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Simulating trade-offs between socio-economic and conservation objectives for Lake Victoria (East Africa) using multispecies, multifleet ecosystem models

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A R T I C L E I N F O

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A B S T R A C T

Most small scale inland fisheries worldwide are open access, and fishing provides the only source of employment and livelihood for the riparian communities. Management of these fisheries requires information on trade-offs between fish production, profits from fishing, employment, and conservation objectives. We use the non-linear optimization procedure in Ecopath with Ecosim (EwE) modelling package to determine long-term, gear-specific fishing effort that can maximize economic, social, and conservation objectives for Lake Victoria (East Africa). Then, the resulting "optimal fishing effort" levels are applied in both EwE and Atlantis models to simulate long-term changes in the ecosystem. Results show profit maximization to be more compatible with conservation objectives than is the maximization of catch (or employment). However, maximizing economic value, while maintaining ecosystem structure, would require a reduction in fishing effort of almost every fishing gear. This trade-off can be severe (high social cost) for fishing communities with limited alternative livelihoods. This study provides an understanding of relative risks and benefits of various management objectives, which will enable stakeholders and the public to conduct informed discussions on future management policies.

1. Introduction

Fisheries management globally relies predominantly on "tactical" advice, focusing on output control measures, such as quotas, and input controls, either in the form of gear limitations or closures of fishing areas (Stefansson and Rosenberg, 2005). The main goal is to maintain harvested stocks at levels that produce maximum sustainable yield (MSY, the highest equilibrium yield that can be continuously harvested from a stock under existing environmental conditions) (Tsikliras and Froese, 2018). However, achieving this goal in multispecies fisheries is a challenge because the biological yield of predators or prey species shift depending on the changes in abundance of their prey or predator species (Walters et al., 2005). For example, the fishing effort required to produce MSY for less vulnerable stocks, i.e., those with high turnover rate, can be high, thereby increasing the risk of overexploiting or even collapsing the vulnerable stocks. The alternative strategy that is increasingly receiving attention towards addressing this challenge is ecosystem-based fisheries management (EBFM), which takes into account species interactions, feedbacks, and trade-offs within fished systems (FAO, 2003; Stefansson et al., 2019; Townsend et al., 2019).

Despite the growing popularity of EBFM, the actual adoption and implementation of the approach have remained slow globally (Pitcher et al., 2009). Many factors have been suggested as leading to the slow adoption of EBFM, but mainly the lack of inclusion of all stakeholders in the implementation process, leading to misunderstandings about the approach (Patrick and Link, 2015; Trochta et al., 2018). However, EBFM may also be hampered by the conflicting nature of management objectives. Previous studies have shown, for example, that total ecosystem yield may not be maximized without depleting some stocks and that protecting all exploited species from overfishing would require reducing fishing effort and catch (and hence employment) in every fishing sector/fleet (Hilborn et al., 2004; Hilborn, 2007a, b, 2010;

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Sparholt and Cook, 2010; Pascoe et al., 2013; Andersen et al., 2015; Forrest et al., 2015). Whereas these conclusions apply to many exploited fisheries across the globe, trade-offs can also be influenced by local conditions such as productivity (the rate of natural population increase) of the individual stocks and strength of species interactions (Voss et al., 2014; Shelton et al., 2014) and the socio-economic setup of the fishing communities (Bene, 2009). Additional investigations, therefore, are needed on how to achieve a balance between fish production, profits from fishing, employment, and conservation objectives at the local scale.

Analysis of trade-offs between management objectives at the local scale is particularly important for small scale inland fisheries because of the level of dependence of riparian communities on these fisheries for livelihoods. These fisheries are generally open access, with fishing providing the only source of employment and livelihood for the riparian communities (Cooke et al., 2016). In these fisheries, most fishers (boat crew) are underprivileged, and illegal fishing activities are rampant; also, the least productive species have been overexploited, and the stocks need rebuilding (Bene, 2009; FAO, 2018). However, reducing fishing effort and catch to levels appropriate for rebuilding the declining stocks may impose a high social cost, considering that most of these fisheries are in the underdeveloped nations that generally lack social benefit systems and alternative livelihoods (Bene et al., 2010). Managers of these fisheries need information on the potential risks associated with maintaining high fishing effort and catch against the long-term benefits associated with smaller, more profitable catch, but at the expense of livelihoods and employment.

