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ANALYSIS

Fishing down the value chain: Biodiversity and access regimes in freshwater fisheries — the case of Malawi

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ABSTRACT

This paper considers the connection between the diversity of catch in a multi-species fishery and the productivity of the fishery under different access regimes. A modified Gordon–Schaefer model is used to analyse the importance of the level of diversity in a fishery in open access and profit maximising regimes. The modified model, which includes both environmental and bioeconomic variables, is fitted to data from a gillnet fishery in Lake Malawi. Pressure on stocks is shown to be greater at all levels of biodiversity in open access than it is in profit maximising regimes. However, in a profit maximising regime both catch and the productivity of fishing effort is highest when there is a single marketed species. By contrast, in an open access regime catches are maximised at higher levels of bioeconomic diversity than in profit maximising regimes. Implications for policy are discussed.

1. Introduction

A long-held view of the development of fisheries is that they initially exploit more abundant, more easily caught species, and switch over time to increasingly less abundant, less easily caught species. Moreover, it is argued that the rate at which less easily caught species are substituted for more easily caught species is accelerated in open access fisheries. A study of the mean trophic level of species groups reported in FAO global fisheries statistics has shown that the succession has been from long-lived high trophic level piscivorous bottom fish towards short-lived low trophic level invertebrates and planktivorous pelagic fish (Pauly et al., 1998). There is less evidence on changes in the biodiversity of freshwater than marine fisheries. Nonetheless, it is known that the depletion of fish species has affected the functioning of freshwater ecosystems in which the most sensitive components of food webs, energy flows and biogeochemical cycles are those where the number of species carrying out functions is small (Schindler, 1990). Pauly (1997) has argued strongly that open access freshwater aquatic systems are particularly threatened by an influx of people displaced from other parts of the economy.

Recently, the FAO has questioned the degree to which the state of African fisheries is determined by the institutional conditions under which fishing takes place, arguing that changes in fish stocks may be more heavily influenced by environmental conditions than by fishery management (FAO, 2003a,b, 2004). The fishery investigated in this paper, in the south west arm of Lake Malawi,[1] does not appear to

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[1] The south-west arm of Lake Malawi is approximately 50 km long, 30 km across and reaches a depth of 100 m in the centre. Most of the shoreline, and particularly the south coast is heavily reeded, backed by extensive marshes and small lagoons. The north east coast is rocky, and the north west coast has extensive sandy bays with a few minor rocky outcrops. Three islands called the Maleris are an important feature of this region and an intensive chilimira fishery operates in their vicinity. A major river, the Linthipe, flows into the lake opposite the Maleri Islands (Tweddle et al., 1994).

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bear this out. The fishery has been exploited by traditional methods for many years, and traditional fisheries still account for the bulk of output. Modern regulated commercial fishing started in 1935 when purse seining was introduced, but it was not until the introduction of trawling in 1968 that the commercial fishery became a significant industry (Tweddle and Magasa, 1989). There is, however, striking evidence of changes in the fishery since that time. An initial report on the fishery (Tarbit, 1972) was followed by detailed investigations of changes that had occurred as the fishery developed (Turner, 1977; Tweddle and Turner, 1977). These studies showed significant changes in the species composition of catches as the trawl fishery intensified, with larger cichlid species disappearing from the catches and smaller species increasing in abundance. The initially abundant medium and large cichlid species, *Lethrinops stridei* and *L. macracanthus*, for example, were replaced by small cichlids such as *Otopharynx argyrosoma*, *Pseudotropheus livingstonii* and *Lethrinops auritus*. A decline in the large catfish *Bagrus meriodionalis* and *Bathyclarias* spp. was also noted (Banda et al., 1996).

During this period the FAO (1976) used experimental trawls to estimate total fish biomass and to recommend maximum sustainable yields (MSY) for each of seven fishing areas. These estimates were used to specify total allowable catches and to determine the number of trawling licenses to be issued. Following the FAO (1976) study, the minimum legal trawl cod-end mesh size was increased from 25 to 38 mm. This was intended to reverse the decline in population of the larger cichlid species. However, no attempt was made to see if the hoped-for recovery of the large cichlid species had occurred until 1989, when experimental trawls were again used to estimate total fish biomass and the species composition of the catch. They showed that species such as *L. mylodon* and *L. macracanthus*, which had declined substantially in the 1970s, had become locally extinct. Other large species such as *Taeniolethrinops furcicauda*, *T. praeorbitalis* and *Ctenopharynx* spp., had also declined. In the areas where most trawling had taken place, the large and medium-sized benthic zooplanktivores, large sediment feeders and large piscivores had all declined. The standing stock of small sediment feeders and pelagic species had remained largely unchanged (Turner et al., 1995; Banda et al., 1996). A one-year moratorium on trawling in the south-west arm of the lake in 1992/1993 had a marked effect on both fish biomass and species composition, returning composition to that in other areas. In a follow-up study by Banda et al. (1996), no further major changes in species composition were detected.

