cover, its production level and associated cropping techniques. Previous studies (Wortmann & Kaizzi, 1998) in the Lake Victoria region used land-use erosion values ranging from 0.05 to 0.4: 0.4 for annual sole crops; 0.3 for annual intercrop associations and sole crop banana; 0.2 for banana intercrop associations. We would apply, however, the land cover

**Figure 7.5:** Estimated rain erosivity (a), soil erodibility (b), and Length/Slope factor map (c).



factor values reported by Wenner (1981) as representative for Kenya. For normal Kenya conditions the land cover factor would varies from an average 0.7 for maize cultivation to 0.6 for sorghum, millet and groundnuts; 0.5 for cassava, cotton and tobacco; 0.2 for cover crops with trees as coffee and palms; and 0.01 for forest and savannah. The input in our analysis is a digitalized map showing the percent of tree cover distribution in Kenya (ILRI). In other words, considering tree cover rather than fallow distribution, as a measure of agricultural intensity, the map in Figure 6.6a represents the spatial distribution of the parameter $\beta$ in our general model. We would assume maize as the representative cultivation in the Yala watershed. The spatial distribution of the land cover factor $C$ is therefore obtained through the following equation:

$$C = 0.01\beta + 0.7(1-\beta) \qquad [7.40]$$

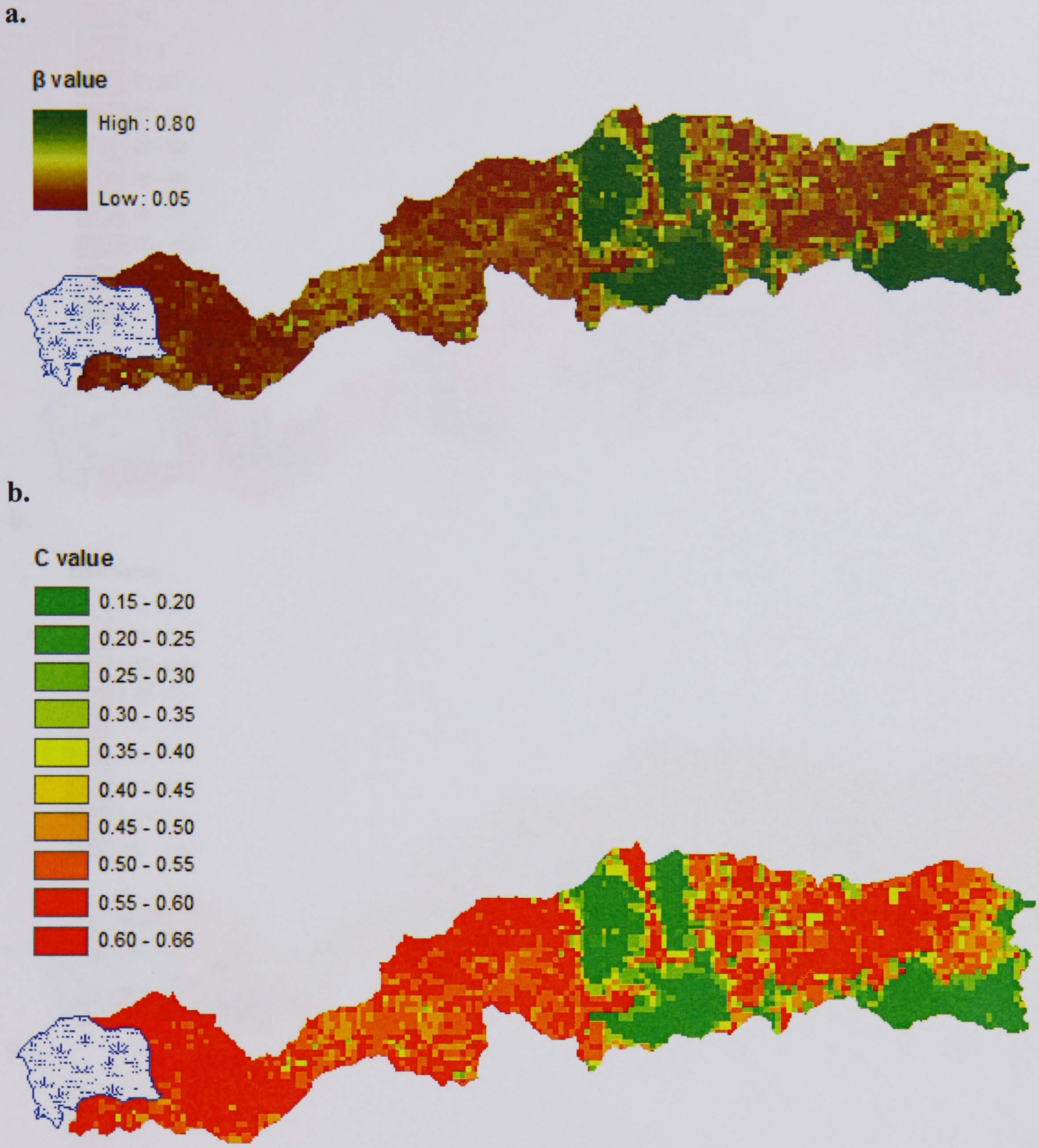
with,

$0 \leq \beta \leq 1$

In other words, at pixel level, $C$ is given by the sum of the percent ($\beta$) of the spatial unit which is covered by trees, multiplied by its relative land cover factor, plus the remaining fraction ($1$-$\beta$), which is crop land, multiplied by the respective land cover factor for maize cultivation. The estimated distribution of the land cover factor $C$ is showed in Figure 7.6b. Values range from 0.15 to 0.66 with a mean value of 0.47 for the entire watershed.

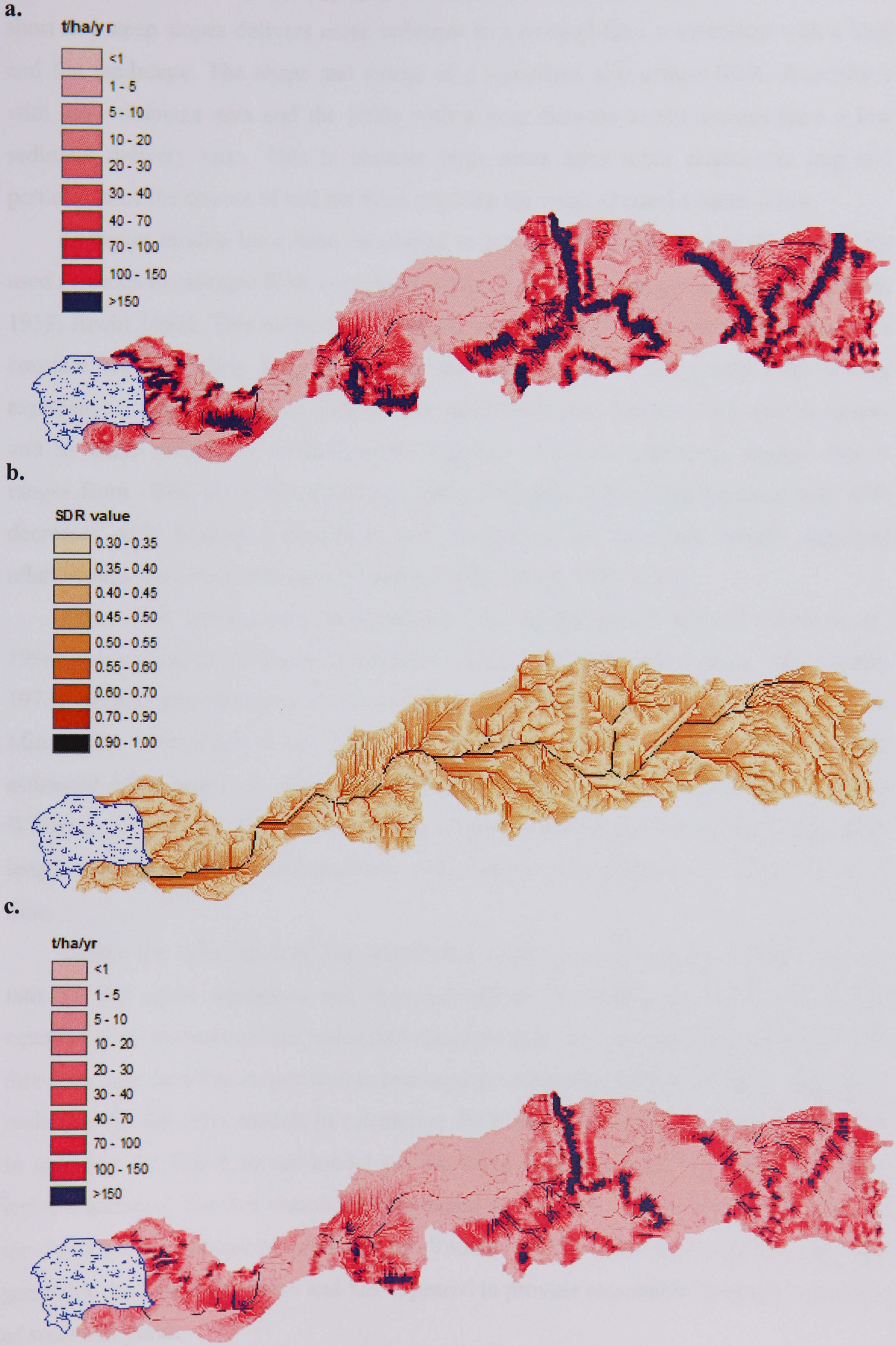
Through map overlay, by multiplying cell by cell the values of each map layer according to equation [2.50], we obtain the spatial distribution of potential soil erosion (Figure 7.7a). It must be noticed that the value of the factor expressing soil management practice ($P$) in the USLE equation is equal to one, assuming there is no soil management to reduce soil erosion in the Yala basin. Our map shows how high soil erosion is concentrated along the most pronounced contour ridges, where the soil loss is above 150 t $ha^{-1}$ $yr^{-1}$. In the central and upper region of the basin this is only partially mitigated by a high percent of tree cover. The central region of the basin has also the highest mean annual precipitation and therefore rain erosivity, and is prone to being the most vulnerable to soil erosion. In the rest of the watershed, associated to a lower slope degree, soil erosion is much less severe and in general below 20 t $ha^{-1}$ $yr^{-1}$. For the entire watershed the mean of the estimated potential soil erosion is 80 t $ha^{-1}$ $yr^{-1}$.

**Figure 7.6:** Percent tree cover distribution (a) and estimated land cover factor map (b).



In relatively large watersheds, most sediment gets deposited within the watershed and only a fraction of soil that is eroded from hillslopes will reach the stream system or watershed outlet. This fraction or portion of sediment that is available for delivery is referred to as the Sediment Delivery Ratio (SDR). This ratio is then multiplied by the predicted erosion rate to estimate the percent of eroded material/sediment/pollutant to reach the watershed outlet.

Figure 7.7: Estimated soil erosion (a), SDR distribution (b) and sediment yield map (c).



SDR is affected by the topographic features of the watershed. A watershed with short and steep slopes delivers more sediment to a channel than a watershed with a long and flat landscape. The shape and extent of a watershed also affects SDR. Watersheds with large drainage area and the fields with a long distance to the streams have a low sediment delivery ratio. This is because large areas have more chances to trap soil particles, thus the chance of soil particles reaching the water channel system is low.

Different models have been developed to estimate the SDR. One of the commonly used methods to estimate SDR at regional scales is the SDR-area power function (Maner, 1958; Roehl 1962). This method is expressed as the product of the catchment area by a constant and a scaling exponent which are both empirical parameters. The scaling exponent contains key physical information about catchment, sediment transport processes and its close linkage to rainfall-runoff processes. Field measurements suggest that it ranges from –0.01 to –0.025 (Walling, 1983; Richards, 1993), which means that SDR decreases with increasing catchment area. Following this approach several empirical relationships have been estimated (Vanoni, 1975; USDA, 1979, 1983).

Other SDR models have been based on the rainfall-runoff factors (Arnold *et al.*, 1996), slope gradient (Williams & Berndt, 1972), relief-length ratio (Maner, 1958; Renfro 1975), particle size (Walling, 1983) and water runoff travel time (Ferro, 1997; Ferro & Minacapilli, 1995; Ferro *et al.*, 1998). Sun & McNulty (1998) and Yagow *et al.* (1988) estimated SDR based on distance and distance-slope equations respectively, whereas Williams (1977) found the sediment delivery ratio is correlated with drainage area, relief-length ratio, and runoff curve numbers. They propose that SDR can be related to travel time.