1.1. Ecosystem models

Ecosystem models are important decision support tools (DST) for EBFM (Collie et al., 2016; Stefansson et al., 2019; Townsend et al., 2019). Many ecosystem modelling tools exist globally, but the most widely used are Ecopath with Ecosim (EwE, Christensen and Walters, 2004a), with over 400 published models (Colleter et al., 2015), and Atlantis (Fulton et al., 2011), which has been applied in all continents across the globe (CSIRO, 2020). These tools have also been used in many socio-ecological studies involving trade-offs between fisheries management objectives (Christensen and Walters, 2004b; Araujo et al., 2008; Cheung and Sumaila, 2008; Forrest et al., 2015).

Ecosystem models, however, often give contrasting results, although qualitative similarities have been observed in some comparative studies (Forrest et al., 2015; Natugonza et al., 2019; Pope et al., 2019). This uncertainty is likely the main reason hampering the use of ecosystem models directly in fisheries management decision making and EBFM (Link et al., 2012; Hyder et al., 2015; Collie et al., 2016). Consequently, presenting model outputs with a realistic cognizance of their uncertainty has become a requirement for their use in EBFM (Grüss et al., 2017).

Model uncertainty takes multiple forms, including natural variability, observation error, inadequate communication (especially among scientists, decision-makers and stakeholders), outcome uncertainty, unclear or nonspecific management objectives, or structural complexity of the model(s) (Link et al., 2012). These forms of uncertainty enter the modelling process at different stages, making it impossible to address all forms by a single technique. In models that are developed for exploring fishery policy scenarios, more emphasis is put on structural uncertainty, arising from parameter estimates (parametric uncertainty), model framework, functional form, spatial/temporal scaling, or complexity trade-off (Link et al., 2012). For parametric uncertainty, sensitivity analysis (Klepper, 1997; Fennel et al., 2001; Essington, 2007; Wu et al., 2014; Steenbeek et al., 2018), model fitting to time series data (Heymans et al., 2016; Grüss et al., 2017), and skill assessment (Stow et al., 2009; Olsen et al., 2016) are recommended to show how the model deviates from reality. However, due to the large number of parameters involved in some complex end-to-end models, such as Atlantis, full-scale sensitivity analysis is computationally prohibitive (Ortega-Cisneros et al., 2017; Morzaria-Luna et al., 2018; Sturludottir et al., 2018). Consequently, fitting the model to time series data and optimising prediction skill remain the most widely used approaches for addressing uncertain parameters in complex ecosystem models (Grüss et al., 2017). For the remaining four subcategories of structural uncertainty (model framework, functional form, spatial/temporal scaling, and complexity trade-off), more emphasis has been put on ensemble/structured modelling, involving multimodel inference (Espinoza-Tenorio et al., 2012; Gårdmark et al., 2013). The rationale is that sound management decisions can be supported if multiple ecosystem models, despite their different assumptions, lead to consistent and converging results (Fulton and Smith, 2004). Recent studies show that this approach has potential to offer some ‘insurance’ against the uncertainty that comes with modelling complex ecosystem-level processes (Pope et al., 2019; Bauer et al., 2019), but more comparative assessments of various ecosystem models are needed, especially for inland lakes.

1.2. The case of Lake Victoria (East Africa)

Lake Victoria is the largest tropical freshwater lake in terms of surface area (about 68,800 km^2) and is shared by three countries (Uganda, Kenya, and Tanzania). The catchment area has one of the highest population densities in Africa, approaching 500 people km^{-2} (Kolding et al., 2014). This population has put pressure on fishery resources, with indications that the fishery is experiencing effects of “Malthusian overfishing” (Pauly, 1990, 1994; Teh and Sumaila, 2007). The typical fishers (boat crew) around Lake Victoria have a poor financial savings culture, and, except for the few islands with agricultural activities, the only source of employment and livelihood for the resident communities is fishing (Nunan, 2010; Johnson and Bakaaki, 2016). While the overall landings from the commercial species, for example, Nile perch (*Lates niloticus*), Nile tilapia (*Oreochromis niloticus*), and the silver cyprinid (*Rastrineobola argentea*), locally called dagaa, have not plummeted, the catch rates (catch per unit effort, CPUE) for all the species have declined (LVFO, 2016a). Consequently, the use of illegal fishing gears and techniques, notably the small gillnets less than 5 in. (stretched mesh size), beach seine and monofilament gillnets, has increased during the past decade (LVFO, 2015; Mpomwenda, 2018). The main focus of the current Lake Victoria Fisheries Management plan (LVFO, 2016b) is to halt and/or reverse this trend. However, the social cost associated with reducing fishing effort and rebuilding fisheries remains unknown.