This paper addresses one of the questions raised by the FAO (2003a,b) studies: namely what is the effect of fisheries management and access rules on the biodiversity of the fishery. Traditional fisheries in Malawi are of several types. The most commonly used gear is the gillnet, which accounts for over 40% of total traditional catches of around 40,000 t per annum. In this sector, catch per unit effort in the south west arm of Lake Malawi showed a sharp decline from about 20 kg per set in the mid 1950s to 5 kg per set in the early 1970s. The decline was largely accounted for by the disappearance of nchila (*Labeo mesops*) which had comprised about half of the catch in the 1950s (Walker, 1976). Since that time the species composition of catch has continued to change with the percentage contribution of tilapiines (*Oreochromis* spp.) falling and the contribution of usipa (*Engraulicypris sardella*) and haplochromines (kambuzi, utaka and chisawasawa) increasing (Munthali, 1997).

We ask how both the size and species composition of catches is influenced by management and access regimes. We do not find the expected relation between open access regimes and loss of biodiversity. We do find that the performance of the fishery under different access regimes may be quite sensitive to the diversity of fish stocks. More particularly, we find that traditional open access regimes are associated with a higher diversity of fish catches than regulated access profit-maximising regimes. We discuss the implications of this for the management of freshwater fisheries in developing countries.

The paper is presented in six sections. The following section discusses Malawi's fisheries management and access regimes. This is followed in Section 3 by a presentation of the methodology used to derive results on the potential impact of biodiversity that provide a testable set of hypotheses about the behaviour of multi-species fisheries under different access regimes. Section 4 gives empirical results for the case of the Malawi gillnet fishery. Sections 5 and 6 then use the results to draw conclusions about the relation between biodiversity and access regimes in this fishery.

2. Fisheries management and access regimes in lake Malawi

Our discussion of fisheries access regimes uses Bromley's (1991) classification of natural resource regimes. Bromley distinguishes between common property and non-property (open access) regimes. Under common property regime a group of owners has a right to exclude non-members and non-members have a duty to abide by the exclusion. On the other hand, individual members have both rights and duties with respect to use rates and maintenance of the resource. Under open access there is no defined group of users or owners and benefit stream is available to anyone. Thus, individuals have a privilege but no right with respect to use rates and maintenance of the resource.

In pre-colonial times the Lake's resources of water, fish and shores were under a common property regime. Indeed, until the end of the First World War the fishery was regulated by family heads, village headmen and chiefs. The first fishing regulations were introduced in 1930. These prohibited fishing by traps, weirs and poisoning and excluded foreign commercial fishers from operating within two miles of traditional fishing grounds. However, enforcement of the regulations was lax (Chirwa, 1996). A Fisheries Department was established in 1946, and charged with the responsibility for regulating fishing and the fish-trade, for compiling fisheries statistics, and for conducting research. This implicitly transferred the control and ownership of the lake's resources from the communities to central government. During the next two decades the Department experimented with different types of fishing gear and boats, and proposed new fishing regulations that introduced licenses for commercial fishers. However, enforcement remained a problem due to the mobility of fishers and their seasonal fishing practices, and monitoring remained weak (Chirwa, 1996).

After independence, the Malawi government's policy on fisheries was to maximize sustainable yield from stocks that could be economically exploited in national waters. It also aimed at improving the efficiency of exploitation, processing and marketing of the fish (OPC, 1971). These policies were reflected in the Fisheries Act (1974), which authorized the use of three instruments: licensing, gear restrictions and closed seasons. First, licenses were required for commercial fishing, fishing vessels, trout fishing and fish trading. There was a provision in the Act to limit the number of licenses for any class of fishing in Malawi. The aim was to prevent the depletion of fish stocks by limiting the number of licenses. Second, certain fishing methods were prohibited including the use of certain nets, explosives and poison. Third, the Act prohibited fishing during certain specified periods, either with respect to particular areas of the country or to Malawi as a whole. Policing of these regulations was unsuccessful due to lack of trained staff and patrol equipment, and the low level of penalties for non-compliance.

Initially, the Fisheries Department was seen as a guardian of fish stocks. More recently the emphasis has shifted towards concern for the fishing community and consumers, and fish resources are seen more as a source of sustainable benefits to both. As elsewhere, the emphasis has shifted towards co-management designed to exploit local knowledge and skills (Bland and Donda, 1994). This is reflected in the Fisheries Conservation and Management Act (1997) that implements the National Fisheries and Aquaculture Policy. The most important feature of the new Act is the provision it makes for co-management through legally binding agreements between the government and recognized fisher associations. Only fishing areas not subject to a co-management agreement continue to be managed by the Fisheries Department (EAD, 2000).

