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Indirect economic indicators in bio-economic fishery models: agricultural price indicators and fish stocks in Lake Victoria

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We consider the potential for using prices as leading indicators of changes in stocks and yields in the freshwater capture fishery in Lake Victoria. Fertilizer run-off from agricultural land is a major cause of nutrient loading, along with soil erosion, atmospheric deposition, and point pollution from industrial and domestic affluent. The interactions between fertilizer applications, water quality, fish stocks, and yields are modelled in order to predict the effect of changes in the price of fertilizers on the fishery. The fishery model includes a measure of Chl *a* concentration (a proxy for phytoplankton density). The consequences of changes in Chl *a* concentration for fish stocks are modelled using Ecopath. We show that fertilizer prices are effective leading indicators of changes in fish biomass and yield.

Keywords: ecological-economic modelling, fisheries, land–water interactions.

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Introduction

The dominant indicators guiding use of freshwater and marine resources worldwide are the prices of those resources, and the goods and services they produce. While government agencies responsible for fishery management have traditionally used catch levels and, more recently, biodiversity or ecosystem health indicators (Charles, 1995), most independent users employ prices to guide their decisions. Because price changes indicate variation in the scarcity of stocks relative to other resources of interest, they contain additional information to traditional single-stock indicators. In theory, if markets were "complete" and there was full information, prices would be sufficient indicators of the scarcity of environmental resources. However, the effects of government policy, the public-good nature of many environmental resources, and the lack of well-defined property rights mean that markets are generally incomplete. Hence, fish prices are generally inadequate measures of the relative abundance of fish stocks (Perrings, 2000). Nevertheless, because market prices drive resource allocation decisions, they constitute a potentially important but little used set of indicators. If we understand the interactions between land-use decisions and exploitation of marine and freshwater stocks, market prices may provide important early warnings of impending changes in those stocks in circumstances where traditional stock assessment does not help.

Traditional stock assessment involves estimating the current status of a stock using one of the classic single-stock models (Ricker, 1954; Schaefer, 1954; Beverton and 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. By including an environmental variable in a biomass dynamic model, for instance, Ikeda and Yokoi (1980) explained the decrease in fish biomass and fish catches as a function of nutrient enrichment in a eutrophic inland sea in Japan. Fréon *et al.* (1993) suggested that in surplus models, such as the Gordon–Schaefer fishery model, environmental factors could 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 an effect on K and r together. Kasulo and Perrings (2002) studied the influence of changes in environmental conditions for Lake Malawi by introducing a biodiversity variable in the form of bio-economic indices that capture both biological and economic factors into an aggregated Gordon–Schaefer fishery model (Gordon, 1954; Schaefer, 1954, 1957).

Environmental conditions in many fisheries are directly affected by other, linked economic activities. Land-use change, for example, has had major effects on marine 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. Coral reefs and their communities have suffered localized damage by pollution from agricultural, industrial, commercial, and domestic housing developments (Grigg, 1994; Szmant, 2002).

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 (cpue), 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 (Hodgson and Dixon, 1988; Ruitenbeek, 1989) 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.

With a surface of 68 800 km^2, 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 area, respectively, and use its resources for fisheries, freshwater, and transportation. Our aim is to investigate the potential for using prices as advance indicators of changes in stocks and yields within the fishery. Among the anthropogenic effects that directly or indirectly impact fish productivity, cultural eutrophication (for a definition, see Jørgensen, 1980) has remained a major concern in lakes (Oglesby, 1977; Nixon, 1988). Nutrient enrichment may have a positive effect on yields in nutrient-limited environments such as oligotrophic or mesotrophic lakes (Stockner and Shortreed, 1988). However, there is growing evidence that sustainable harvests at upper trophic levels may decline as lakes and estuaries become more and more eutrophic (Beeton, 1969; Lee *et al.*, 1991; Caddy, 1993).

In eutrophic environments, excess nutrients affect fish productivity through changes in the amount of available food (Bootsma and Hecky, 1993) and in available habitat (Hammer *et al.*, 1993). In the latter case, this may be due to an increase in the volume of deoxygenated water (boosting mortality), or to increased sedimentation (spoiling nursery grounds and damaging eggs). This decreases the transfer efficiency of primary productivity through the foodweb and, along with fishing pressure, can have a severe impact on stock biomass and yields (Kemp *et al.*, 2001).