Unlike the other models, the area-based equation calculates the average delivery ratio for the entire watershed and does not require the support of GIS analysis. This equation does not account for watershed characteristics such as land use, relief, and flow direction, and therefore it may lead to less accurate estimates. GIS on the other hand is the main tool for the other models in calculating the SDR for each grid cell of the watershed. In applying the USLE to our model we are using mean estimates representative of the entire watershed. For this reason the area-based approach it may be appropriate. We use the functional relationship proposed by Vanoni (1975) since this equation has been generally used for many years and has appeared to provide reasonable "average" estimates of sediment yield:

$$S_{DR} = 0.4724L^{-0.125} \quad [7.41]$$

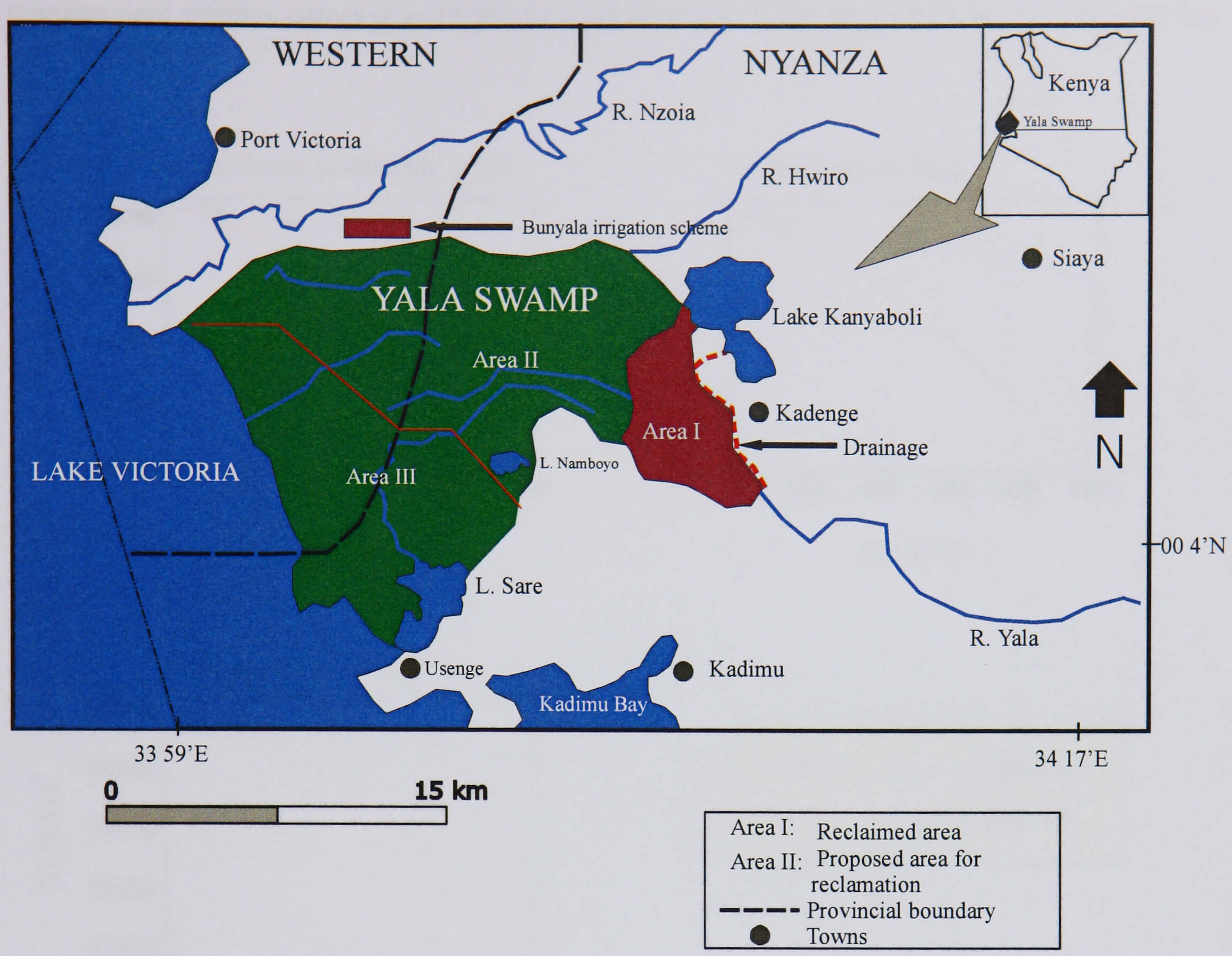
where $L$ is the catchment area (km$^2$). However, we apply equation [7.41] to GIS in order to get a more precise estimate of the mean SDR for the Yala watershed. In doing so we would use the flow accumulation grid, transformed in km$^2$, to estimate the uphill draining area for each cell. This would require considering the exponent in equation [7.41] in positive terms to get the highest value in proximity of drainage network, whilst the lowest for the most distant pixels. We therefore obtain a spatially distributed area-based SDR (Figure 7.7b) with values ranging from 1 to 0.31, and a mean value of 0.42. By multiplying the SDR grid by the soil erosion grid we obtain the sediment yield map (Figure 7.7c). This map estimates for each pixel the amount of sediments, as a proportion of the eroded soil that would finally reach the basin outlet.

In the following paragraph we will use the mean values from the rain erosivity ($R$ = 631), soil erodibility ($K$ = 0.28), length/slope factor ($LS$ = 1.13) and sediment delivery ratio ($SDR$ = 0.42) obtained by the produced rasters. The objective is to express sediment yield as a function of $\beta$ alone. Moreover, the estimated sediment yield map would be used to obtain a spatially distributed externality tax.

## 7.5. The impact of the conversion of the Yala wetland on the Kenyan fisheries of Lake Victoria

We are interested in evaluating the externality of wetland conversion in Lake Victoria, focusing on the Yala swamp, one of the largest wetlands in the Kenyan basin of Lake Victoria. The swamp has already been heavily affected by human development and much of the remaining area has been demarcated for 'reclamation' since the mid 1960s. After 2,300 ha (Area I in Figure 7.8) of the original 17,500 ha were drained in the 1960s, a further 9,200 ha (Area II) were scheduled for conversion to agriculture in the Lake Basin Development Authority Five Years Plan 1989-1993 (Mwakubo *et al.*, 2004). The total extent of the Yala wetland reported by Mwakubo *et al.* (2004) is slightly smaller than the area from the map of Kenya Wetlands (ILRI). However, given the scale of the latter study, the data reported by Mwakubo *et al.* (2004) seems more reliable.

**Figure 7.8:** The Yala wetland.



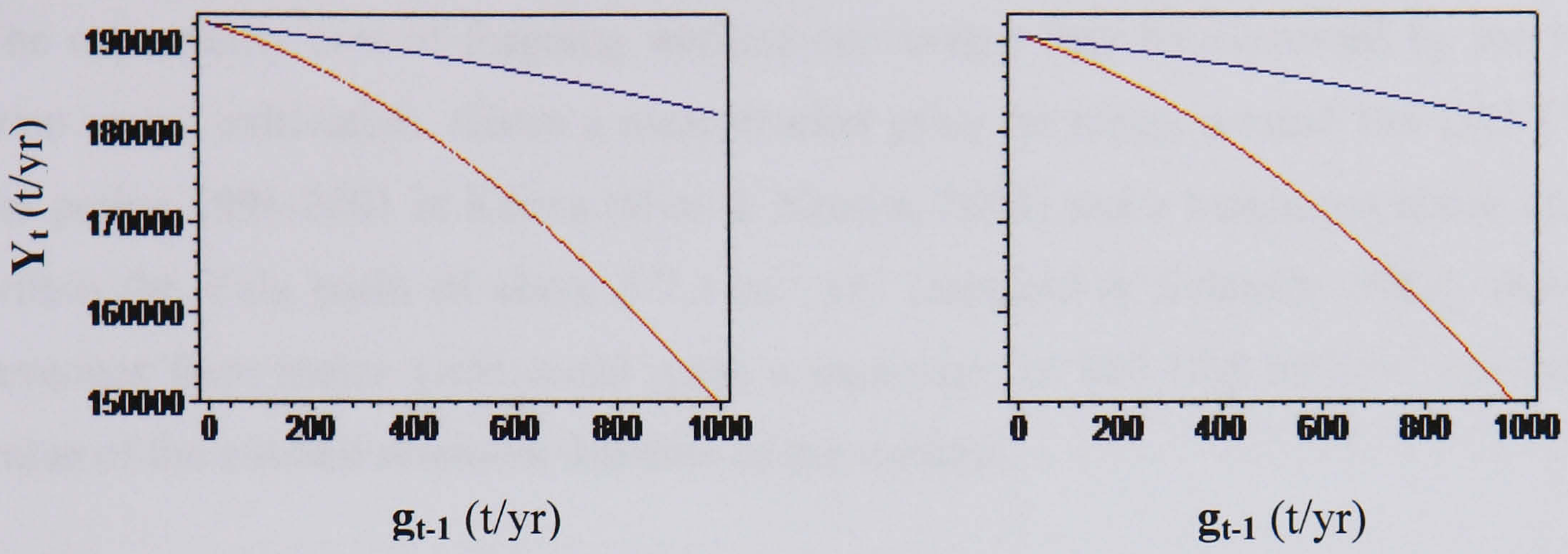
***Source:*** Mwakubo *et al.* (2004).

The next step is to model the impact of changes in the extent of the wetland under this and other reclamation programmes. We assume (a) constant values for wetland area ($L_W$), (b) a water discharge of $1114*10^6$ $m^3$ $yr^{-1}$ from the Yala river (Balirwa & Bugenyi, 1988), (c) an average of 102 t $yr^{-1}$ of phosphorus loading from the Yala catchment (COWI, 2002), and (d) accounting for the long term atmospheric deposition and phosphorus loading from the rest of the Kenyan watershed (COWI, 2002). We find that conversion of Area II will have a moderate negative impact on the steady-state path of the Kenyan fishery under any possible management regime. This is illustrated in Figure 7.9.

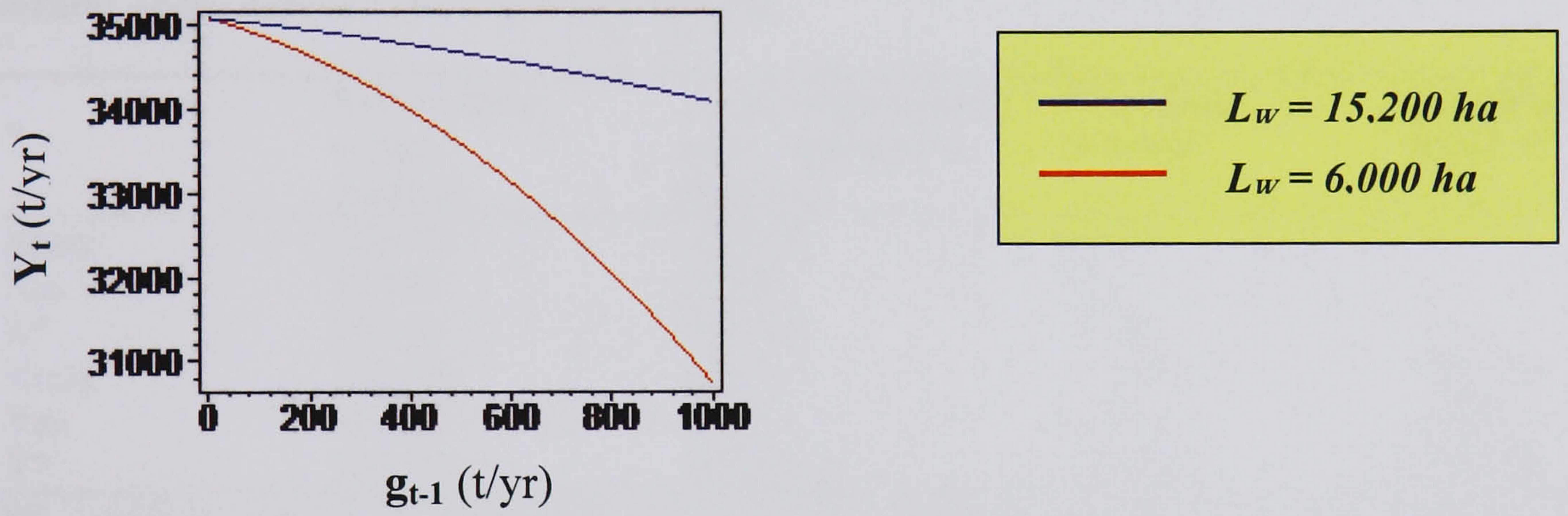
**Figure 7.9:** Prediction of steady-state yield paths for the Kenyan fishery under different management regimes before (*Lw*=15,200 ha) and after (*Lw*=6,000 ha) reclamation of Area II of the Yala wetland.