In summary, the fish resources of Lake Malawi were initially managed as a traditional common property resource until government regulations were introduced. The introduction of government control led to the disappearance of the common property regime, since all natural water bodies in Malawi and its resources became the property of the state. The Fisheries Department as a government institution was given the mandate to control the fish industry, but was largely unsuccessful. In particular regulation of traditional fisheries was weak. In fact, since the introduction of government regulation, the traditional fishery can rightly be considered as being an open-access regime. The introduction of co-management regimes has returned some authority to fishing communities, but given that stocks are shared between many communities the return to each community depends on the actions of neighboring and distant communities who fish the same stocks (EAD, 2000).

3. A modified Gordon–Schaefer model of a multi-species fishery

To investigate the relation between the diversity of harvest and the productivity of a fishery regime, we use a modified Gordon–Schaefer model. Stock production models of this type have been criticized mostly because they do not use biological information on natural mortality, growth or age. Nevertheless, they are still widely used and accepted particularly in tropical lakes where biological parameters of catches are not easy to get, and where the species concerned are numerous and short-lived so that aging is often difficult (Fréon et al., 1993).

The modification to the Gordon–Schaefer includes both the effect of the species diversity of fish catches on yields, and the effect of environmental conditions on the growth of fish stocks. In this case, we use a measure of water pollution. In Lake Malawi, as in most African lakes, water pollution is an emerging problem that needs close monitoring. Increasing levels of biological oxygen demand leads to a reduction in fish productivity through eutrophication and this reduces both the food and habitat available to some fish species (Hecky et al., 1999). Since most pollution of the lake is due to run-off from agricultural areas, we take of rainfall in the catchment area of the south-west arm of Lake Malawi as a proxy for the level of pollution.[2] This reflects the fact that annual yields of sediments and nutrients deposited in the lake are highly, and positively, correlated with rainfall in the catchment. For example in the high rainfall year of 1997/1998, deposition of suspended sediments from the southern catchment was nearly 50% greater than in 1996/1997 (Hecky et al., 1999). The use of rainfall figures also indirectly captures the impact of river discharges from Linthipe River, which is one of the largest riverine contributors of nutrients and sediments into the lake (Mwichande et al., 1999). Mkanda and Barber (1999) have used rainfall energy derived from rainfall data to develop a geographical information system (GIS) model for the prediction of soil loss, and sedimentation in Lake Malawi.

Biodiversity is measured by an index of the species diversity of the harvest and is assumed to reduce the catch per unit effort. Two indices are used: an unweighted and a price-weighted Simpson's index. The unweighted Simpson's index of catch is:

$$B_t = \sum_{i=1}^{s} \left(\frac{Y_i}{Y}\right)^2 \tag{1}$$

where Y represents the total fish catch and Y_i is the catch of the ith species. The price-weighted Simpson's biodiversity index of catch is:

$$\bar{B}_t = \sum_{i=1}^{s} \left(\frac{P_i Y_i}{TR}\right)^2 \tag{2}$$

where P_i is the unit price of species i, and TR is the market value of the total fish catch. The value of $\bar{B}_t$ is determined by changes in market conditions for different species and of changes in species composition overtime due to natural causes and to fishing pressure. In both the unweighted and weighted indices, a loss of biodiversity is reflected in an

[2] An ideal measure of water quality should have taken into account the level of chlorophyll *a*, or the concentration of nutrients and suspended sediment levels in the surface mixed water-layer of the lake. Unfortunately, this detailed information is not available on a long-term basis.

increase in the value of the biodiversity index, which ranges between zero (as the number of species harvested tends to infinity) and one (if the number of species harvested is one).

If each species caught has the same market value, then the bioeconomic Simpson's index of catch is the same as the unweighted index. If different species have different market values, the impact of price weighting depends on the relative abundance of more and less valued species. The economic biodiversity index of a community dominated by species of high (low) market values will be greater (less) than the corresponding ecological biodiversity index of the same community. That is

$$\frac{d\bar{B}_t}{dp_t} > (<) 0 \Rightarrow \bar{B}_t > (<) B_t$$

where $\bar{B}_t$ and B_t are the bioeconomic and biological diversity indices, respectively. Since the weighted biodiversity index reflects both the economic scarcity and the relative abundance of species, an ecologically dominant species will became more (less) dominant in the weighted index if it is more (less) valuable.

The use of a weighted biodiversity measure reflects the fact that consumers do not regard all species as equally valuable, and that the differences between species inform the decisions of both fishers and regulators. Empirically, the sequence of exploitation is from more to less valued species (Boechlert, 1996; Pauly et al., 1998), a phenomenon we have called 'fishing down the value chain.' By using these biodiversity indices, the model is able to capture important properties of multi-species fisheries without describing all the species interactions. It does this by aggregating the different species involved and then modeling the effect of the diversity of those species on the dynamics of the aggregated fishery. In this way, the effect of changes in biodiversity within the fishery on productivity is captured while maintaining the simplicity of the Gordon–Schaefer model.