It is now widely recognized that many of the services provided by lake ecosystems are affected not only by direct exploitation, but also by changes in land use, vegetative cover, and other activities within the catchment area (Postel and Carpenter, 1997). Fertilizer run-off from agricultural land is among the major causes of nutrient loading and hence eutrophication, others being soil erosion, atmospheric deposition, and point pollution from industrial and human wastewater sources. It follows that agricultural policies may be implicated in changing fishery yields.

As in the other two riparian countries, fish production in Kenya has grown sharply over the past four decades. Output increased from around 17 000 t per year in the 1960s to more than 200 000 t in the early 1990s. This increase is attributable to the introduction of Nile perch (*Lates niloticus*) into Lake Victoria in the early 1960s. In the 1980s, catches of that species increased exponentially, rising in a few years from virtually zero to almost 60% of total yield (Ogutu-Ohwayo, 1990). Since 1994, however, landings of all fish have been in sharp decline, but especially those of Nile perch. By 1998, Nile perch landings were half of those at the beginning of the decade, despite increased fishing effort. Overfishing is one cause of the trend, but it is clearly not the only one. Eutrophication is also an important factor. Lake Victoria has progressively shifted from a mesotrophic to a eutrophic state (Gophen *et al.*, 1995; Lung'ayia *et al.*, 2001).

We use the interaction between land use and the fishery to identify catchment-based indicators, in particular price indicators, 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 us to use current prices to predict future changes in stock size.

Modelling the fishery

Following Kasulo and Perrings (2002), we include Chl *a* concentration (a proxy for phytoplankton density) as a measure of nutrient enrichment in a Gordon–Schaefer fishery model (Gordon, 1954; Schaefer, 1954, 1957), allowing for a delay in the impact of water quality change on stock growth, in discrete time:

$$\dot{X} = rX_tW_{t-1}\left(1 - \frac{X_t}{KW_{t-1}} - eW_{t-1}\right) - qE_tX_t \tag{1}$$

where X_t is the aggregate stock biomass at time t (in tonnes), W_{t-1} is Chl *a* concentration at time $t-1$ (mg m^{-3}

Table 1. Maximum sustainable yield (MSY), open access (oa) and profit-maximizing (*) steady-state solutions for Models 1 (standard Gordon−Schaefer model) and 2 (with environmental variable (p, market price of fish; c, cost of fishing effort; δ, discount rate).

Model 1	Model 2
$X_{msy} = \frac{K}{2}$	$X_{msy}(W) = \frac{K}{2}[W(1 - eW)]$
$Y_{msy} = \frac{rK}{4}$	$Y_{msy}(W) = \frac{rK}{4}\left[W^2(1 - eW)^2\right]$
$E_{msy} = \frac{r}{2q}$	$E_{msy}(W) = \frac{r}{2q}[W(1 - eW)]$
$X_{oa} = \frac{c}{pq}$	$X_{oa}(W) = \frac{c}{pq}$
$Y_{oa} = \frac{cr(pqK-c)}{p^2q^2K}$	$Y_{oa}(W) = \frac{cr\left(pqKW - pqKeW^2 - c\right)}{p^2q^2K}$
$E_{oa} = \frac{r(pqK-c)}{pq^2K}$	$E_{oa}(W) = \frac{r\left(pqKW - pqKeW^2 - c\right)}{pq^2K}$
$X^* = \frac{K}{4}\left[\left(\frac{c}{pqK} + 1 - \frac{\delta}{r}\right) + \sqrt{\left(\frac{c}{pqK} + 1 - \frac{\delta}{r}\right)^2 + \frac{8c\delta}{pqKr}}\right]$	$X^*(W) = \frac{K}{4}\left[\left(\frac{c}{pqK} + W - eW^2 - \frac{\delta}{r}\right) + \sqrt{\left(\frac{c}{pqK} + W - eW^2 - \frac{\delta}{r}\right)^2 + \frac{8c\delta}{pqKr}}\right]$
$Y^* = rX^*\left(1 - \frac{X^*}{K}\right)$	$Y^*(W) = rX^*(W)W\left(1 - \frac{X^*(W)}{KW} - eW\right)$
$E^* = \frac{Y^*}{qX^*}$	$E^*(W) = \frac{Y^*(W)}{qX^*(W)}$