a) Maximum sustainable yield

b) Profit maximising



c) Open access



The impact on both fish stocks and yields, given the present loading of phosphorus from the Yala watershed, translates as a direct economic loss[9] of up to US$ 0.7 million per year, depending on the fishery regime (Table 7.2). At a discount rate of 5 per cent, this implies that the value of the converted wetland (its social opportunity cost) is US$ 14.7 million. Notice, though, that the loss of ecosystem services is increasing in the level of nutrient loading into the wetland. If nutrient loading into the wetlands grows at the same time as the wetland is converted, eutrophication of the lake and direct losses to the fishery will be accelerated. Moreover, while this is a first approximation of the value of the nutrient retention function of the wetland, it ignores other ecosystems services such as groundwater recharge, carbon sequestration or habitat provision. Hence the opportunity cost of the wetland that is scheduled to be converted, defined in terms of its nutrient

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[9] Assuming on average that the market price of the fish landed is 508.55 US$ per ton.

retention function only, is a lower bound of its value. Comparison with estimates of the value of the converted land, approximated by the sale price of that land, provides a simple check on the efficiency of the conversion plan. Based on the simplifying assumptions used to in this paper, conversion at less than US$1,600 per hectare is unlikely to be efficient. The opportunity cost of forgoing wetland conversion may be expressed by the revenues from maize cultivation. Given a mean market price for maize around 156 US$ $t^{-1}$ during the perion 1991-2001 in Kenya (Gitu & Nzuma, 2003) and a maximum maize crop yield within the Yala basin of about 5.7 t $ha^{-1}$ $yr^{-1}$ (Jaetzold & Schmidt, 1982), the forgone revenues from maize yield could reach a maximum of 889 US$ $ha^{-1}$ $yr^{-1}$, far below the value of the nutrient retention function of the wetland.

**Table 7.2:** Potential economic loss to the Kenyan fishery due to reclamation of Area II of the Yala wetland under different management regimes.

| | Steady state values (tonnes) | Steady state values after reclamation (tonnes) | Difference (tonnes) | Economic loss (US$/yr) |
|---|---|---|---|---|
| Xmsy | 224,777 | 223,965 | - 812 | — |
| Xoa | 21,595 | 21,595 | — | — |
| X* | 205,447 | 204,649 | - 798 | — |
| Ymsy | 191,348 | 189,968 | - 1380 | 701,799 |
| Yoa | 35,000 | 34,867 | - 133 | 67,637 |
| Y* | 189,933 | 188,555 | - 1378 | 700,782 |

*Note:* X and Y are respectively fish biomass and yield under maximum sustainable yield (msy), open access (oa), and profit maximising (*) regimes.

### 7.5.1. The land use model for the Yala basin

After a preliminary estimation of the economic loss to the fishery due to partial conversion of the Yala wetland, we now extend the analysis to include the variables expressing fertilizer application and land use/cover. Indeed, the link between fertilizer use and fishery has already been tested (§ 7.3.2), since we show the relationships between price of fertilizers and fish stock dynamics in the Kenyan waters of Lake Victoria. We therefore apply our general model and assumptions (§ 7.2) to the Yala basin.

The several functional relationships of the model are obtained using regional specific parameters from the literature. The nutrient runoff function [7.7] and soil nutrient dynamics [7.9] require site specific estimates of the soil bulk density, soil erosion, phosphorus content in the harvested crop, atmospheric deposition and nutrient build up from fallow rotation. Mean nutrient concentration (kg $t^{-1}$) in the soil depends on the specific soil bulk density for a given thin of superficial layer of soil. The Ministry of Agriculture (1987) reports for the Siaya District a soil bulk density of 1.32 g $cm^{-3}$. Considering the 20 cm of topsoil layer, it gives a soil weight of 2,640 t $ha^{-1}$, which is equivalent to a soil bulk factor $\theta$ = 1/2640 = 0.000379. To obtain the soil phosphorous concentration in terms of kg per tonne of soil we can multiply by $\theta$ any nutrient value expressed in kg $ha^{-1}$.

Soil erosion ($A_t$) is estimated using the Universal Soil Loss Equation approach (§ 2.4.1). The parameters for the USLE are astimated as mean values from the entire watershed using GIS application as showed in section § 7.4. The land cover factor ($C$) would be a function of the percent of tree cover as in equation [7.40]. Soil erosion, in terms of t $ha^{-1}$ $yr^{-1}$, would then be expressed by the following relationship:

$$A_t(\beta_t) = 631 * 0.28 * 1.13 * (0.01\beta_t + 0.7(1-\beta_t)) * 1 = 139.7539 - 137.7574\beta_t \qquad [7.42]$$

It must be noticed that we consider soil management factor as ininfluent in our model and therefore setting it to the value of one. To overcome the potential overestimation of soil erosion (§ 2.4.1.6), a sediment delivery ratio ($S_{DR}$) is applied. The $S_{DR}$ is also estimated in a GIS environment at watershed level (§ 7.4). Phosphorus runoff due to erosion is estimated from average soil phosphorus concentration. Because fine particles are dislodged first in the process of erosion, eroded soil is richer in nutrients than soil in situ. Therefore, an enrichment factor of about 1.5 (Avnimelech & McHenry, 1984; Stocking, 1984) is generally used to multiply the estimated phosphorus content in eroded soil.

Atmospheric deposition may be one of the mayor sources of nutrient loadings. It is estimated that for Lake Victoria it may contribute to the 71.6% and 35.7% of the entire loading of nitrogen and phosphorous respectively (Scheren *et al.*, 2000). Over the last decades it has been reported a considerable increase of phosphorous concentration in rainwater, from 0.053 μmol $l^{-1}$ in 1958 to 3.4 μmol $l^{-1}$ in 1991, while nitrogen concentration has been substantially unchanged, from 64 μmol $l^{-1}$ to 66 μmol $l^{-1}$ (Bootsma & Hecky, 1993). Lindenschmidt *et al.* (1998) estimated, from concentration of nutrients in rainfall reported by Scheren (1995) and Bootsma & Hecky (1993), a loading into Lake Victoria of 1,353 kg $km^{-2}$ $yr^{-1}$ for nitrogen and 70 kg $km^{-2}$ $yr^{-1}$ for phosphorous. Comparative estimates from Bootsma *et al.* (1996) for Lake Malawi report values of 1,243 kg $km^{-2}$ $yr^{-1}$ for nitrogen and 53 kg $km^{-2}$ $yr^{-1}$ for phosphorous. The cause of this increase in atmospheric phosphorous flux into the lake is uncertain. Unlike nitrogen, phosphorous is more closely associated with particulates. Increased atmospheric phosphorous concentrations may be caused by a greater atmospheric particulate load, which could be the result of burning or increased soil erosion by wind due to deforestation and cultivation. In our model the atmospheric deposition should be expressed by the product of the mean concentration of nutrients in rain water for the mean annual precipitation multiplied by the unit area. Unfortunately we do not have a hystorical database of nutrient concentrations in rainwater for our area of study. For this reason, wet and dry atmospheric deposition is estimated using an empirical function for sub-Sahara Africa available from the literature and linking the nutrient input with the mean annual rainfall (Stoorvogel & Smaling, 1990; Smaling *et al.*, 1993; Stoorvogel *et al.*, 1993):

$$Dep_t = 0.053\Omega_t^{0.5} \qquad [7.43]$$

where $Dep_t$ is atmospheric deposition of phosphorous (kg $ha^{-1}$ $yr^{-1}$) and $\Omega_t$ is mean annual precipitation (mm $yr^{-1}$). Mean annual precipitation for the yala watershed has been estimated at 1,536 mm $yr^{-1}$ using GIS analysis (§ 7.4).

During a fallow period a gradual build-up of nutrients takes place. Brubacher *et al.* (1989) report values of 8.7, 11.0 and 18.3 kg $ha^{-1}$ of phosphorus build-up for fallow aged 1, 2 and 3 years respectively. Nye & Greenland (1960) published a value of 54 kg $ha^{-1}$ for phosphorus build-up with 5-year secondary forest fallow vegetation in central Africa. Since our model assume 1-year fallow rotation we would consider a value $\chi = 8.7$ kg P $ha^{-1}$ $yr^{-1}$ under fallow-pasture and $\chi = 11$ kg P $ha^{-1}$ $yr^{-1}$ under secondary forest fallow vegetation.

The Yala basin lies within three different agricultural zones: the Western Lowlands (Siaya and Kisumu districts), Western Transitional (Kakamega and Bungoma districts), and High Potential Maiz Zone (Kakamega, Bungoma, Bomet, Nakuru, Narok, Trans Nzoia, and Uasin Gishu districts). In these three regions, maize is the most important cultivation. The percent share of maize in total crop area is 56.4%, 39.2% and 54.2% in the Western Lowlands, Western Transitional, and High Potential Maiz Zone respectively (Owuor, 1998). It is therefore reasonable to assume maize as the representative crop within the basin. Stoorvogel & Smaling (1990) provide a wide range of data on nutrients content in harvested crops for sub-Sahara Africa. Assuming all the crop production area as maize cultivation, we would adopt a crop nutrient content of $\eta$ = 9.4 kg P per each tonne of harvested product (Stoorvogel & Smaling, 1990).

Given the selected parameters, the nutrient runoff $g$ (t yr$^{-1}$) for the Yala watershed can be expressed in terms of $0 \leq \alpha \leq 1$, $0 \leq \beta \leq 1$, and soil phosphorus content $Q$ in kg ha$^{-1}$:

$$g_t = 10^{-3} * \alpha_t * L_{TOT} * 0.42 * A_t(\beta_t) * 1.5 * 0.000379 * Q_t(Q_{t-1}, P_{t-1}, \beta_{t-1}) \qquad [7.44]$$

which, given equation [7.42] and the estimated (§ 7.4) total watershed area $L_{TOT}$= 309,027 ha, is equivalent to:

$$g_t = 0.07379\alpha_t(139.7539 - 137.7574\beta_t) * Q_t(Q_{t-1}, P_{t-1}, \beta_{t-1}) \qquad [7.45]$$

Soil phosphorus content is subject to soil nutrient dynamics according to expression [7.9], which becomes:

$$Q_t = Q_{t-1} + P_{t-1}(1 - \beta_{t-1}) + 2.077 + 11\beta_{t-1} - 9.4y_{At-1}(Q_{t-1}, P_{t-1}, \beta_{t-1}) - .000239Q_{t-1}A_{t-1}(\beta_{t-1}) [7.46]$$

Soil nutrient dynamics are embedding a production function for the most representative crop, which in the Yala basin is maize. The lack of consistent datasets on input fertilizer and crop productivity in the region does not allow any econometric estimation of an ad-hoc agricultural production function for our model. Among the few published yield response functions for the Lake Victoria region, a maize production model can be adapted by Mugunieri *et al.* (1997). Their model estimates the yield response function to nitrogen and phosphorous applications at three specific experimental field stations in the Kisii District of Western Kenya. Their equation includes three site specific dummy variables accounting for soil characteristics. By taking equal to one the dummy of the site with similar soil nutrient conditions to our area of study, we obtain:

$$Y = 924.6 + 31.87P + 11.35N - 0.218P^2 \qquad [7.47]$$

where the maize yield ($Y$) is express in kg ha$^{-1}$; N is applied nitrogen (kg ha$^{-1}$); and P is phosphate application (kg ha$^{-1}$).