The model is summarized as follows. First, suppressing time subscripts, the growth and sustainable yield functions are:

$$X = rX(1 - eW - X/K) - qBEX \tag{3}$$

$$Y = qKBE(1 - eW - qBE/r). \tag{4}$$

Where X denotes fish biomass, r is the intrinsic growth rate, W is the environmental quality variable, e is a parameter that gives the amount by which a unit change in the environmental variable depresses the natural growth rate of fish biomass, K is the maximum environmental carrying capacity, q is the catchability coefficient, E is effort, and Y is harvest. The implications of the model for the level of effort and stock size under different access regimes are obtained by solving for the optimal values of E and X under each access regime. Defining p to be the unit price of harvested fish biomass, c to be the unit cost of effort, and δ to be the discount rate, the results are summarized in Table 1.

The effect of a change in the species diversity of catch under different access regimes is then obtained by differentiating the steady state values of E and X under each access regime with respect to B. The results are summarized in the following proposition:

If an increase in biodiversity reduces the effectiveness of fishing effort then, in a multi-species fishery, the maximum sustainable yield level of stock size is not affected by changes in biodiversity, but the effort required to catch the MSY increases (decreases) as fish diversity increases (decreases). In open access and profit maximising fisheries, an increase (decrease) in fish biodiversity implies an increase (decrease) in stock size. However, the impact on the open access and profit maximising levels of effort depends on the level of biodiversity. For small numbers of species an increase (decrease) in fish biodiversity implies an increase (decrease) in effort levels. As the number of species rises, the effect on effort levels reverses, and an increase (decrease) in fish biodiversity implies a decrease (increase) in effort.

That biodiversity has no impact on MSY is an artefact of the way it has been modelled. In freshwater fisheries where species have been introduced, there is some evidence of an increase in total fish biomass over some time horizon despite a reduction in the number of species. The Lake Victoria fishery in East Africa is an example. The introduction of the Nile Perch into this fishery was associated with the loss of some 200 haplochromine cyclids, but a significant increase in total fish biomass (Kasulo, 2000; Perrings, 2000). Where species are merely deleted from an existing fishery, however, the evidence is less clear. It has been assumed here that biodiversity does not affect the productivity of the fishery, but does affect the effectiveness of fishing effort. The impact of the bioeconomic diversity of the catch on effort levels depends on parameter values. For low levels of biodiversity (B close to 1), the addition of a marketed species increases the optimal level of effort. As the number of species increases, however, the effect of an additional species on optimal effort falls, eventually becoming negative.

If the diversity of species had no implications for either the value of harvest or for the growth of fish stocks, the optimal strategy would be to reduce the number of species to one. From an ecological perspective, elimination of all species and all trophic levels bar one would effectively destroy the system, but simplification of a fishery to the point where the number of target species is reduced to one is not uncommon. It has to be noted that the value-adjusted or bioeconomic

Table 1 – Effort and stock size under different access regimes

Access regime	Level of effort	Stock size
Open access	$\frac{r(1-eW-c/pqBK)}{qB}$	$K(1-eW-qBE_{oa}/r)$
Profit maximising	$\frac{qB\xi \pm [((qB\xi)^2 - 8\phi q^2B^2\theta(\phi r(1-eW)-c))]^{\frac{1}{2}}}{4\phi q^2B^2}$ where $\xi = \phi(\theta + 2r(1-eW)) - c$ $\theta = \delta + r(1-eW)$	$\frac{c(\delta + r(1-eW) - qBE_0)}{pqB(\delta + r(1-eW) - 2bBE_0)}$
MSY	$\frac{r(1-eW)}{2qB}$	$\frac{K(1-eW)}{2}$

diversity index used in the model measures the diversity of catch weighted by the economic value of the species involved. Indeed, in the empirical case investigated in this paper it measures the diversity of species weighted by their market value. In most cases, the aggregate value of the catch is not independent of the diversity of species in that catch. The value of the aggregate catch increases over at least some range of biodiversity. This has implications for the fishery response to any change in environmental conditions.

4. Estimation of the modified Gordon–Schaefer model

The modified aggregate Gordon–Schaefer model is applied to the gillnet fishery of the south west arm of Lake Malawi. We first estimate the parameters of the model, and then use these parameters to estimate the open access, MSY and optimal solutions of the gillnet fishery. They are also used in the investigation of the biodiversity–productivity relationship, which analyses the impact of changes in biodiversity on open access and optimal solutions.