or $\mu g\,l^{-1}$), E_t fishing effort at time t (thousand boat-days), r the intrinsic growth rate of the stock, q its catchability coefficient, and e is a coefficient representing the amount by which a unit change in Chl *a* depresses r. The model implies that nutrient loading positively affects growth until some maximum, after which further increases cause a decrease in the maximum sustainable yield (MSY), open access, and profit-maximizing levels of effort, and stock size. Which measure is appropriate depends on the management regime and the set of property rights. The degree to which the Lake Victoria fisheries have been regulated or policed varies over time. For completeness, Table 1 reports the steady-state solutions for fish stock (X), catch (Y), and fishing effort (E) for each management regime by comparison with those of the standard Gordon−Schaefer model.

Equation (1) is estimated using Schnute's (1977) method, which involves transforming the Gordon−Schaefer model into a linear form, then fitting by linear regression. We use annual data (1989−1998; Table 2) on fishing effort and cpue for the Kenyan fisheries of Lake Victoria (Othina and Tweddle, 1999), and a time-series of water quality data estimated through an Ecosim (Walters *et al.*, 1997, 2000) dynamic simulation using Ecopath (Christensen and Pauly, 1992). Biological inputs and diet composition data were taken from the Ecopath model for the Kenyan waters of Lake Victoria developed by Villanueva and Moreau (2002).

From these data, the estimated biomass for 1989 was used, together with the time trend in relative effort, as starting value in a nine-year dynamic simulation. The dynamic model was then fitted to the time-series of cpue for the Nile perch fishery, and two observations (1994−1995, 1997−1998) of average Chl *a* concentration (Kenyanya, 1999; Lung'ayia *et al.*, 2000) were converted to phytoplankton biomass ($t\,km^{-2}$). We assume that the difference between the first simulation of the dynamic model and the model that best fits the observed data is due to variations in annual primary production alone. The resulting long-term forcing function is then used to predict annual average phytoplankton biomass for the period 1989−1998. These values were then converted into Chl *a* concentrations (Table 2), assuming an average mixing depth of 10 m and a phytoplankton:chlorophyll ratio of 70 (Scheffer, 1998), although the Chl *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).

Following Schnute (1977), and given the basic Schaefer (1954) assumption that cpue (U) is proportional to stock abundance (U = qX), the model is transformed to

$$\frac{\dot{U}}{U_t} = rW_{t-1}\left(1 - \frac{U}{qKW_{t-1}} - eW_{t-1}\right) - qE_t \qquad (2)$$

Table 2. Estimated Chl *a* concentration (W_t) in Kenyan waters of Lake Victoria (cpue and fishing effort data from Othina and Tweddle, 1999).

Year	cpue ($kg\,boat^{-1}\,d^{-1}$)	Effort ('000 boat-days)	W_t ($mg\,m^3$)
1989	180	1 202	16.5
1990	152	1 387	17.6
1991	145	1 496	12.6
1992	137	1 606	15.3
1993	115	1 862	13.1
1994	93	1 862	16.0
1995	86	2 007	19.3
1996	78	1 971	12.0
1997	93	2 008	24.4
1998	86	2 190	23.0

Table 3. Parameter values for Models 1 (standard Gordon−Schaefer) and 2 (with environmental variable) with t-ratio in parenthesis (**$p < 0.02$; *$p < 0.05$).