Lake Victoria is generally open-access, with limited output restrictions. Approximately one million tonnes of fish are harvested from the lake each year; this catch is worth the US $600 − 850 million annually from the direct sales of fish at the landing sites (LVFO, 2016a). In terms of employment, the fishery directly employs about one million people through fishing and value chain activities; with the inclusion of their dependents, the lake can be seen to support the livelihoods of approximately four million people (Mkumbo and Marshall, 2015). The decline in catch rates over the years suggests that the stocks are exhibiting the effects of intensive fishing (Mkumbo and Marshall, 2015; LVFO, 2016b) and that these multiple benefits (high catch and employment) may not be sustained for a long time into the future.

Recently, the authorities on the Ugandan side of Lake Victoria have adopted a management objective aimed at maximizing net economic returns, through limiting access, as the primary management objective (Johnson and Bakaaki, 2016; Glaser, 2018). However, a wealth-dominated approach in small scale fisheries mainly affects the poor, vulnerable fishers, especially those fishing for food in nearshore areas, whose overall impact on the fishery may be insignificant; also, it encourages illegal fishing activities (Bene et al., 2010). On Lake Victoria, these aspects will be exacerbated because the affected groups of fishers have limited livelihood options and the wealth generated for governments may not necessarily be used to create alternative employment for

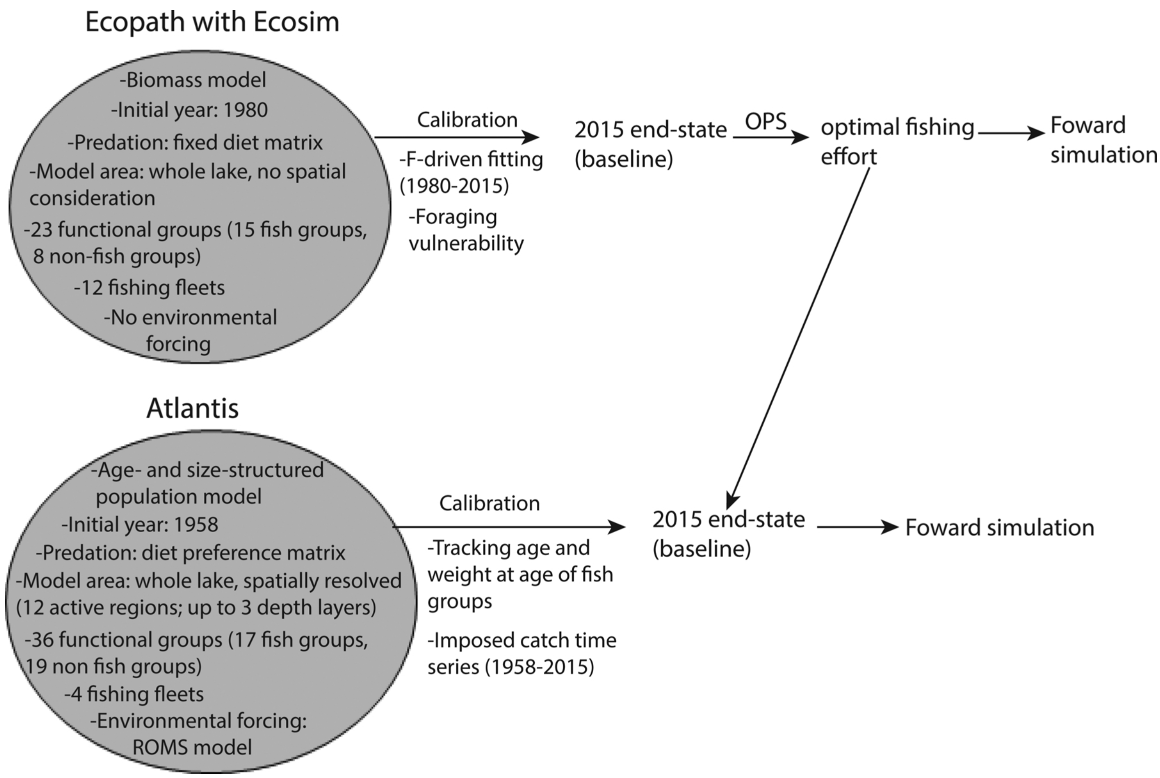

Fig. 1. Schematic diagram showing the major features of EwE and Atlantis models of Lake Victoria, and overall study design. F stands for fishing mortality; OPS stands for optimal policy search. Modified from Natugonza et al. (2019).