The parameters of the model are estimated using data on fish catch and price per species, fishing effort, cost of effort, and rainfall as a proxy for the level of water pollution. The period of study is between 1976 and 1998. Data on rainfall is collected by the Meteorological Department and is available from the Malawi National Statistical Yearbook. Data on fish catch and price per species and fishing effort were obtained from the Malawi Fisheries Department. No data are regularly collected on the cost of effort because it is very difficult to cost fishing effort in a traditional fishery (FAO, 1993). The cost of effort is therefore calculated on the assumption that rents are dissipated due to near open access condition.

The following two equations were estimated:

$$\ln\hat{X}_t^* = r - q\hat{E}_t^* - \frac{r}{qK}\hat{U}_t^* - reW_t^* + \mu \tag{5}$$

$$\ln\bar{X}_t^* = r - q\bar{E}_t^* - \frac{r}{qK}\bar{U}_t^* - reW_t^* + \mu \tag{6}$$

where

$$\hat{X}_t^* = \frac{\hat{U}_t^*}{\hat{U}_{t-1}^*};\quad \bar{X}_t^* = \frac{\bar{U}_t^*}{\bar{U}_{t-1}^*};$$

$$\hat{E}_t^* = \frac{(\hat{E}_{t-1} + \hat{E}_t)}{2};\quad \bar{E}_t^* = \frac{(\bar{E}_{t-1} + \bar{E}_t)}{2};\quad \hat{E}_t = E_t^* B_t;\quad \bar{E}_t = E_t^* \bar{B}_t$$

$$\hat{U}_t^* = \frac{(\hat{U}_{t-1} + \hat{U}_t)}{2};\quad \bar{U}_t^* = \frac{(\bar{U}_{t-1} + \bar{U}_t)}{2};\quad \hat{U}_t = \frac{Y_t}{E_t B_t};\quad \bar{U}_t = \frac{Y_t}{E_t \bar{B}_t}.$$

$$W_t^* = \frac{W_{t-1} + W_t}{2}.$$

Y_t represents the aggregate fish catch, E_t is the level of fishing effort[3] and U_t is the catch per unit effort. W_t is an environmental variable and μ is the error term. The annual level of fish biodiversity is measured by B_t. The weighted biodiversity index is represented by $\bar{B}_t$. Thus, Eq. (5) uses the unweighted biodiversity index while Eq. (6) uses the price-weighted biodiversity index. The dependent variable for the equations is an index of relative change in fish biomass. The parameters to be estimated are; r, the intrinsic growth rate, q the catchability coefficient, K the environmental carrying capacity, and e the parameter that relates how much a unit of the environment variable W_t reduces the relative growth of fish biomass.

Regression results for Eqs. (5) and (6), corrected for autocorrelation using a Prais–Winsten transformation, are reported in Table 2.

For Eq. (5), all parameter estimates with the unweighted biodiversity index are statistically significant at the 5% level, and the model explains 66 of the variation in fish biomass. The corresponding F-statistic is also significant. Addition of the weighted bioeconomic diversity index (Eq. (6)) further improves the significance of the estimated parameters, and the goodness of fit. The model now explains 74% of the variation in fish biomass.

5. Biodiversity in the fishery

Now consider the importance of biodiversity. Since we wish to relate biodiversity to the structure of property rights and the regulatory regimes in the fishery, we first consider the open access, MSY and profit maximising outcomes in the absence of a bioeconomic diversity index. The corrected parameter estimates from Eq. (6) are $r=1.3591$, $q=0.0000019$, $K=25954.98$, and $e=1.06$. These parameters are used to calculate the open access, MSY and optimal solutions, assuming $B=1$. The results are reported in Table 3. The price of output is set at the average beach price for fish in 1997, MK562.14 per tonne.[4] Cost per unit effort is again set at the average for the fishery over the same year, K1.70. Pollution is proxied by rainfall. This takes the form $W_{tn}=W_t/W_{av}$ where W_{tn} is the normalised index, W_t is the amount of rainfall in year t and W_{av} is the average rainfall. Since the value of B ranges between 1, where only one species is marketed, and $1/s$, where s species are marketed, the assumption that $B=1$ implies that for this exercise we are ignoring the impact of changes in the marketed value of different species. That is, we are treating the fishery as if it was a 'single species' fishery.

These results are typical of 'single species' Gordon–Schaefer fisheries models whether in freshwater or marine fisheries (Conrad and Adu-Asamoah, 1986; Gallastegui, 1983). The open access solution produces the lowest catch level associated with the highest level of effort. The MSY solution gives the highest level of catch and the profit maximising solution gives the lowest level of effort. Figures for the actual average indicate over-exploitation by comparison with the 'single species' fishery, since the average catch is higher than the MSY catch and the average effort is above the MSY effort level. Since MSY does not change with the biodiversity index the overexploitation is not an artefact of the choice of $B=1$. Indeed it is widely recognised that the fishery was overexploited during the period analysed, 1976–1998.

[3] For the gillnet fishery, effort is expressed in number of sets of 91 m (stretched length) net per year.