Coefficient	Model 1	Model 2
β_0	2.01 (3.0)*	—
β_1	—	0.20 (3.1)*
β_2	−0.0070 (3.3)**	−0.0060 (3.7)**
β_3	−0.00075 (3.0)*	−0.00063 (3.3)*
β_4	—	−0.0059 (2.8)*
r^2	0.67	0.76
Adj r^2	0.56	0.62
p-value (F-statistic)	0.04	0.05

By integrating over time and using a time-averaged expression, $(W_{t-1} + W_t)/2$, for water quality at year $t-1$, we have

$$\ln\left(\frac{U_{t+1}}{U_t}\right) = r\left(\frac{W_{t-1}+W_t}{2}\right) - \frac{r}{qK}U_t - re\left(\frac{W_{t-1}+W_t}{2}\right)^2 - qE_t \qquad (3)$$

Equation (4) conforms to the multiple linear regression (Model 2):

$$Y = \beta_1 X_1 - \beta_2 X_2 - \beta_4 X_1^2 - \beta_3 X_3 + \varepsilon \qquad (4)$$

while the standard Gordon−Schaefer equation (Model 1) is

$$Y = \beta_0 - \beta_2 X_2 - \beta_3 X_3 + \varepsilon \qquad (5)$$

where $\beta_0 = r$, $\beta_1 = r$, $\beta_2 = r/qK$, $\beta_3 = q$, and $\beta_4 = re$, are coefficients to be estimated, and ε is the error term. All estimated coefficients for both models (Table 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 and Walters, 1992). The model with the environmental variable fitted the data much better than the standard model, consistent with the findings of Kasulo and Perrings (2002).

The parameter values indicate that the growth of the stock is positively affected by Chl *a* concentration at low levels, but that a maximum is reached at $W_{t-1} = 17.1$ mg m^{-3}. At this point, the maximum sustainable yield attainable is 195 000 t and the stock 227 000 t (Figure 1a). A further increase in

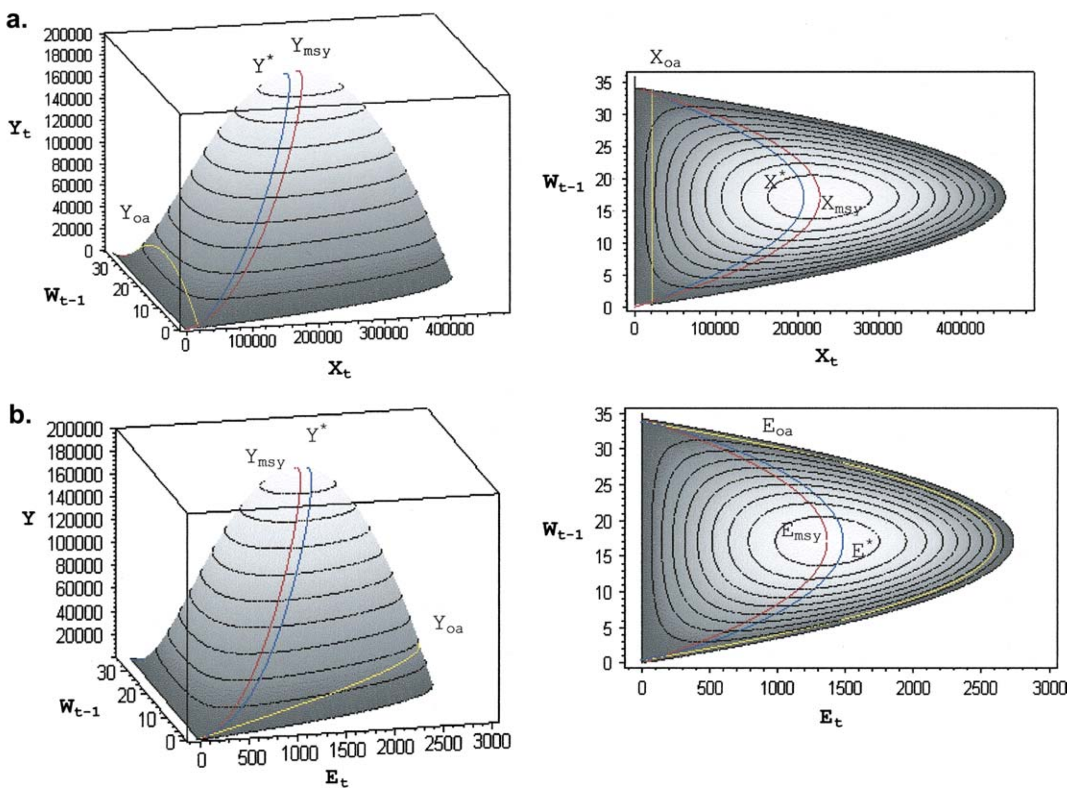

Figure 1. Effect of Chl *a* concentration (W) on (a) yield−biomass (Y/X), and (b) yield−effort surface (Y/E; cf. Table 1 for subscript identification).