the affected fishers (Nunan, 2014; Johnson and Bakaaki, 2016). The need for information on how to achieve a balance between fish production, profits from fishing, employment, and conservation objectives is apparent and is the focus of this study.

1.3. Objective of the study

This study aims to explore the trade-offs between socio-economic and conservation objectives in fisheries management using Lake Victoria as a case study. The study involves estimating long-term, gear-specific fishing effort (optimal fishing effort) that can maximize predefined management objectives (economic, social, and conservation benefits) and assessing the response of the ecosystem to each management objective. We hope that by understanding the trade-off relationships between ecological, economic, and social objectives, managers will be in a position to weigh risks and benefits associated with each objective to make informed decisions. To minimise the effect of structural uncertainty associated with multispecies models, we explore trade-offs using two structurally-distinct modelling frameworks, EwE and Atlantis. For parametric uncertainty, although the sensitivity of outputs to different input parameter combinations in both EwE and Atlantis models of Lake Victoria is unknown, both models are fitted to time series data and their predictive skill can be assessed from the associated multiple skill assessment metrics (Nyamweya et al., 2016a,Nyamweya et al., 2016b; Natugonza et al., 2020). The values of the skill metrics (see below) can be used to judge how the models perform against data and any caution that may be applied while interpreting model outputs.

2. Materials and methods

2.1. Modelling software

EwE and Atlantis, the most widely used ecosystem modelling tools globally (Fulton et al., 2011; Colléter et al., 2015) and on the African Great Lakes (Musinguzi et al., 2017), were the main modelling tools in this study. These modelling tools represent food webs, abiotic environment (including climate impacts), and fisheries, but at different scales and varying levels of complexity. EwE is 0-dimensional biomass model, where predation is regulated by explicit diet parameters, through a fixed diet matrix, and foraging vulnerability (Christensen and Walters, 2004a). Atlantis, on the other hand, offers a more detailed and explicit representation of biochemistry, physical processes, and lower trophic level dynamics. The model is age- and size-structured; the spatial domain is resolved in 3 dimensions; predation is regulated by a diet preference matrix, although the actual resulting diet is subject to mouth-gape limitations and prey availability (Audzijonyte et al., 2017a,b). Previous studies have shown that these modelling tools can give consistent qualitative policy evaluations, especially for the fishery-induced effects of the directly targeted species, although quantitative results differ (Smith et al., 2015; Forrest et al., 2015; Pope et al., 2019; Natugonza et al., 2019). The two modelling tools have also been used extensively in studies linking fisheries socio-economics and ecology (e.g., Christensen and Walters, 2004b; Cheung and Sumaila, 2008; Araujo et al., 2008; Ainsworth and Pitcher, 2010; Kaplan et al., 2014).

2.2. The models

The EwE and Atlantis models of Lake Victoria were parameterized independently. The EwE model is described in detail (mass balances, data sources and uncertainty, model fitting to time series data, and skill assessment) in Natugonza et al. (2020). The model run file is freely accessible at https://doi.org/10.6084/m9.figshare.7306820.v6. For quick reference, the basic parameters are given in the Supplementary Table S1. The Atlantis model is described in detail in Nyamweya et al. (2016a,b). The model is also freely accessible at https://doi.org/10.6084/m9.figshare.4036077.v1. Key features highlighting the differences in structure and assumptions between the two models are

Table 1
Functional groups used in the Lake Victoria EwE and Atlantis models.