[4] MK = Malawi Kwacha, the local currency in Malawi.

Table 2 – Regression results corrected for autocorrelation

Equation	R	Q	r/qK	re	R^2	F statistic	DW statistic
3	1.3755 (3.3150)	−0.0000022 (−3.462)	22.563 (2.2750)	−1.4817 (−4.206)	0.655	10.7810	1.7615
4	1.3591 (3.3330)	−0.0000019 (−4.232)	27.4507 (2.3140)	−1.4422 (−4.434)	0.742	16.3267	1.7542

The bioeconomic index captures the effect of market valuation of harvest. The unweighted Simpson's index of may be a reasonable approximation of a bioeconomic index in a subsistence fishery in which there is no by-catch, and all species are consumed. The weighted index reflects the fact that demand is higher for some species than others, and that this influences which species are targeted by fishers. Lower bioeconomic diversity implies that fewer species are commercially valuable, and/or that fewer species are targeted by fishers. Since most fisheries in Lake Malawi, including the gillnet fishery, are market-driven, the weighted index is the appropriate measure of the diversity of catch.

The unweighted and weighted biodiversity indices for the fishery over the period are shown in Fig. 1. Two features are worth noting. The first is that the weighted index is generally higher than the unweighted index — sometimes very much higher. Fishers focus on the most highly valued species. The second is that the period divides into two. During the first ten years, both indices are increasing in value indicating that the diversity of catch was falling. It was also quite volatile, indicating that the range of species caught varied sharply from year to year. During the second ten years the trend was reversed. Both indices fell indicating that the diversity of catch was rising slightly. The variance in both indices also declined.

During the first period the fishery was becoming dominated by the high value chambo following the collapse of nchila stocks in the 1970s. During the second period, as chambo stocks themselves came under increasing pressure, emphasis switched to other less highly valued but more diverse stocks of usipa (*Engraulicypris sardella*) and haplochromines (kambuzi, utaka and chisawasawa). Fishing down the value chain ironically increased the bioeconomic diversity of catch.

We are interested in the linkage between these measures of biodiversity and the institutional and technological characteristics of the fishery. In a Malawian context the open access case corresponds to a poorly regulated fishery in which rents are exhausted. The profit maximising case corresponds to a fishery subject to well-defined property rights in which marginal revenues and marginal costs are equated. Since the persistence of an unregulated fishery is just as much a policy choice as the introduction of regulated fishery, we ask whether there is a connection between the diversity of harvest and the structure of the industry.

Recall that our proposition holds that in both open access and profit maximising fisheries, an increase in the bioeconomic diversity of catch implies an increase in the size of the fish stock, but that the impact on the optimal level of effort in each case depends on the parameters of the model. For B close to 1, an increase in the bioeconomic diversity of catch implies an increase in stock size. As the number of species rises, the effect on optimal stock size reverses, and an increase (decrease) in fish biodiversity implies a decrease (increase) in stock size.

To investigate the diversity–productivity link in the fishery, we consider the relationship between biodiversity, fish stocks and catch under both open access and profit maximising rules using the same data set. Specifically we calculate the sensitivity of the open access and profit maximising solutions to variation in the bioeconomic diversity of catch. Both cost per unit effort and the discount rate are held constant. Moreover, because we are interested in changes in relative prices only, the average landed price of fish is also held constant. The results are used to fit relationships between biodiversity, fish stocks, effort levels and catch under both an open access regime (Fig. 2) and a profit maximising regime (Fig. 3). This enables us to get some idea of the relative performance of the two fishery regimes at different levels of catch diversity, holding other characteristics of the fishery constant. While fishers have some capacity to determine the level of catch diversity through, for example, choice of fishing gear, catch diversity is largely driven by (a) the diversity of fish stocks and (b) market conditions, both of which are given.

Fig. 2 indicates the predicted level of effort, stock size and harvest for the open access fishery. It shows that fish stocks would be predicted to decline as the bioeconomic diversity index rises (i.e. as bioeconomic diversity falls). If the bioeconomic diversity of catch were high, both effort and catch would be predicted to increase as diversity falls. The stock size, on the other hand, would be predicted to decrease. If the bioeconomic diversity of catch were low, effort would be predicted to increase as bioeconomic diversity falls, but both stock and catch would be predicted to decline. The mean bioeconomic diversity index for the gillnet fishery in Lake Malawi is shown as a point of reference.