nutrient load has a negative impact on growth and on harvests. The values for MSY, given the increasing level of Chl *a* (Figure 1), are supported by previous MSY estimates, which increased from an average of 20 000−50 000 t during the 1960s, 1970s, and 1980s (Geheb, 1997), to 150 000 t in the late 1980s (Republic of Kenya, 1988), then declined to 107 000−135 000 t (Brown *et al.*, 2002).

Steady-state conditions under both open access (Y_{oa}) and profit-maximizing regimes (Y^*), assume constant values for ex-vessel prices [$p = 6198$ Ksh t^{-1} (US\$376 at 1987 values)], cost of fishing effort [$c = 84\,188$ Ksh (US\$5102 per thousand boat-days at 1987 values)], and a nominal discount rate ($\delta = 0.255$). These are average values during the period 1980−2000. They turn out to be sensitive to water quality (Figure 1a).

Similarly, the related yield−effort function depends also on water quality. Assuming the same values for p, c, and δ, 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 Chl *a* concentration (Figure 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 over the period 1980−1998 (Figure 3) based on the estimated trend in effort (E_t) and Chl *a* concentration (W_{t-1}). Because of the lack of data before 1989, effort had to be estimated for that period. Using the 1989−1998 data set (Table 2; Othina and Tweddle, 1999), we estimated the output price elasticity of effort (Table 4), under the assumption that prices at time t (p_t) are a measure of the expectations of fishers at the beginning of the year, according to which they choose the amount of effort at time t. The resulting ln-ln regression fitted the data much better when using nominal prices ($r^2 = 0.97$) rather than deflated values ($r^2 = 0.44$; t-ratio in parenthesis):

$$\ln E_t = 5.8085_{(56.9)} + 0.1517_{(16.3)} \ln p_t \tag{6}$$

Chl *a* concentration for the period 1980−1988 was assumed constant at 8 mg m^{-3}, based on the lowest values from measurements documented by Ochumba and Kibaara (1989) for 1986; for subsequent years the Ecosim values (Table 2) were used. Given these time-series of effort and Chl *a*, Figure 2a compares the observed yield with the predicted harvest from Models 1 [Y(E)] and 2 [Y(E,W)], as well as with a model in which Chl *a* was kept constant at 8 mg m^{-3} for the entire period [Y(E,W = 8)]. Clearly, the model with the environmental variable better approximates the path of the observed catch than the standard Gordon−Schaefer model [Y(E)], and a shift in primary productivity is required to describe the observed pattern. This implies that the sharp increase in fish landings over the past decade may be explained as much by primary productivity as by fishing effort.

Table 4. Estimated fishing effort (E) from the price elasticity equation (based on observed effort, 1989−1998; cf. Table 2) and ex-vessel prices (ρ_t: at constant 1987 value; ρ_n: nominal prices; adapted from Republic of Kenya: Statistical Abstracts, 1980−1993, and Fisheries Bulletins, 1997−2000).

Year	E ('000 boat-days)	ρ_t (Ksh t^{-1})*	ρ_n (Ksh t^{-1})*
1980	1 005	5 966	2 918
1981	1 026	5 460	2 987
1982	1 040	4 102	2 703
1983	1 016	2 832	2 079
1984	1 059	3 064	2 479
1985	1 103	3 139	2 872
1986	1 129	3 394	3 228
1987	1 162	3 701	3 701
1988	1 239	4 481	5 032
1989	1 230	3 314	4 225
1990	1 429	6 629	9 778
1991	1 478	5 778	10 192
1992	1 550	4 889	10 975
1993	1 801	6 158	20 185
1994	1 943	6 116	25 822
1995	1 946	6 001	25 720
1996	2 083	7 884	36 825
1997	2 024	5 284	27 443
1998	2 160	7 114	39 457
1999	2 148	6 131	35 947
2000	2 204	6 008	38 752

*In 1987 US\$1 was equal to 16.5 Ksh (Kenyan shillings).