Fertilizer use varies substantially across agro-regional zones, by crop, whether and by the use of hybrid seeds (Wanzala *et al.*, 2001). The main types of fertilizers consumed in Kenya are compound fertilizers which provide both nitrogen and phosphate. Planting fertilizers for grain (DAP, NPK) comprise the majority of the fertilizers consumed in Kenya, while straight nitrogenous fertilizers, such as CAN and Urea, are used for top-dressing. DAP is used on maize, MAP on wheat, NPK 25:5:5 is used on tea, NPK 17:17:17 and MOP (Muriate of Potash) on coffee, and speciality fertilizers are used on horticultural crops, particularly in the flower industry (Wanzala *et al.*, 2001). Maize receives more fertilizer than any other crop in Kenya, the majority of this is DAP, accounting for the 40% of the fertilizer consumed in Kenya in 1998 (Wanzala *et al.*, 2001).

In applying equation [7.47] to our model, it is important to make the distinction between fertilizer use (the amount of fertilizer material applied in kg ha$^{-1}$) and fertilizer nutrient use (the amount of fertilizer nutrients applied in kg ha$^{-1}$). Typically, a bag of fertilizer will contain both nutrients (nitrogen, phosphorus, potassium, sulphur) and fertilizer material, which is just dry non-chemical matter which facilitates the packaging, handling and spreading of fertilizer by hand. The 50 kg bag of DAP fertilizer actually holds 23 kg of nitrogen, 9 kg of phosphorus, and the remaining 18 kg is dry material. The proportion between N and P fertilizer applications is therefore fixed at 2.55. Ideally we would like to consider soil phosphorus content. According to our model, soil nutrient dynamics would influence the amount of fertilizer applications and crop yield. The yield response fuction would therefore require the inclusion of soil nutrient content as one of the independent variables. In equation [7.47] this is accounted for by the constant, which however can not describe any dynamics. For our purpose we could therefore include soil phosphorus ($Q$) espressed in kg ha$^{-1}$ and the fixed N/P ratio in equation [7.47] as follows:

$$Y = 924.6 + 31.87(P + Q) + 28.94P - 0.218P^2 \qquad [7.48]$$

Our assumption is supported by few field experiments within the Yala basin. Onyango (1994), in studying maize yield response to fertilizer application at several site plots in the ditricts within the Yala basin, reported control yields, without any fertilizer input, ranging from 1,100 to 2,021 kg ha$^{-1}$, with a mean of 1,606 ka ha$^{-1}$. The study reports soil phosphorus values ranging from 2.3 to 32.9 mg kg$^{-1}$ with a mean of 11.46 mg kg$^{-1}$

equivalent to 30.25 kg $ha^{-1}$. Assuming $Q$ = 30.25 and $P$ = 0, equation [7.48] would estimate 1,888 kg $ha^{-1}$ which better approximates the mean maize control yield than without embedding the soil phosphorus variable in the equation. In our model crop productivity ($y_A$) would be a function of both the production function and $\beta$:

$$y_{At} = (1 - \beta_t) * \left(924.6 + 60.806P_t + 31.866Q_t - 0.218P_t^2\right) * 10^{-3} \qquad [7.49]$$

where $y_A$ is maize crop yield (t $ha^{-1}$) and both $P$ and $Q$ are expressed in kg $ha^{-1}$.

Combining equations [7.42], [7.45], [7.46] and [7.49] with expression [7.35], and accounting in the term $D_{TP}$ of equation [7.35] for the sum of athmospheric deposition (1,460 t P $yr^{-1}$) and superficial loading (1,823 t P $yr^{-1}$) from the rest of the Kenya watershed of Lake Victoria (COWI, 2002), we can estimate the maximum sustainable yield, profit maximizing and open access yield solutions as function of the Yala wetland extent ($\alpha$), percent of tree cover in the Yala basin ($\beta$), and amount of phosphorus fertilizer applications ($P$). The variations in the maximum sustainable yield values are shown in Table 7.3, considering fixed value for $\alpha$ ($\alpha$ = 0.950813 at a wetland area of 15,200 ha), $\beta$ and $P$, and assuming the mean soil phosphorus concentration in the watershed, $Q_{t-1}$ = 30.25 kg $ha^{-1}$, as initial value in the soil stock dynamics.

**Table 7.3:** Fishery maximum sustainable yield (t $yr^{-1}$) under different scenarios of fertilizer application (kg P $ha^{-1}$) and tree cover percent (β) in the Yala basin.

| *P* / *β* | *100* | *90* | *80* | *70* | *60* | *50* | *40* | *30* | *20* | *10* | *0* |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *0* | 185,057 | 185,988 | 186,832 | 187,594 | 188,281 | 188,898 | 189,451 | 189,944 | 190,383 | 190,771 | 191,112 |
| *0.1* | 186,256 | 186,969 | 187,619 | 188,210 | 188,746 | 189,232 | 189,670 | 190,064 | 190,417 | 190,732 | 191,012 |
| *0.2* | 187,314 | 187,845 | 188,334 | 188,781 | 189,190 | 189,562 | 189,901 | 190,207 | 190,484 | 190,732 | 190,954 |
| *0.3* | 188,241 | 188,626 | 188,982 | 189,310 | 189,612 | 189,888 | 190,141 | 190,372 | 190,582 | 190,771 | 190,941 |
| *0.4* | 189,051 | 189,319 | 189,568 | 189,799 | 190,013 | 190,210 | 190,392 | 190,558 | 190,710 | 190,848 | 190,973 |
| *0.5* | 189,755 | 189,932 | 190,097 | 190,251 | 190,394 | 190,527 | 190,650 | 190,764 | 190,867 | 190,962 | 191,048 |
| *0.6* | 190,364 | 190,472 | 190,573 | 190,668 | 190,756 | 190,839 | 190,916 | 190,987 | 191,052 | 191,112 | 191,167 |
| *0.7* | 190,887 | 190,945 | 191,000 | 191,051 | 191,100 | 191,145 | 191,187 | 191,226 | 191,263 | 191,296 | 191,326 |
| *0.8* | 191,332 | 191,357 | 191,381 | 191,404 | 191,425 | 191,445 | 191,463 | 191,480 | 191,496 | 191,511 | 191,525 |
| *0.9* | 191,709 | 191,715 | 191,721 | 191,727 | 191,732 | 191,737 | 191,742 | 191,747 | 191,751 | 191,755 | 191,758 |
| *1* | — | — | — | — | — | — | — | — | — | — | 192,023 |

Given the present eutrophic conditions of the Kenyan waters of Lake Victoria, the predicted $Y_{MSY}$ reaches its maximum at 192,023 t $\text{yr}^{-1}$, when the phosphorus runoff from the Yala basin is minimum, in other words at 100% of tree cover in the basin. At this point soil erosion is only 2 t ha $\text{yr}^{-1}$, while the phosphorus loading into the wetland is almost irrelevant with 6.1 t $\text{yr}^{-1}$, implying there is no marginal contribution to further eutrophication from the Yala watershed. Following the nutrient loading increment, the predicted $Y_{MSY}$ deacreses at progressively higher level of phosphorus fertilizer applications and/or lower percent in tree cover. With 100 kg $\text{ha}^{-1}$ $\text{yr}^{-1}$ of phosphorus fertilizer application and all the land under intensive crop exploitation ($\beta = 0$), soil erosion would be extremely high at 140 t ha $\text{yr}^{-1}$ and the associated nutrient inflow to the wetland at 754 t P $\text{yr}^{-1}$, which leads to a 1.21 mg $\text{m}^{-3}$ increase in chlorophyll-a concentration in the Kenyan waters of Lake Victoria from the lowest extreme of 18.70 mg $\text{m}^{-3}$ at $P = 0$ and $\beta = 1$.

**Table 7.4:** Soil phosphorus balance (kg $\text{ha}^{-1}$) under different scenarios of fertilizer application (kg P $\text{ha}^{-1}$) and tree cover percent (β) in the Yala basin.

| $P$<br>$\beta$ | *100* | *90* | *80* | *70* | *60* | *50* | *40* | *30* | *20* | *10* | *0* |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *0* | 46.65 | 38.47 | 30.70 | 23.35 | 16.40 | 9.86 | 3.73 | -1.99 | -7.30 | -12.19 | -16.68 |
| *0.1* | 43.29 | 35.93 | 28.94 | 22.32 | 16.06 | 10.18 | 4.66 | -0.48 | -5.26 | -9.67 | -13.71 |
| *0.2* | 39.93 | 33.39 | 27.18 | 21.29 | 15.73 | 10.50 | 5.60 | 1.02 | -3.22 | -7.14 | -10.73 |
| *0.3* | 36.57 | 30.85 | 25.41 | 20.26 | 15.40 | 10.82 | 6.53 | 2.53 | -1.19 | -4.62 | -7.76 |
| *0.4* | 33.21 | 28.31 | 23.65 | 19.23 | 15.06 | 11.14 | 7.46 | 4.03 | 0.85 | -2.09 | -4.79 |
| *0.5* | 29.86 | 25.77 | 21.88 | 18.20 | 14.73 | 11.46 | 8.40 | 5.54 | 2.88 | 0.43 | -1.81 |
| *0.6* | 26.50 | 23.23 | 20.12 | 17.18 | 14.40 | 11.78 | 9.33 | 7.04 | 4.92 | 2.96 | 1.16 |
| *0.7* | 23.14 | 20.68 | 18.35 | 16.15 | 14.06 | 12.10 | 10.26 | 8.55 | 6.95 | 5.48 | 4.14 |
| *0.8* | 19.78 | 18.14 | 16.59 | 15.12 | 13.73 | 12.42 | 11.20 | 10.05 | 8.99 | 8.01 | 7.11 |
| *0.9* | 16.42 | 15.60 | 14.83 | 14.09 | 13.40 | 12.74 | 12.13 | 11.56 | 11.03 | 10.54 | 10.09 |
| *1* | — | — | — | — | — | — | — | — | — | — | 13.06 |

At different combinations of $P$ and $\beta$ it is also possible to estimate the soil phosphorus balance for our model (Table 7.4). At low fertilizer rates and high percent of land under crop production (i.e. low $\beta$), soil phoshorus balance tends to be negative. These negative values are also reflected, at a certain extent, by a slight reduction in nutrient loading, as it can be seen from the positive marginal impact on fishery yields due to deacresing values of $\beta$ when $P < 20$ and $\beta < 0.3$ (Table 7.3). At this point, most of the soil phosphorus is depleted by the phosphorus content in the harvested crop, reducing the phosphorus stock in the eroded soil. It is interesting to note that at phosphorus applications up to 50 kg ha$^{-1}$ a decrease in tree cover percent ($\beta$) negatively affects the phosphorus balance in the soil. At application rates of 60 kg ha$^{-1}$ and above, lowering $\beta$ produces a positive effect on the soil phosphorus balance. In other words, at low phosphates applications, nutrient build-up from tree cover has a higher weight, as nutrient input, than mineral fertilizers themselves. On the other hand, at high fertilizer rates the opposite is true and decreasing the percent of land under crop production would have a negative effect on soil nutrient balance.

There are no clear estimates of the actual phosphorus fertilizer consumption applicable to our study. However some regional data provide a general indication. Wanzala *et al.* (2001) report data from a survey of 612 households in the Kenya western provinces during 1996/98, including, among others, the Western Lowlands, Western Transitional, and High Potential Maiz Zone. They reported that more than 70% of the households use mineral fertilizers, of which 48% is DAP, whilst the remaining percent was CAN (21%), NPK (16%), Urea (5%), and others (10%). Looking at the disaggregated subregional data for year 1997/98, in the Western Lowlands region only 13% of households applied mineral fertilizers, 79% in the Western Transitional, and 88% in the High Potential Maiz Zone. The mean application rates were 34 kg ha$^{-1}$, 41 kg ha$^{-1}$ and 83 kg ha$^{-1}$ respectively, which correspond to 4 kg ha$^{-1}$, 32 kg ha$^{-1}$, and 73 kg ha$^{-1}$ considering all the agricultural land for the three zones. This translates into 2 kg ha$^{-1}$, 15 kg ha$^{-1}$, and 35 kg ha$^{-1}$ of DAP consumption respectively. Given that the phosphate content in DAP is 18%, the mean phosphorus application would be 0.36 kg P ha$^{-1}$, 2.7 kg P ha$^{-1}$, and 6.3 kg P ha$^{-1}$ in Western Lowlands, Western Transitional, and High Potential Maiz Zone respectively. Assuming an equal weight for the three zones within the Yala basin, we would assume a mean phosphate application rate of around 3 kg P ha$^{-1}$ for the entire watershed. This is consistent with estimates of 4 kg P ha yr$^{-1}$ for the Kakamega district (Van den Bosch *et al.*, 1998).