Fig. 3 shows the same data for a profit maximising fishery. As in the open access case, stock size would be predicted to decline monotonically as the bioeconomic diversity of catch

Table 3 – Estimates of catch, effort and stock levels for open access, MSY and profit maximising exploitation in the 'single species' fishery

Variable	Open access	MSY	Profit maximising	Actual average
Catch (Y)	879.84	1943.30	1910.40	2194.94
Effort (E)	290936	167227	145468	439490
Stock (X)	1585.35	6091.92	6884.59	2628.57

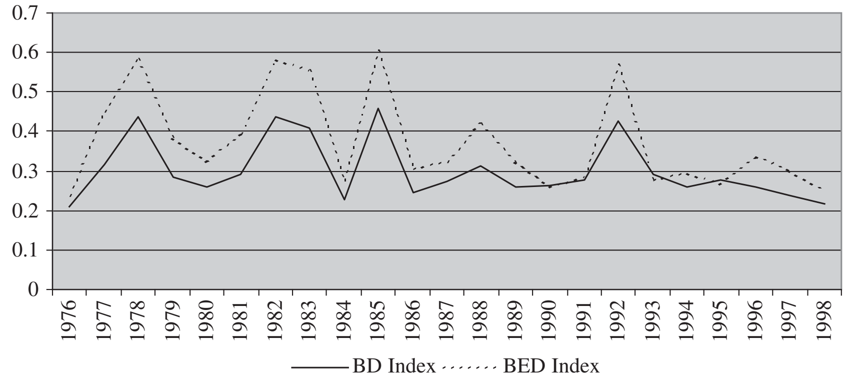

Fig. 1 – The biodiversity and bioeconomic diversity indices for catch in the gillnet fishery of the S.E. arm of Lake Malawi.

declines. However, both effort and catch would be predicted to rise monotonically. Indeed, both catch and the productivity of effort would be expected to be highest when the bioeconomic diversity index is 1 — there is a single marketed species.

Note that both figures reflect the parameter values of the particular fishery being evaluated, and the relation between institutional conditions, biodiversity and productivity may be quite different in different fisheries. In the Lake Victoria fishery, for example, the introduction of the Nile Perch had a strongly negative effect on the biodiversity of harvest, but a strongly positive effect on stock size. Nevertheless, we believe that our findings would hold in many traditional fisheries.

Consider the relative performance of the fishery in each of the two states. Gordon's (1954) observation – that any increase in productivity will result in a positive rent that will attract new entrants – holds here. Thus, under open access all the potential benefits of a fall in bioeconomic diversity are dissipated. Furthermore, the increase in productivity of fishing effort as biodiversity declines is offset by the steep decline in fish stocks resulting from free entry to the fishery. The result is that fish stocks under open access are lower at all levels of bioeconomic diversity than is the case in a profit maximising regime.

However, it is also clear that the relative efficiency of the two states is related to the diversity of catch. For $B=1$ a profit maximising regime clearly dominates an open access regime. As B falls, however, the performance gap between the two regimes closes, and for B below the mean bioeconomic diversity index estimated for the Lake Malawi fishery, an open access regime yields higher levels of predicted harvest and higher catch per unit effort than a profit maximising regime (Fig. 4). The implication is that for a fishery characterised by high levels of bioeconomic diversity of harvest, it is not obvious that a commercial (profit maximising) fishery dominates an traditional (open access) fishery.

6. Discussion

What are the inferences to be drawn from this? The bioeconomic diversity measure used in this paper captures the effect of a change both in the relative abundance of marketed species, and in the relative prices of those species. If an increase in the bioeconomic diversity of catch reduces the effectiveness of fishing effort, as this paper shows that it does in the Lake Malawi fisheries, there will be greater pressure on low diversity than on high diversity systems in both open access and profit maximising regimes. As one would expect, the pressure on stocks is greater at all levels of biodiversity in open access than it is in profit

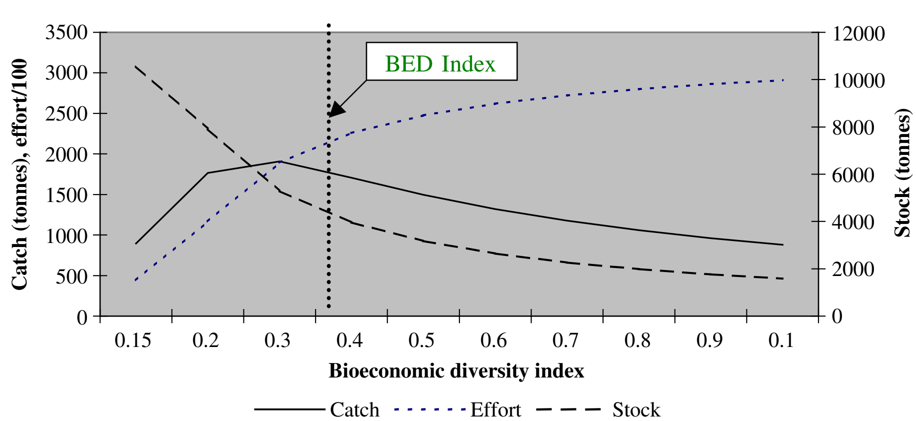

Fig. 2 – The effect of bioeconomic diversity of catch on stock size, effort and catch in an open access fishery.