The Nile perch population explosion during the 1980s, and the increased productivity of the fishery during the early 1990s, attracted new and largely unregulated investment in the sector, causing a profound transformation of the fishing industry, and a steady increase in fishing effort. In the early years, the success of the Nile perch fishery was recognized not only in east Africa, because an international market for the species was quickly established (Crean *et al.*, 2002). This prompted an unprecedented inflow of national and international capital, transforming the once artisanal 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 and Ikiara, 2000). After 1990, the estimated effort departed from the steady solutions for MSY (E_{msy}), and the industry appeared to change from an almost-profit-maximizing effort level (E_{oa}) in the 1980s, towards an open access rent-dissipating effort level (E^*) in the 1990s (Figure 2b).

Modelling land−water interactions

A relation between land use and nutrient loading in Lake Victoria has long been argued in the literature (Hecky and

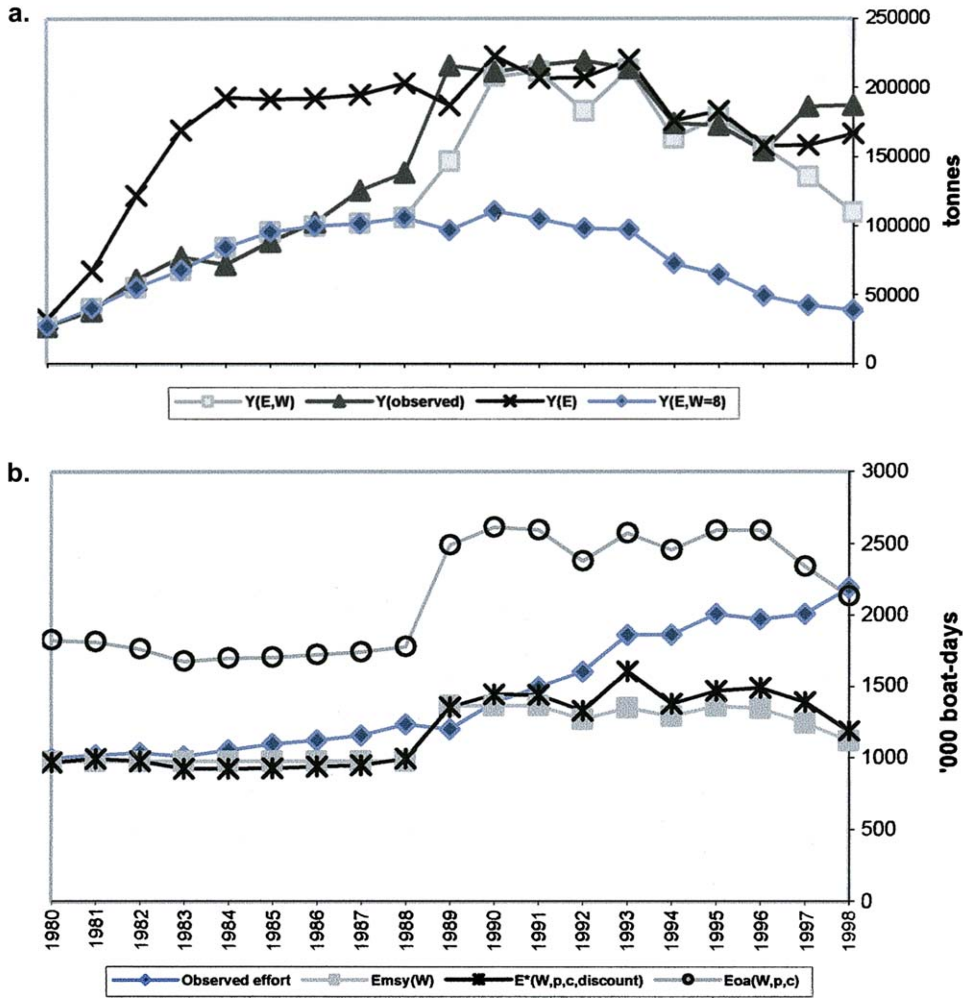

Figure 2. (a) Comparison of observed fish yields against model-predicted yields, and (b) observed/estimated fishing effort in comparison with steady-state solutions for effort (cf. Table 1 for subscript identification; starting value of stock biomass in 1980: 43 627 t; Kudhongania and Cordone, 1974).