At a mean phosphorus application rate of 3 kg ha$^{-1}$ and given the estimated (§ 7.4) percent of tree cover ($\beta = 0.36$) for the Yala basin, our model predicts 158 t yr$^{-1}$ of phosphorus loading into the Yala wetland and the $Y_{MSY}$ of 190,914 t yr$^{-1}$. The predicted phosphorous loading is higher than the 102 t yr$^{-1}$ reported at the Yala river mouth (COWI, 2002). Our model, however, considers the whole Yala wetland watershed, of which river

Yala is the most important, but not the only river of the system. Indeed, in the lower part of the basin, whose soil phosphorus content is relatively higher, several other short streams drain directly into the wetlands. It is therefore reasonable to predict a higher nutrient loading than the official estimates for the river Yala alone.

Predicted soil phosphorus balance is negative at -5.14 kg ha$^{-1}$ yr$^{-1}$, which is close to the observed soil phosphorus balance of -4 kg ha$^{-1}$ yr$^{-1}$ for Kakamega district (Smaling *et al.*, 1997). This is an important evidence that the agricultural sector in the Yala basin, as in most of the Kenya, cannot sustain itself in the long run without increasing fertilizer application (mineral or organic or both) or changing land use and soil management practices. Given the current tree cover percent, soil phosphorus balance would be in equilibrium at fertilizer applications of 20 kg P ha$^{-1}$ yr$^{-1}$.

The predicted potential crop yield, with the application of 3 kg P ha$^{-1}$ yr$^{-1}$, would be 2.07 t ha$^{-1}$ yr$^{-1}$, which at $\beta = 0.36$ is 1.32 t ha$^{-1}$ yr$^{-1}$, since we assume only 64% of each hectare of agricultural land is under crop production while 36% is tree vegetation. This last value is in line with observed maize yields, without fertilizer applications, ranging from 1.1 to 2 t ha$^{-1}$ yr$^{-1}$ (Onyango, 1994). At 20 kg P ha$^{-1}$ maize yield would increase to 1.9 t ha$^{-1}$ yr$^{-1}$.

These considerations underline the importance of the nutrient buffering function exerted by the Yala wetland. There is a need to ensure long term sustainability of the agricultural sector by reaching the equilibrium of the soil nutrients balance. In the case of phosphorus this would require an increase of 17 kg ha$^{-1}$ yr$^{-1}$ from the current 3 kg ha$^{-1}$ yr$^{-1}$ of phosphate fertilizer. The impact on the fishery sector, in terms of $Y_{MSY}$ lost, would be only 259 t yr$^{-1}$ or US$ 131,714. The wetland would be able to absorb most of the external impact to the fishery, therefore supporting the much needed agricultural development in the basin.

The scenario, however, would be different at the Yala wetland size reduced to 6,000 ha from the current 15,200 ha (Table 7.5 and Table 7.6). At the present level of phosphorus fertilizer application and percent of tree cover, the wetland reclamation project would lead to a $Y_{MSY}$ reduction of 2,475 t yr$^{-1}$, equivalent to a loss of US$ 1.26 million to the fishery sector. At 20 kg P ha$^{-1}$ yr$^{-1}$ fertilizer application the $Y_{MSY}$ reduction would be 3,130 t yr$^{-1}$ and the economic loss would increase to US$ 1.59 million.

**Table 7.5:** Fishery yield loss (t yr$^{-1}$) under MSY regime due to reclamation of Area II of the Yala wetland under different scenarios of fertilizer application (kg P ha$^{-1}$) and tree cover percent (β) in the Yala basin.

| *P*<br>*β* | *100* | *90* | *80* | *70* | *60* | *50* | *40* | *30* | *20* | *10* | *0* |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *0* | -21,982 | -18,449 | -15,343 | -12,631 | -10,282 | -8,262 | -6,537 | -5,078 | -3,852 | -2,833 | -1,993 |
| *0.1* | -17,451 | -14,848 | -12,547 | -10,522 | -8,751 | -7,211 | -5,879 | -4,735 | -3,759 | -2,932 | -2,235 |
| *0.2* | -13,618 | -11,762 | -10,107 | -8,639 | -7,343 | -6,202 | -5,204 | -4,334 | -3,581 | -2,931 | -2,375 |
| *0.3* | -10,417 | -9,142 | -7,996 | -6,970 | -6,054 | -5,239 | -4,518 | -3,881 | -3,322 | -2,832 | -2,407 |
| *0.4* | -7,776 | -6,941 | -6,184 | -5,499 | -4,881 | -4,326 | -3,828 | -3,384 | -2,988 | -2,638 | -2,330 |
| *0.5* | -5,627 | -5,113 | -4,643 | -4,213 | -3,821 | -3,465 | -3,143 | -2,852 | -2,590 | -2,355 | -2,146 |
| *0.6* | -3,903 | -3,613 | -3,345 | -3,098 | -2,870 | -2,661 | -2,470 | -2,295 | -2,137 | -1,993 | -1,865 |
| *0.7* | -2,541 | -2,398 | -2,264 | -2,139 | -2,023 | 1,916 | -1,817 | -1,726 | -1,642 | -1,566 | -1,497 |
| *0.8* | -1,483 | -1,427 | -1,374 | -1,324 | -1,278 | -1,234 | -1,193 | -1,156 | -1,121 | -1,090 | -1,061 |
| *0.9* | -676 | -663 | -651 | -640 | -629 | -619 | -609 | -600 | -592 | -584 | -577 |
| *1* | — | — | — | — | — | — | — | — | — | — | -73 |

**Table 7.6:** Potential economic loss (million USD) to the Kenyan fishery under MSY regime due to reclamation of Area II of the Yala wetland under different scenarios of fertilizer application (kg P ha$^{-1}$) and tree cover percent (β) in the Yala basin.

| *P*<br>*β* | *100* | *90* | *80* | *70* | *60* | *50* | *40* | *30* | *20* | *10* | *0* |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *0* | -11.18 | -9.38 | -7.80 | -6.42 | -5.23 | -4.20 | -3.32 | -2.58 | -1.96 | -1.44 | -1.01 |
| *0.1* | -8.87 | -7.55 | -6.38 | -5.35 | -4.45 | -3.67 | -2.99 | -2.41 | -1.91 | -1.49 | -1.14 |
| *0.2* | -6.93 | -5.98 | -5.14 | -4.39 | -3.74 | -3.15 | -2.65 | -2.20 | -1.82 | -1.49 | -1.21 |
| *0.3* | -5.30 | -4.65 | -4.07 | -3.54 | -3.08 | -2.66 | -2.30 | -1.97 | -1.69 | -1.44 | -1.22 |
| *0.4* | -3.95 | -3.53 | -3.14 | -2.80 | -2.48 | -2.20 | -1.95 | -1.72 | -1.52 | -1.34 | -1.18 |
| *0.5* | -2.86 | -2.60 | -2.36 | -2.14 | -1.94 | -1.76 | -1.60 | -1.45 | -1.32 | -1.20 | -1.09 |
| *0.6* | -1.98 | -1.84 | -1.70 | -1.57 | -1.46 | -1.35 | -1.26 | -1.17 | -1.09 | -1.01 | -0.95 |
| *0.7* | -1.29 | -1.22 | -1.15 | -1.09 | -1.03 | -0.97 | -0.92 | -0.88 | -0.83 | -0.80 | -0.76 |
| *0.8* | -0.75 | -0.73 | -0.70 | -0.67 | -0.65 | -0.63 | -0.61 | -0.59 | -0.57 | -0.55 | -0.54 |
| *0.9* | -0.34 | -0.34 | -0.33 | -0.32 | -0.32 | -0.31 | -0.31 | -0.30 | -0.30 | -0.30 | -0.29 |
| *1* | — | — | — | — | — | — | — | — | — | — | -0.04 |

## 7.6. Policy implications

In most cases, the potential loss of wetland services is not factored into the decisions of those whose behaviour affects the wetland. Since the effects of agricultural activities are not reflected in market prices they are ignored by those responsible. They are external to the market. That is, any cost imposed or benefit conferred on other members of society is not taken into account in decisions about, for example, wetlands reclamation to agricultural expansion. Estimation of the opportunity cost of such externalities depends on our capacity to calculate the changes in the value of output they induce. If the policy makers makes the assumption that the agricultural activities in the Yala basin have no effect on the downstream fishery sector, they will encourage farmers to expand agricultural area by reclaiming the wetlands. They will also promote agriculture intensification by both decreasing fallow area and increasing fertilizer applications up to the point where the direct marginal net benefits are equal to zero. On the other hand, if we can identify the nature and significance of the interdependence between activities that either depend on or affect the characteristics of the wetlands, we can also identify ways in which policy makers can intervene to assure the sustainable use of wetland resources.

Policy interventions may rely on a series of measure which include, among others, taxes on polluting inputs (in our case fertilizers), taxes on emissions (i.e. nutrient loading), product taxes (i.e. maize production) and subsidies on pollution reduction (i.e. subsidies to promote the expansion of area under fallow). If policy makers decide for a taxation measure, the tax rate should be set to a level that achieves the desired reduction in the externality, satisfying the necessary conditions for the minimisation of the cost to society (Baumol & Oates, 1988). This implies the tax rate should be equal to the discounted shadow cost of the externality which is espressed by the right hand terms in equations [7.28], [7.29] and [7.30] respectively.

When the optimum for the agricultural sector coincides with the social optimum the tax rate is equal to the shadow price of externality abatment. In our model, the abatment cost for the agricultural sector describes the cost of reducing its emission, which in this case is nutrient loading. Since in our assumptions we are not allowing for the option of improving soil management practice, the cost is represented by forgone agricultural production due to input reduction (i.e. fertilizer and land under crop). Internalizing the nutrient loading externality into sectoral decisons through the imposition of a tax implies that, at a given wetland extent ($\alpha$), the agricultural sector would chose an optimal allocation of $P$ and $\beta$ such that the marginal taxation equals the marginal abatment cost.

In our case we should estimate the marginal impact on water quality and fishery yield due to fertilizer input and land use decisions by the upstream agricultural sector. Assuming the MSY is the management regime in the fishery sector, the policy maker should then set a land use tax which equals the shadow price of nutrient loading such that:

$$T = \frac{p\left[Y_{msy}(\beta = 1, P = 0, \alpha) - Y_{msy}(\beta, P, \alpha)\right]}{\alpha L_{TOT}} \qquad [7.50]$$

where $p$ is the price of fish (508.55 US\$ $t^{-1}$). In other words, the per hectare nutrient loading tax in the basin is equal to the market value of the forgone gross marginal profit in the fishery sector due to the change from the initial scenario of no agricultural activity in the watershed, which implies all land is forest ($\beta$ = 1) and implicitly no fertilizer application ($P$ = 0), divided by the agricultural land in term of hectares. Therefore, given a level of wetland extent (1-$\alpha$), we would have different tax rates for a set of combinations of $\beta$ and $P$ (Table 7.7).