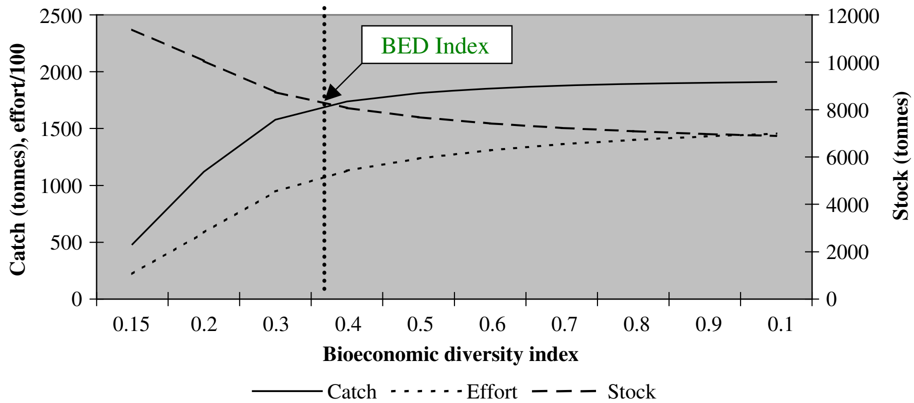

Fig. 3 – The effect of bioeconomic diversity of catch on stock size, effort and catch in a profit maximising fishery.

maximising regimes. However, the relative performance of the fishery under each regime type varies with the diversity of catch. In the relatively biodiversity rich fishery of the south west arm of Lake Malawi, a traditional poorly regulated gillnet fishery (open access) yields higher catches and catch per unit effort than a profit maximising fishery based on the same technology and fishing the same set of stocks.

There is less of an efficiency cost to the persistence of traditional open access regimes in this case, and less of an incentive to simplify the fishery. This is at least consistent with the general evidence on freshwater fisheries where traditional open access regimes have typically been less selective in the set of harvested species than profit-maximising regimes. This does seem to have important implications for the management of fisheries in Malawi, and potentially in other developing countries. The main point is that it is not obvious that the introduction of a regime based on the establishment of clearly defined access rights will offer a significant productivity advantages over traditional open access regimes in the Malawian fishery, and potentially elsewhere. More importantly, if biodiversity conservation in fisheries is an objective, then the differences in performance of the fishery under different access regimes should at least prompt investigation of the biodiversity implications of a regime change. Finally, we emphasise that our findings are based on a single fishery. Since the Malawian fisheries share many of the same characteristics as other multi-species freshwater fisheries in Africa and elsewhere, however, we think it likely that replication of the study would confirm at least the general relation between property rights, diversity of catch and productivity identified here.

Finally, to return to the findings of the FAO (2003a,b, 2004) studies on freshwater fisheries in Africa, while it is clear that year on year fluctuations in catches of particular species are indeed driven by environmental conditions, the targeting of species has historically had a significant impact on the relative abundance of those species. The commercial extinction of nchila (*Labeo mesops*) is a case in point, as is the decline in chambo (*Oreochromis* spp.). A characteristic feature of traditional fisheries in Lake Malawi, which has persisted in the partially regulated gillnet fishery studied in this paper, is the non-specificity of catches. For that reason such traditional fisheries perform relatively better than mono-specific commercial fisheries in highly diverse ecosystems, where by-catch imposes a cost disadvantage on the latter.

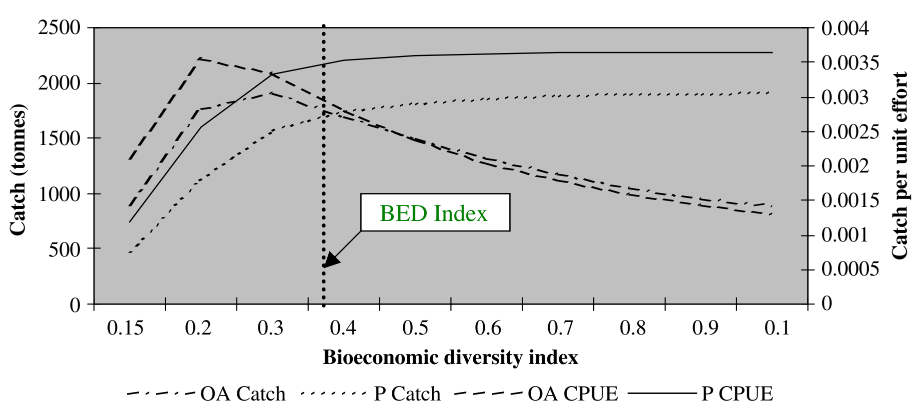

Fig. 4 – The relation between bioeconomic diversity of harvest on catch and catch per unit effort in open access and profit maximising fisheries.

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