Bugenyi, 1992). However, lack of long-term monitoring data and the complexity of ecosystem changes have made it difficult to estimate a land use–water quality function (Verschuren *et al.*, 2002). Nevertheless, our working hypothesis here is that the observed change in productivity is at least partly the result of excess nutrient loading from the drainage basin.

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 and Nevejan (1996) on pollution sources in the Kenyan part of Lake Victoria's catchment reported that P loadings from fertilizer use varied between 5000 and 22 000 t y^{-1}, whereas animal manure contributed roughly 3000 t y^{-1}, and domestic waste just 100 t y^{-1}. By contrast, N loading from fertilizers varied between 4000 and 22 000 t y^{-1}, compared with 40 000 t y^{-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 Chl *a* concentration. We suggest a simple water quality (W_t in mg m^{-3} Chl *a*) "production function" of the Cobb–Douglas form, in which fertilizer imports (F_t in thousand tonnes) are the sole argument (Table 5). 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):

$$\ln W_t = -5.6568_{(2.11)} + 0.6825_{(3.16)} \ln F_{t-1} \tag{7}$$

Table 5. Annual imports (I) and prices (ρ_t; at constant 1987 value; from World Bank, 1992, 1998, 2003; Gitu and Nzuma, 2003) of fertilizers and predicted Chl *a* concentration (W_t) based on Pf_{t-1}.

Year	I ('000 t)	ρ_t (Ksh t^{-1})*	W_t (mg m^{-3})
1980	130	4 984	—
1981	209	4 223	11.4
1982	130	3 654	13.0
1983	151	4 539	14.6
1984	74	4 626	12.3
1985	268	4 240	12.1
1986	318	3 303	13.0
1987	226	3 452	15.9
1988	226	3 873	15.3
1989	302	3 609	14.0
1990	160	2 819	14.8
1991	273	2 597	18.1
1992	241	2 772	19.4
1993	295	3 444	18.4
1994	269	3 009	15.4
1995	144	3 912	17.2
1996	295	3 501	13.8
1997	287	2 939	15.2
1998	229	2 767	17.5
1999	345	2 712	18.4
2000	337	2 502	18.7

*In 1987 US$1 was equal to 16.5 Ksh.

As fertilizer use depends on its prices (ρ, 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 $= 0.003$):

$$\ln F_t = 22.0741_{(7.74)} + 1.2011_{(3.43)} \ln \rho_t \tag{8}$$

Equations (7) and (8) allow us to explain changes in fish stocks and yields at time t as a function of ρ_{t-2} via the effect of the latter on W_{t-1}. The yield−effort curve can then be explained as a function of the two-year lagged price of fertilizer (Figure 3a). The relations between fertilizer prices and fish biomass and yields under each management regime are indicated in Figure 3b and c, respectively.

It is intuitive that as ρ_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 ρ_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 ρ_t will induce farmers to apply more, so increasing nutrient loading and hence fish production. However, as ρ_t falls further and fertilizer applications increase, ultimately the lake will converge to a eutrophic state, with increasingly adverse effects on fish production. Figure 3b and c indicates that both biomass and yield peak at a little over 3000 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 2 years in advance. The predicted yields using the observed