Species/taxa included	Common name	Occurrence	Habitat	Feeding mode	Atlantis	EwE
Haliaeetus vocifer, *Ceryle rudis,* Cormorants	Fish-eating birds	Native	Domain	Piscivore	Yes	Yes
Crocodylus niloticus	Crocodiles	Native	Domain	Carnivore	Yes	Yes
Lates niloticus	Nile perch	Introduced	Demersal	Piscivore	Yes	Yes
Clarias gariepinus	African catfish	Native	Benthopelagic	Omnivore	Yes	Yes
Bagrus docmak	Semutundu	Native	Benthopelagic	Omnivore	Yes	Yes
Protopterus aethiopicus	Marbled lungfish	Native	Demersal	Molluscivore	Yes	Yes
Synodontis victoriae, S. afrofisheri	Squeakers	Native	Benthopelagic	Insectivore	Yes	Yes
Momyrus kanume, Gnathonemus spp.	Snout fishes	Native	Demersal	Insectivore	Yes	Yes
Schilbe intermedius	Silver catfish	Native	Pelagic	Piscivore	Yes	Yes
Labeobarbus altianalis	Rippon barbell	Native	Benthopelagic	Omnivore	Yes	Yes
Enteromiuss spp.	Small barbs	Native	Benthopelagic	Omnivore	Yes	Yes
Brycinus jacksoni, B. sadleri	Robbers	Native	Pelagic	Omnivore	Yes	Yes
Labeo victorianus	Ningu	Native	Demersal	Phytoplanktivore	Yes	Yes
Haplochromis spp. (Phytoplanktivorous, Benthivorus, and Piscivorous haplochromis)	Haplochromines	Native	Benthopelagic	Variable[a]	3 groups	1 group
Rastrineobola argentea	Silver cyprinid	Native	Pelagic (schooling)	zooplanktivore	Yes	Yes
Oreochromis niloticus	Nile tilapia	Introduced	Benthopelagic	Omnivore	Yes	Yes
O. esculentus and *O. variabilis*	Other tilapias	Native	Benthopelagic	Herbivore	Yes	Yes
Caridina nilotica	Shrimp	Native	Demersal	Detritivore	Yes	Yes
Macroinvertebrates, Benthic filter feeder, Shallow filter feeder, Deep filter feeder Microphtybenthos	Insects and molluscs		Water surface or demersal	Detritivore	5 groups	1 group
Microzooplankton, Mesozooplankton,	Zooplankton		Pelagic	Phytoplanktivore	2 groups	1 group
Macroalgae, Large phytoplankton, Dinoflagellates, Pico-phytoplankton	Phytoplankton		Pelagic		4 groups	1 group
Periphyton, epiphyton	Benthic producers		Domain		No	Yes
Pelagic and sediment bacteria	Bacteria				Yes	No
Labile and refractory detritus	Detritus		Benthic		2groups	1 group

[a] More than 15 trophic groups (Witte and van Densen, 1995).

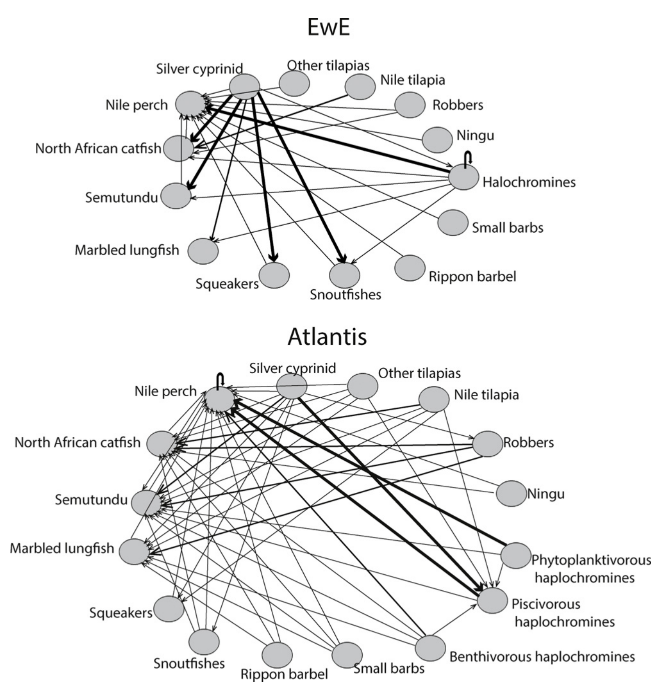

Fig. 2. Schematic representation of predation interactions in EwE and Atlantis models of Lake Victoria. Model groups shown here are only for fish species, which are represented in both models, to ease comparisons. Note that arrows move towards the predators and arrow thickness is consistent with the contribution of prey to the predator's diet. Thick and black arrows indicate that the prey species make up more than 30 % of the predator's diet, while thin arrows indicate that the prey species make up less than 5% of the predator's diet. From Natugonza et al. (2019).