**Table 7.7:** Nutrient loading externality tax (US\$ $ha^{-1}$ $yr^{-1}$) under different scenarios of fertilizer application (kg P $ha^{-1}$ $yr^{-1}$) and tree cover percent (β) in the Yala basin.

| ***P*** ***β*** | ***100*** | ***90*** | ***80*** | ***70*** | ***60*** | ***50*** | ***40*** | ***30*** | ***20*** | ***10*** | ***0*** |
|---|---|---|---|---|---|---|---|---|---|---|---|
| ***0*** | 12.06 | 10.44 | 8.98 | 7.66 | 6.48 | 5.41 | 4.45 | 3.60 | 2.84 | 2.17 | 1.58 |
| ***0.1*** | 9.98 | 8.75 | 7.62 | 6.60 | 5.67 | 4.83 | 4.07 | 3.39 | 2.78 | 2.23 | 1.75 |
| ***0.2*** | 8.15 | 7.23 | 6.38 | 5.61 | 4.90 | 4.26 | 3.67 | 3.14 | 2.66 | 2.23 | 1.85 |
| ***0.3*** | 6.54 | 5.88 | 5.26 | 4.70 | 4.17 | 3.69 | 3.26 | 2.86 | 2.49 | 2.17 | 1.87 |
| ***0.4*** | 5.14 | 4.68 | 4.25 | 3.85 | 3.48 | 3.14 | 2.82 | 2.53 | 2.27 | 2.03 | 1.82 |
| ***0.5*** | 3.92 | 3.62 | 3.33 | 3.07 | 2.82 | 2.59 | 2.38 | 2.18 | 2.00 | 1.84 | 1.69 |
| ***0.6*** | 2.87 | 2.68 | 2.51 | 2.35 | 2.19 | 2.05 | 1.92 | 1.79 | 1.68 | 1.58 | 1.48 |
| ***0.7*** | 1.97 | 1.87 | 1.77 | 1.68 | 1.60 | 1.52 | 1.45 | 1.38 | 1.32 | 1.26 | 1.21 |
| ***0.8*** | 1.19 | 1.51 | 1.11 | 1.07 | 1.03 | 1.00 | 0.97 | 0.94 | 0.91 | 0.89 | 0.86 |
| ***0.9*** | 0.54 | 0.53 | 0.52 | 0.51 | 0.50 | 0.49 | 0.49 | 0.48 | 0.47 | 0.46 | 0.46 |
| ***1*** | — | — | — | — | — | — | — | — | — | — | 0 |

The wetland reclamation project would increase the impact of the loading externalities which should be reflected by the level of taxation on agricultural land use. In this case, the expression which identifies the level of the tax rate would internailze the changes in wetland area:

$$\Delta T = \frac{p\left[Y_{msy}(\beta, P, \alpha = 0.950813) - Y_{msy}(\beta, P, \alpha = 0.980584)\right]}{0.980584 L_{TOT}} \qquad [7.51]$$

where $\alpha$ = 0.950813 and $\alpha$ = 0.980584 correspond to a wetland area of 15,200 ha and 6,000 ha respectively, while the total land in the basin is $L_{TOT}$= 309,027 ha. The set of tax rates in Table 7.8 represents the taxation increment due to wetland reclamation for a given set of combinations of $\beta$ and $P$.

**Table 7.8**: Wetland reclamation tax (US\$ ha$^{-1}$ yr$^{-1}$) under different scenarios of fertilizer application (kg P ha$^{-1}$ yr$^{-1}$) and tree cover percent (β) in the Yala basin.

| *P*<br>*β* | *100* | *90* | *80* | *70* | *60* | *50* | *40* | *30* | *20* | *10* | *0* |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *0* | 36.89 | 30.96 | 25.75 | 21.20 | 17.26 | 13.86 | 10.97 | 8.52 | 6.47 | 4.75 | 3.34 |
| *0.1* | 29.29 | 24.92 | 21.06 | 17.66 | 14.69 | 12.10 | 9.87 | 7.95 | 6.31 | 4.92 | 3.75 |
| *0.2* | 22.85 | 19.74 | 16.96 | 14.50 | 12.32 | 10.41 | 8.73 | 7.24 | 6.01 | 4.92 | 3.99 |
| *0.3* | 17.48 | 15.34 | 13.42 | 11.70 | 10.16 | 8.79 | 7.58 | 6.51 | 5.57 | 4.75 | 4.04 |
| *0.4* | 13.05 | 11.65 | 10.38 | 9.23 | 8.19 | 7.26 | 6.42 | 5.68 | 5.01 | 4.43 | 3.91 |
| *0.5* | 9.44 | 8.58 | 7.79 | 7.07 | 6.41 | 5.82 | 5.27 | 4.79 | 4.35 | 3.95 | 3.60 |
| *0.6* | 6.55 | 6.06 | 5.61 | 5.20 | 4.82 | 4.47 | 4.14 | 3.85 | 3.59 | 3.34 | 3.13 |
| *0.7* | 4.26 | 4.02 | 3.80 | 3.59 | 3.40 | 3.21 | 3.05 | 2.90 | 2.76 | 2.63 | 2.51 |
| *0.8* | 2.49 | 2.39 | 2.31 | 2.22 | 2.14 | 2.07 | 2.00 | 1.94 | 1.88 | 1.83 | 1.78 |
| *0.9* | 1.13 | 1.11 | 1.09 | 1.07 | 1.05 | 1.04 | 1.02 | 1.01 | 0.99 | 0.98 | 0.97 |
| *1* | — | — | — | — | — | — | — | — | — | — | 0.12 |

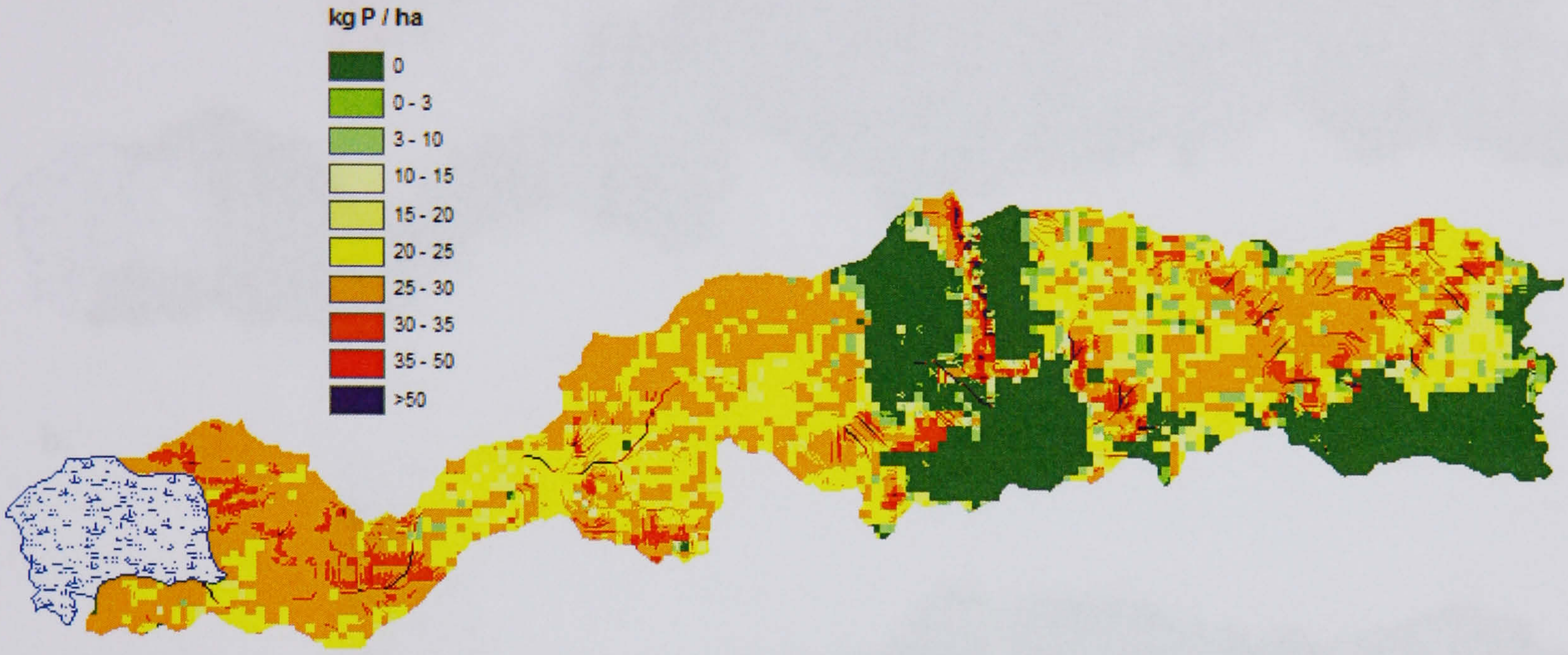
Policy makers can decide on how the taxation measure is going to be imposed to the agent producing the externality. The easiest option is applying a mean tax rate to each agricultural household without distinction. This obviously brings equity issues with it. The marginal externality produced varies considerably within the basin. This depends on both environmental characteristics, such as soil quality, rainfall and relief, and household decisions on fertilizer application and fallow extent. Applying a unique rate of taxation would lead to free riding behaviour from those households producing the highest amount of nutrient loading, since they would shift most of the cost of their externality to the other tax payers whose nutrient loading is minimal. If the policy maker's intention is modifying the agricultural agent's behaviour through taxation, especially after wetland reclamation, this approach would not reach the objective since the per-hectare tax would be too low to produce any appreciable impact to the average household in the basin. A partial solution could be internalizing the tax in the market price of fertilizers. This, however, may be difficult to implement at the basin level and does not eliminate efficiency problems.

In the case of fertilizers, as for many potentially polluting substances, ambient concentrations are dependent not just on the total amount of runoff/emission, but also on their spatial location (Hanley *et al.*, 1997). Therefore, for non-uniformly mixed pollutants a single tax rate will no longer be efficient, since the tax rate should vary across sources according to their marginal impact (Baumol & Oates, 1988). In our model we should therefore estimate the marginal impact on fishery yield sector due to allocation decisions by the $i^{th}$ agricultural household. Thus a unique tax rate exists for each hectare in the Yala basin:

$$t^i = p\left[Y_{msy}\left(\beta^i = 1, P^i = 0, \alpha\right) - Y_{msy}\left(\beta^i, P^i, \alpha\right)\right] \qquad [7.52]$$

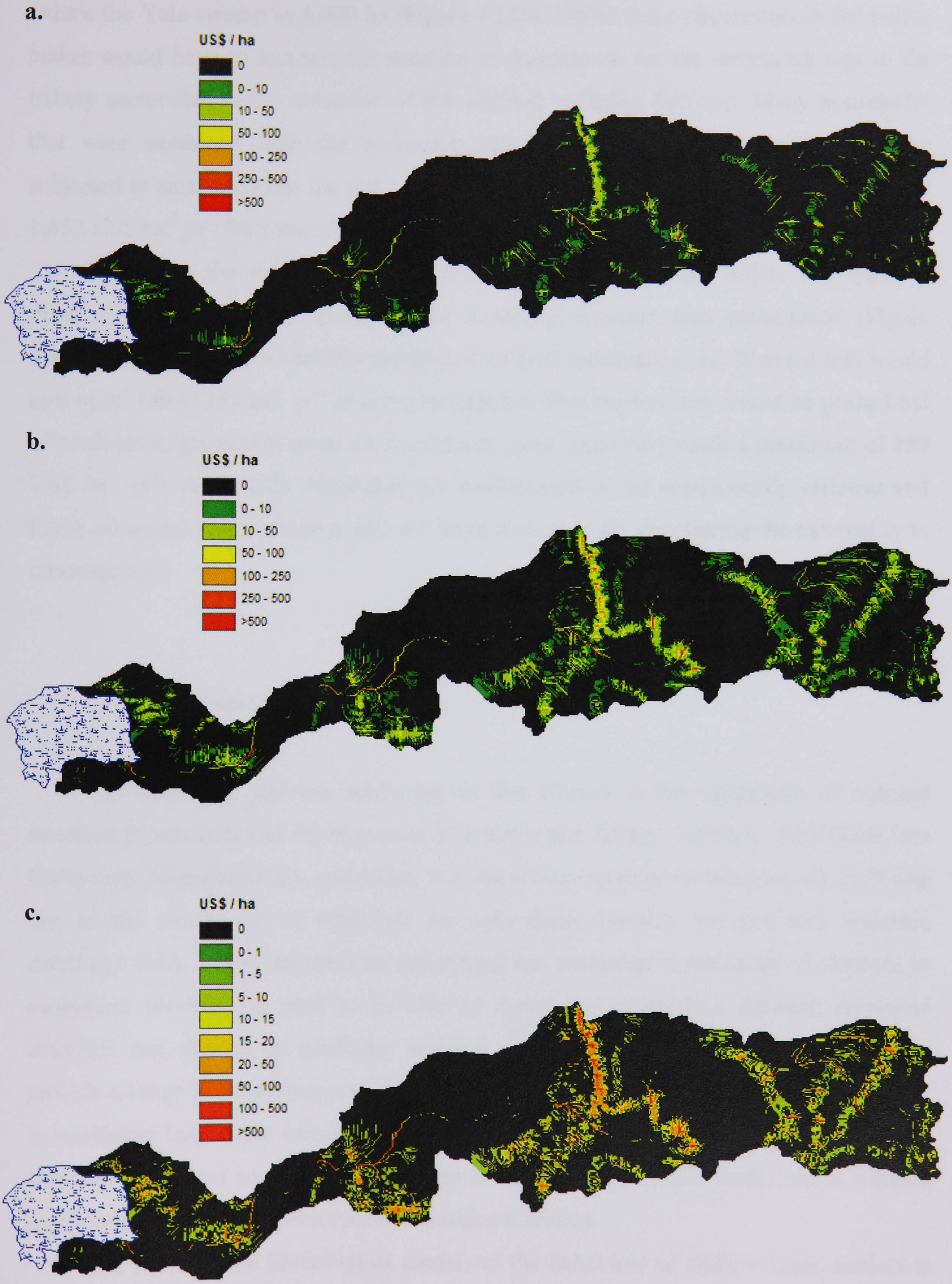
We estimate the tax scenario keeping in mind the minum objective of the agricultural sector. This is reaching the soil phosphorus balance for each $i^{th}$ hectare which guarantees the long term sustainability of crop production in the watershed. The objective requires a specific amount of phosphorus fertilizer application. This varies for each hectare according to the natural conditions influencing soil erosion (site specific rain erosivity, soil erodibility and length/slope factors, and the *SDR*) and to the actual land use ($\beta$ value), which impacts both soil erosion, through the land cover factor (*C*), and soil phoshprus dynamics. The required phosphorus application for each $i^{th}$ hectare is obtained by solving for *P* the soil nutrient balance equation [7.9]. The result is an espression which gives $P^i$ as a function of $\beta^i$ and the net soil erosion or sediment yield $S_{DR}{}^i * A^i$. We estimate the equation in GIS environment through the raster calculator function by using tree cover percent (Figure 7.6a) and sediment yield (Figure 7.7c) maps as input grids, and assuming the initial soil phosphorus concentration of Q = 30.25 kg ha$^{-1}$ for each $i^{th}$ hectare. The resulting map identifies the amount of phosphorus fertilizer (kg ha$^{-1}$ yr$^{-1}$) required at each $i^{th}$ hectare site (Figure 7.10).