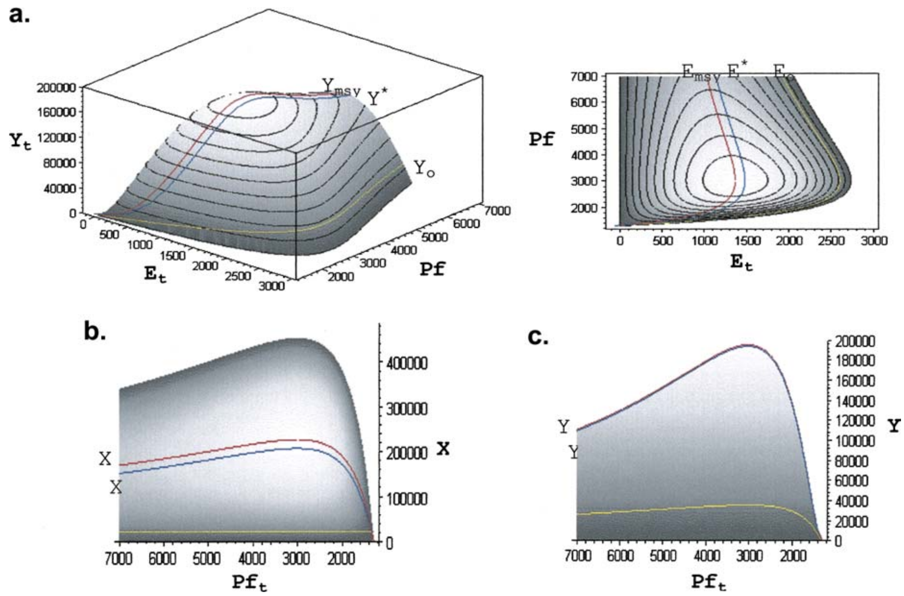

Figure 3. (a) Yield−effort (Y/E) surface in relation to fertilizer price (ρ) and steady-state solution paths for (b) fish stock biomass (X) and (c) yield (Y).

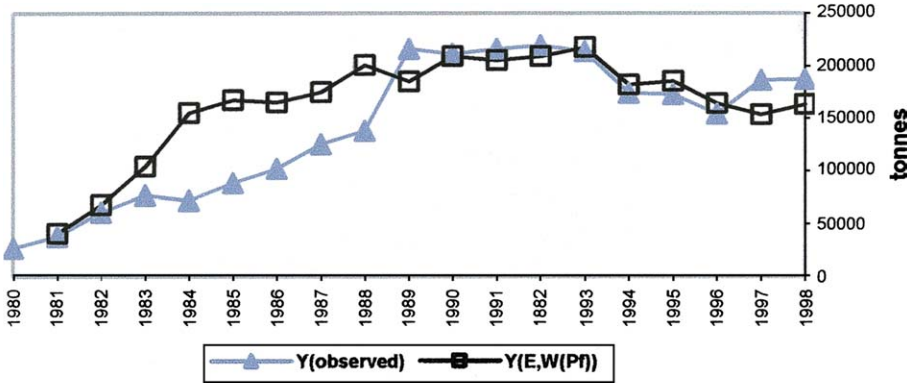

Figure 4. Comparison of observed yields and model-predicted yields based on fertilizer prices.

trend in real fertilizer prices [$Y(E,W_{Pf})$] closely approximate (Figure 4) 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 2 years ahead.

Conclusions

Traditional stock assessments depend on simple population models in which environmental variability is taken to be exogenous. Given the interdependence between terrestrial and aquatic activities, however, price signals that guide land-use decisions may be used as indicators of impending changes in fish stocks and yield. This is particularly interesting when the relationship between two sets of activities is a lagged one.

If the population biology models used to support stock assessments are integrated into ecological-economic models of the interactions between sectors, the range of indicators that can signal changes in stocks may be extended well beyond the direct fishery indicators traditionally used. This has advantages both where the dynamics of the interactions are such that the indirect indicators lead the direct indicators, and where the indirect indicators may be subject to less measurement error, or may be more reliably or frequently sampled than the direct indicators. Price data are not error free, but they are ubiquitous. Moreover, because prices and price expectations guide resource allocation decisions in the sectors that impact fisheries, they typically lead changes in the state of fisheries.

Perrings (2000) made the point that where markets are incomplete, prices are not sufficient indicators of resource scarcity. In the case discussed here, fertilizer prices do not reflect the impacts of fertilizer run-off on the fisheries of Lake Victoria. That impact may be positive or negative, depending on initial conditions in the lake, but in either case fertilizer prices will not be affected. The impact on the fishery is an externality of fertilizer use. From a fishery management perspective, understanding that impact allows the manager to predict and to adapt to impending changes in stocks; from an economic perspective, it allows corrective action to mitigate the effects on stocks. In either case, the prices that guide fertilizer use may be a key early warning of changes in fish stocks.

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