given in Fig. 1. The representation of functional groups differs for the lower trophic levels (TLs, i.e., biomass pools), but the vertebrate groups are comparable, except haplochromines, which are modelled as one group in EwE and three groups in Atlantis (Table 1). Feeding interactions are also fairly comparable; the main differences relate to the strength of the diet dependencies and representation of cannibalism in Nile perch and haplochromines (Fig. 2). Despite these variations, the previous comparisons of these two models using species-specific fishing scenarios (e.g., doubling/halving the baseline fishing mortality of key functional groups: Nile perch and haplochromines) resulted in consistent qualitative predictions (Natugonza et al., 2019). What this study intends to explore is whether the models can give similar results for scenarios that involve changing, simultaneously, the fishing mortality of all fished groups in the system.

Some adjustments were made to the gear set-up in the EwE model to cater for variation in the cost of fishing associated with the fishing area and mode of operation of the fishing gear. The EwE model of Natugonza et al. (2020) considered four gears: gillnets, longlines, small seines, and 'others'. Gillnets target most species except small fishes such as silver cyprinid. Longlines target Nile perch and other demersal and benthopelagic species (Table 1). Small seines target silver cyprinid; freshwater shrimp (*Caridina nilotica*) and haplochromines are by-catch. 'Others' is an aggregation of gears (e.g., beach seines, cast nets, traps), targeting a variety of fish species from shallow inshore regions. However, the mode of operation of the fishing gears used in Lake Victoria differs by region and water depth. In the shallow inshore waters ($<$ 20 m), gillnets and longlines are operated with small to medium-sized paddled canoes, while in coastal (20 − 40 m) and deep ($>$ 40 m) waters (Fig. 3), both gears are operated with large sail or outboard engine-powered Sesse boats (LVFO, 2015). These fishing areas and mode of operation of fishing gear have implications for the fishing costs and net profit (Onyango, 2018), which are inputs in the optimization for economic benefits (see below). These gears were, therefore, separated into 12 distinct fleets, reflecting the actual mode of operation, using information from catch assessment and Frame surveys (Supplementary Table S2).

2.2.1. Model uncertainty

The sensitivity of model outputs to input parameters and data have not been explicitly analysed for both EwE and Atlantis models of Lake Victoria. Instead, model uncertainty has been assessed using proxy

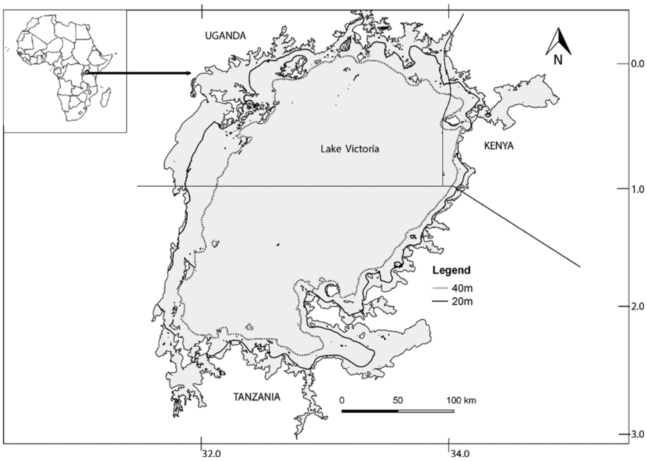

Fig. 3. Location of Lake Victoria in East Africa.

indicators, including pedigree criteria (Christensen and Walters, 2004a), model fit to time series data (Scott et al., 2016; Audzijonyte et al., 2017a), and prediction skill (Stow et al., 2009; Olsen et al., 2016). The pedigree criteria involve estimation of an index, the pedigree index (PI), showing whether the parameters are of low quality (uncertain), i.e., guestimates (PI = 0) or of high quality, i.e., parameters arising from an accurate sampling of the modelled system (PI = 1). Skill assessment involves comparing model predictions with data using multiple metrics that measure both scale mismatch (magnitude of the difference between predictions and observations) and correlation (similarity of trends). The commonly used metrics are modelling efficiency (MEF), reliability index (RI), and coefficient of variation (CV), for scale mismatch, and Pearson correlation (r) for trend comparison (Ortega-Cisneros et al., 2017; Sturludottir et al., 2018).