**Figure 7.10:** Phosphorus fertilizer required (kg ha$^{-1}$ yr$^{-1}$) for long term soil nutrient balance.



The marginal impact of each $i^{th}$ agricultural household decisions to the maximum sustainable yield of the fishery sector can be estimated by combining equations [7.42], [7.45], [7.46] and [7.49] with expression [7.35] as showed earlier. This time, however, we estimate the espression using the GIS raster calculator with the tree cover percent (Figure 7.6a), sediment yield (Figure 7.7c) and required phosphorus fertilizer (Figure 7.10) maps as input grids. By taking the difference between the obtained grid and the raster espressing the $Y_{msy}$ at $\beta = 1$, and multiplying it by the price of the fish according to equation [7.52], we obtain a map espressing the spatial distribution of the nutrient loading externality tax throughout the basin (Figure 7.11a). The tax would varies from virtually zero for the sites that experience relatively low soil erosion to a maximum of US\$ 588 ha$^{-1}$ yr$^{-1}$ for the households with the highest nutrient loading.

**Figure 7.11:** Spatial distribution of nutrient loading externality tax with wetland area of 15,200 ha (a), with wetland area of 6,000 ha (b), and wetland reclamation tax distribution (c) in US\$ ha$^{-1}$ yr$^{-1}$.



Tax distribution would however change considerably when considering the nutrient runoff externality under the scenario of the wetland reclamation project which would reduce the Yala swamp to 6,000 ha (Figure 7.11b). Under these circumstances the policy maker would have to increase the taxation to compensate for the increasing loss to the fishery sector due to the reduction of the wetland buffering function. Many households that were exempted from the externality tax before wetland reclamation would be subjected to taxation under the new scenario, in which the tax could reach a maximum of 1,652 US\$ $ha^{-1}$ $yr^{-1}$ for some sites.

By taking the difference between the two grids, we can isolate the spatially distributed value of the forgone wetland buffering function after reclamation (Figure 7.11c). This represents a spatially distributed wetland reclamation tax. Wetland loss would cost up to 1,064 US\$ $ha^{-1}$ $yr^{-1}$ to some households. This implies they would be pushed out of production, given that revenues from maize yield could only reach a maximum of 889 US\$ $ha^{-1}$ $yr^{-1}$. A spatially distributed tax would therefore be economically efficient and fulfill all equity issues, since it will not leave any room for transferring the externality to other agents.

## 7.7. Discussion

The modelling problem addressed in this chapter is the integration of nutrient retention in wetlands and the dynamics of a freshwater fishery. Globally, the Millennium Ecosystem Assessment has concluded that ecosystem services of this sort are declining due to the conversion of wildlands for agriculture, forestry, industry and domestic dwellings (MA, 2005). In order to understand the economic significance of changes in ecosystem services we need to be able to model the interactions between economic activities that depend on particular services, and changes in those services. Wetlands provide a range of ecosystem services of which nutrient retention is particularly important in mediating land-water interactions. The integrated model here includes both a fishery component adapted to include the effects of changes in nutrient loads, and a wetland component to explain the absorption of passing nutrients.

One result of the limnological models of the behaviour of shallow lakes subject to nutrient loading, is that the oligotrophic and eutrophic states are both locally stable. As nutrients are added to oligotrophic shallow lakes there comes a point at which they flip into a eutrophic state. Once in a eutrophic state, however, they are subject to hysteresis. That is, they cannot be induced to return to an oligotrophic state by reducing the loading to the level that caused the flip in the first place. Loading has to be reduced far below that level and held there for a long time (Carpenter & Pace, 1997; Carpenter & Cottingham, 1997). This has important implications for policy. If policies affecting nutrient loading, whether through wetland conversion or the application of fertilizers, allow the lake to flip into a eutrophic state, it may not be possible to induce a return within a sensible time frame (Mäler *et al.*, 2003).

The fishery component of the model reported in this study does not have this property. Indeed, given the size and depth of Lake Victoria, the shallow lake literature (Carpenter & Pace, 1997; Carpenter & Cottingham, 1997; Mäler *et al.*, 2003) is not particularly relevant to our model. The effect of changes in nutrient loading will induce changes in fish biomass that depend on the initial level of loading and the initial state of the system. In general, however, a reduction of nutrient loads when the lake is in a eutrophic state will lead to an increase in fish biomass. Similarly, the nutrient retention function of wetlands is reversible. If the wetland area or biomass is increased nutrient retention will also increase. In practice, wetland conversion may be irreversible for practical political reasons, but it is not modelled as such.

The model nevertheless enables us to estimate the value of the nutrient absorption externalities of wetland conversion, and to project the consequences of alternative wetland conversion strategies. Following a watershed approach to fishery policy, however, implies looking at the agricultural sector as well. In a country like Kenya the importance of agricultural production has wide implications that go beyond pure economic considerations. Indeed, food security has been given a lot of emphasis in recent years by the Government of Kenya (Republic of Kenya, 1994). Most of the capacity required to attain food security will be met by increasing maize production, which is the staple food crop (Mose *et al.*, 1997). Given the population growth in the Country, this requires an annual growth rate in production of about 2.5% (Republic of Kenya, 1994). However, declining soil quality on farmlands in Kenya and sub-Saharan Africa (§ 1.3.2.) as a whole gravely undermine prospects for bringing an end to chronic poverty and food insecurity in the region (Conway, 2001; FAO 2001; World Bank, 2003; IFDC 2005, 2006). Sanchez *et al.* (1997) report that between 1960 and 1990, an average of 660 kg $ha^{-1}$ of nitrogen, 75 kg $ha^{-1}$ of phosphorus, and 450 kg $ha^{-1}$ of potassium were lost from 200 million hectares of cultivated land in 37 African countries. Recent estimates are that sub-Saharan Africa lost 4.4 million tonnes of nitrogen, 0.5 million tonnes of phosphorous, and 3 million tonnes of potassium during the last decades, costing the continent more than US$ 4 billion worth of soil nutrients per year (Heisey & Mwangi, 1996; Smaling *et al.*, 1997; Kherallah *et al.*, 2000; IFDC, 2006). These losses swamp additions from fertilizer applications, which equal 0.8, 0.26, and 0.2 million tonnes of N, P, and K, respectively (Sanchez *et al.*, 1997). Reversing these trends through investments in soil fertility maintenance via nutrient amendments based on organic and inorganic fertilizer application is crucial to maintaining or advancing agricultural productivity in the region, and thereby reducing widespread poverty (Place *et al.*, 2003). This problem is of particular importance within the Kenyan basin of Lake Victoria, which produces most of the maize in the Country while still having nutrient applications below the required quantity to attain a positive soil nutrient balance and the associated long term sustainability.

The necessary development of the agricultural sector, however, should not undermine the productivity of other economic sectors, such as fishery, that are also crucial in providing food security and social stability to the region. In this context, the role played by wetlands as buffering system between these two economic sectors is important in reducing downstream externalities and ensure higher benefits to the society. Wetlands do provide a number of other ecosystem services (§ 1.3.3), but in this case nutrient absorption may in fact be the most important. The effects of wetland conversion are also much wider than the effects on the commercial fishery in the lake, although it is arguable too that the effects on the fishery are more important than the effects on other sectors.

# Chapter 8

# CONCLUSIONS

In developing countries the link between food security and sustainable use of natural resources is crucial to economic development and social stability. Nevertheless, market failures, sustained population growth and increasing population densities often lead to overexploitation and unsustainable use of those resources, compromising the basis of future development. In this context, agriculture and fishery are key economic sectors. Both sectors, however, are directly or indirectly threatened by land degradation which has a negative effect on their output production.

In some of the poorest regions of the world, agriculture is confronting trends of decreasing productivity. Soils deficient in the main nutrients, such as nitrogen and phosphorus, have been identified as the major problem affecting crop yield in much of sub-Saharan Africa due to a long-term process of soil fertility depletion (Mokwunye *et al.*, 1996; Sanchez *et al.*, 1997; Smaling *et al.*, 1997). Soil is a form of capital that is an essential input to agricultural production (Ekbom, 1995). However, agricultural land use often results in degradation of natural soil fertility through nutrient leaching and soil erosion. Soil erosion not only degrades the land that loses the soil, on-site productivity losses (§ 2.2), but also causes problems far away from the source, off-site environmental problems (§ 2.3). The soil, chemicals and pathogens it contains, are transported into waterways and onto other lands, inflicting off-site effects through the processes of sedimentation, eutrophication and leaching.

Whilst the problem of decreasing agricultural productivity is limited to a specific region only, capture fisheries are in decline worldwide. There are two main reasons for this. One is the fact that many fisheries are still effectively open access resources, which results in the overexploitation of those resources. The other is the effect of land-based pollution. Many freshwater and marine capture fisheries have been severely affected by sewage, nutrients, synthetic organic compounds, sediments, metals, radionuclides, oil/hydrocarbons and polycyclic aromatic hydrocarbons (UN, 2004). The application of fertilizers in agriculture, fossil fuel burning, land clearance and biomass burning have long been recognised as a major source of the nutrient load in freshwater, coastal and estuarine systems (Oglesby, 1977; Nixon, 1988).

Freshwater lakes represent only a small part of the total water on the earth. Nevertheless, they are essential to the maintenance of terrestrial life and in supporting human consumption and activities. The fact that human effects on lakes are growing worldwide, increases the concern that the many lake services are jeopardized by lake degradation. Eutrophication is probably the best understood type of lake degradation (§ 1.1). High phytoplankton biomass and productivity, as in eutrophic lakes, have a detrimental effect on fish growth and reproduction rates through the depletion of oxygen concentration in superficial waters.

The services lakes provide to humans are impaired not only by direct exploitation of lakes, but also by changes in land use, vegetative cover, and other actions that affect the hydrology of the watershed. The changes in aquatic systems that follow from changes in nutrient loading due to land-based economic activities are said to be an externality of those activities. That is, those whose activities on land damage aquatic systems are not confronted by the cost of that damage.

In this context, wetland reclamation may represent a major expression of environmental externality (§ 3.1). From an economic perspective, the nutrient retention function of wetlands makes then a potentially low cost mean of reducing point and non-point pollution (Gren, 1993, 1995; Gren *et al.*, 1997; Byström, 1998; Byström *et al.*, 2000). Nevertheless, interrelated market and intervention failures, which derive from a fundamental failure of information, or lack of understanding of the multitude of values and services that may be associated with wetlands, result in negative effects on wetlands ecosystems (Turner *et al.*, 2000).

The linkages between land-based activities, the buffering function of wetlands and the downstream fisheries need to be widely understood. Freshwater ecosystems and fisheries must therefore be studied within the context of their watershed and its related human activities (§ 1.2). Early approaches either focused on simple correlations between changes in watersheds and changes in fisheries, or else identified the consequences for fisheries if the linkages were of varying strength (Ruitenbeek, 1989; Hodgson & Dickson 1998). Limnologists have investigated the consequences for freshwater aquatic systems of changes in land use, vegetative cover, and fertilizer regimes within the watershed (Postel & Carpenter, 1997; Carpenter & Pace, 1997; Carpenter & Cottingham, 1997). More recently, economists have considered the theoretical problems posed by the interaction between users of lake and catchment resources (Mäler *et al.*, 2003; Carpenter, *et al.*, 1999a). Nevertheless, there are few attempts to model the interactions in real systems, or to estimate the value of land-water externalities of this kind.

We model land-water interactions in a particular system – the Kenyan catchment of Lake Victoria – with a view to understanding the consequences of the conversion of wetlands for the lake fishery. Nutrient enrichment has a positive effect on fishery productivity in nutrient-limited environments such as oligotrophic or mesotrophic lakes (Stockner & Shortreed, 1988; Melack, 1976a; Liang *et al.*, 1981; Hoyer & Jones, 1983; Downing *et al.*, 1990; Quirós, 1990; Gomes *et al.*, 2002). However, there is also evidence that sustainable harvests of fish populations at upper trophic levels decline if the system becomes highly eutrophic (Beeton, 1969; Lee *et al.*, 1991; Caddy, 1993). Excess nutrients affect fish productivity through changes both in the amount of food available (Bootsma & Hecky, 1993) and in the quality of the habitat (Hammer *et al.*, 1993). Deoxygenated water boosts natural mortality of fish. Sedimentation negatively affects nursery grounds and may damage fish eggs. When combined with high fishing pressure, both effects can have a severe impact on fish stock biomass and fishery yields (Kemp *et al.*, 2001).

In Lake Victoria, fish production in all three riparian states has grown dramatically since the introduction of the Nile perch (*Lates niloticus*) in the early 1960s. In the Kenyan waters of the lake, output grew from around 17,000 tonnes per year in the 1960s to more than 200,000 tonnes in the early 1990s. During the 1980s, *Lates* catches increased exponentially rising in few years from virtually zero to almost 60% of total yield (Okemwa, 1984; CIFA, 1988; Ogari & Asila, 1990; Ogutu-Ohwayo, 1990a; Ssentongo & Welcomme, 1985). However, from 1994 fish landings have been in sharp decline, mostly due to declining catches of Nile perch. By 1998 Nile perch landings were half of those at the beginning of the decade despite increased fishing effort (§ 1.3.1).

Several factors are implicated in this. Overfishing (§ 1.3.1.1) is one factor, but it is not the only one. The long-term consequences of invasive species introduction (§ 1.3.1.3) led to the disruption of energy flow patterns and may have exerted a certain influence on the eutrophication process. A change in regional climate (§ 1.3.1.4) associated with changes in precipitation regimes and increasing temperatures could further explain some of the recent dynamics on the lake ecosystem. Eutrophication (§ 1.3.1.2), caused by nutrient run-off from agricultural land and discharges from urban settlements on the lakeshore is also an important factor, and that stems from two phenomena. One is an increase in nutrient loading due to the application of fertilizers on agricultural lands and the growth of human populations on the lake-shore. A second is a reduction in nutrient absorption by wetlands on the lake margins, largely caused by the conversion of wetlands to other uses (§ 1.3.3). A number of water quality analyses over the last decades show that Lake Victoria has progressively shifted from a mesotrophic to an eutrophic state due to both causes. Increasing chlorophyll-a concentrations have been reported (Ochumba & Kibaara, 1989; Gophen *et al.*, 1995; Kenyanya, 1999; Lung'ayia *et al.*, 2000, 2001) against baseline values provided by Talling (1965, 1966) and Melack (1976b).

Our dynamic simulation of the ecosystem model using the Ecopath approach shows how eutrophication may explain the sudden upsurge of the introduced Nile perch during the 1980s (§ 5.3). We hypothesize that the shift in primary productivity was caused by an exceptional climatic event during the period 1977-1980, when heavy rains increased the lake level and large loads of sediments and nutrients where pumped into the lake. The Nile perch population explosion during the 1980s caused a profound transformation of the fishing industry leading to a dramatic increase in fishing effort. Even though these fisheries were highly productive throughout the 1980s and first half of the 1990s, a sharp decline in both yields and catch per unit effort, especially in the Kenyan sector, from the second half of the 1990s suggests that maximum sustainable yields could have been exceeded and the fishery is moving to a state of stock depletion. Indeed, an equilibrium analysis from the base values of an Ecopath model for year 1989 depicts an overfishing situation in which fishing effort goes beyond its MSY level (§ 5.4).

To quantify the combined effect of eutrophication and fishing effort on the aggregate fish stock and fishery of Lake Victoria we estimate a biomass dynamic model introducing a measure of water quality (chlorophyll-a concentration) into a classic Gordon-Schaefer fishery model (§ 6.2). Our model says that the growth of the fish stock is positively affected by chlorophyll-a concentration at relatively low levels. Such growth reaches its maximum at chlorophyll-a concentrations of 17.11 mg $m^3$, beyond which it starts to decline since a further increase in nutrient load has a negative impact on growth and on harvests. Given the current levels of fishing effort and water quality conditions, a decrease in fishing effort is necessary to avoid the collapse of the fishery. However, due to lack of both policy regulation and elasticity in the fishing industry, it is difficult to forecast a decrease in effort. Increasing levels of eutrophication may further complicate the situation.

The agricultural sector in the region is also confronting trends of decreasing productivity (§ 1.3.2). In the riparian countries of Lake Victoria, the agricultural annual production growth rate has decreased from 4.8% in the 1970s, through 3.3% during 1980-

90, to a negative 1.5% in 1990-94 (World Bank, 1998). Around Lake Victoria soil erosion is particularly high (§ 1.3.2.1), representing an economic problem since farmers ignore all or some of the user costs of the soil and off-site costs of erosion (Odendo & Wasike, 1999).

Technologies which add new nutrients to agro-ecosystems are not sufficient. They need to be supplemented by actions which reduce nutrients runoff from soil erosion. Soil conservation in the form of appropriate land use and management is, therefore, a major issue for sustainable agricultural development in the Lake Victoria basin. Nevertheless, sustained population growth and increasing population densities will further increase land degradation with negative effect for both agriculture and fishery productivity. Governments plans that follows experts' recommendations (Baligar & Bennett, 1986; Vlek, 1990) for promoting higher fertilizer applications may worsen the eutrophication problem if not accompanied by appropriate land use management.

In this study we focus on the role played by wetlands in moderating the effect of land based pollution and hence on eutrophication and fishery yields. Several studies have considered the linkage between wetlands and fisheries in coastal and estuarine systems (Turner, 1982; Swallow, 1994; Barbier & Strand, 1997; Barbier & Sathirathai, 2002). All these studies focus on the ecological function of wetlands in supporting fisheries by serving as both spawning ground and nursery for fry. However, to our knowledge, there is no study of the value of wetlands in terms of their impact on water quality and thus fishery. Although the nutrient retention function of coastal and freshwater wetlands is known to be an important determinant of stock biomass in capture fisheries, the economic significance of this has not been evaluated.

Most of the wetlands around Lake Victoria are being reclaimed and degraded (§ 1.3.3.1). This process is driven by the high growth of the local human population and the consequent increasing demand for settlements, food and grazing lands. Over 70% of the population of the three riparian countries is engaged in agricultural production mostly as small-scale farmers cultivating sugar, tea, coffee, maize, cotton, or being engaged in livestock keeping or horticulture within the lake catchment. Kenya's population has increased rapidly: from 11 million in 1970 to 27 million in 1996, which corresponds to an annual growth rate of around 3.5%. In contrast, arable land area per capita has decreased 35% from 0.23 ha in 1980 to 0.15 ha in 1995, and the population is projected to be 36 million by 2010 (World Bank, 1998).

In modelling the relation between land use change and water quality, we are concerned with the nutrient retention function of wetlands and the link between land use and nutrient loading. First, we use the interaction between land use and the fishery to identify catchment-based indicators, in particular price indicators (§ 7.3.2) and wetland extent (§ 7.3.1), which may anticipate changes in environmental variables driving stock assessment models. We then model the interactions between fertilizer applications, phytoplankton growth, and fish production to identify the functional relationships between fertilizer prices and fish stocks and yields.

As following step we study both nutrient loading and wetland reclamation externalities, focusing on the Yala swamp, one of the largest wetlands in the Kenyan basin of Lake Victoria. The swamp has already been heavily affected by human development and much of the remaining area has been demarcated for 'reclamation' since the mid 1960s. Given the estimated (COWI, 2002) annual loading of phosphorus from the Yala river basin, the impact of the 9,200 ha wetland reclamation project on fishery yields would translate in a direct economic loss of up to US$ 0.7 million per year, depending on the fishery regime (§ 7.5). This implies that the value of the wetland to be converted (its social opportunity cost) is US$ 14.7 million. Based on the simplifying assumptions used to in this study, conversion at less than US$ 1,600 per hectare is unlikely to be efficient.

We extend the analysis considering a land use model which includes variables expressing fertilizer application and land use/cover (§ 7.2). The model is applied to the Yala basin using site specific parameter values (§ 7.5.1). In applying the model, we first define the extent of the catchment and identify its stream network. Then, we focus on the estimation of soil erosion and the associated sediment delivery that are key factors in determining nutrient loading (§ 7.4). Given the current fertilizer application and land use/cover in the basin, our model predicts a higher nutrient loading than the estimates provided by other studies (COWI, 2002). This translates in a much bigger loss, up to US$ 1.26 million per year, to the fishery sector due to wetland reclamation (§ 7.5.1).

This is therefore a typical case where human activities as well as inconsistent or contradictory policies result in negative effects on wetlands ecosystems and their related services. Information, market and intervention failures play an important role in this process. In most cases, the potential loss of wetland services is not factored into the decisions of those whose behaviour affects the wetland. Decision makers may not understand the consequences of their actions. Where the consequences of intervention failures include effects on the general public or third parties, then it matters that policy makers also do not understand the functions of wetlands. On the other hand, if we can identify the nature and significance of the interdependence between activities that either depend on or affect the characteristics of the wetlands, we can also identify ways in which policy makers can intervene to assure the sustainable use of wetland resources.

Policy interventions may rely on a series of measures. If policy makers decide for a taxation measure, the tax rate should be set to a level that achieves the desired reduction in the externality, satisfying the necessary conditions for the minimisation of the cost to society (Baumol & Oates, 1988). Wetland reclamation would increase the impact of the loading externalities. This should be reflected by the level of taxation on agricultural land use.

Since the marginal externality produced varies considerably within the basin, a single tax rate will no longer be efficient and the applied taxation measure should vary across sources according to their marginal impact (Baumol & Oates, 1988). Therefore, a spatially distributed tax would be economically efficient and would reach the desired objective of changing farmers behaviour and maximising the total economic benefit to society (§ 7.6). In the long run, optimal land use accounting for both user costs of the soil and off-site costs of erosion, together with appropriate fishery management and wetlands conservation measures, may offer a solution to the reducing trend in per capita food production for both agriculture and fishery sectors at the same time.

Even though our final model is restricted to a specific sub-basin of Lake Victoria, it represents an important case study. Our methodology could certainly provide fundamental tools for potential model upscaling and application to the whole basin. This however would require a considerable amount of data and financial support and it goes beyond the scope of this study. The international community is already aware of the strategic importance of Lake Victoria in the context of the widespread food security and its geographical location. This can be seen by the increasing number of scientific programmes and international cooperation investments targeting the Lake Victoria basin: the "Lake Victoria Environmental Management Project", 1996-2013 (World Bank-GEF, US$ 77.8m phase I, US$ 165m phase II); the "Lake Victoria Fisheries Research Programme" – phase II, 1995-2000 (EC US$ 11.6m); the "Lake Victoria Basin and Ecosystem Management" project, 1996-2000 (World Bank, US$ 10m); the "Regional Programme for Environmental Management of Lake Victoria", 1993-1996 (World Bank, US$ 20m); the "Socio-economics of the Nile Perch Fishery in Lake Victoria", 1996-2004

(IUCN-NORAD); the Lake Victoria Program of the International Decade for the East Africa Lakes (IDEAL) project; and the USGS Lake Victoria Monitoring, among others.

I hope my work may represent a valuable contribution to this common objective.

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