The PI of Lake Victoria's EwE model is 0.53 (Natugonza et al., 2020). Based on PI values of all EwE models published in EcoBase ($0.1-0.7$), this value is suggestive of intermediate parameter/model quality (Morissette, 2007). Regarding prediction skill, Natugonza et al. (2020) used three metrics: MEF, RI, and r. The values of these metrics were positive and close to 1 for most of the groups, suggesting good fit to the reference data. Exceptions were the 'robbers', haplochromines, and other tilapias (see group definitions in Table 1), where values of MEF were negative; nonetheless, r values for haplochromines and other tilapias were positive, suggesting that the simulated biomass trajectories had the same trend as reference data despite the differences in magnitude. Also, the model simulated diet composition that fairly resembled data from stomach content analyses (Natugonza et al., 2020).

For Atlantis model, Nyamweya et al., 2016a,b) also used multiple metrics (MEF, r, and CV) to evaluate model prediction skill. All metrics were positive; also, for most of the functional group, values were close to 1, suggesting that model predictions match well with data (both in terms of magnitude and trend). Besides, the model simulates distributions of nutrients, primary production, and major fish species that match well with the available data (Hecky et al., 2010), suggesting that the model also captures well the heterogeneous nature of the Lake Victoria ecosystem.

2.3. Study design

EwE includes a fisheries optimization routine, the optimal policy search (OPS), which is used in a calibrated model to search for long-term, gear-specific fishing effort (optimal fishing effort) that can maximize a user-defined management objective (Christensen and Walters, 2004a). Atlantis does not have a similar computationally-flexible procedure. For purposes of model comparison, we used a simpler approach, similar to that used in Forrest et al. (2015), by applying the OPS procedure in EwE, where different weightings and objectives could be easily defined, and then applying the resulting optimal fishing effort levels in both EwE and Atlantis to evaluate long-term changes in ecosystem structure (Fig. 1). Also, EwE and Atlantis differ substantially in the way fleets are modelled. In this study, we aimed at minimizing the variations in predictions due to fleet representation and dynamics in Atlantis (Audzijonyte et al., 2017b) by controlling fishing mortality during projections at the level of functional groups.

2.3.1. Optimal policy search (OPS)

The gear-specific optimal fishing effort for each objective was simulated using the OPS routine of the EwE software version 6.5 (freely available at https://ecopath.org/). The OPS routine uses a calibrated EwE model to probe an n-dimensional parameter space (a response surface) for zones that yield increased benefits, where n is equal to the number of gear types in the model (Christensen and Walters, 2004a). The routine uses the non-linear Davidon-Fletcher-Powell optimization procedure (Fletcher, 1987) to improve an objective function by changing relative fishing rates iteratively. This procedure uses a "conjugate-gradient" method, testing alternative parameter values to approximate the objective function locally as a quadratic function of the parameter values, which are updated stepwise. An optimal fleet-effort solution is generated that maximizes benefits over the simulation time horizon and manipulates the ecosystem into a maximally beneficial form. The benefits are defined by a multi-criterion objective function (OBJ), which, in our study, contained terms representing economic, social, and conservation (ecosystem maturity and biomass diversity) benefits (Eq. 1).

$$OBJ = W_{Econ} \bullet \sum NPV_{ij} + W_{Soc} \bullet \sum Jobs_{jt} + W_{Ecol} \bullet \sum \left(\frac{B}{P}\right)_{it} + W_{Biod} \bullet \sum Q90 \tag{1}$$

where, W_{Econ}, W_{Soc}, W_{Ecol} and W_{Biod} are, respectively, relative weightings applied to economic, social, ecosystem maturity and biomass diversity criteria (see below). The summed terms evaluate the benefits of the harvest plan across each functional group i, gear type j, and simulation time step t; the latter is set by default at monthly intervals. For single-objective optimization, a weight of 1 is placed on either economic, social, or conservation criterion, and zero on all others. For a combined objective (mixed objective optimization), either similar or different weights are applied to the objectives, depending on the societal objectives.

2.3.2. Maximizing economic benefits

The net present value (NPV) is the default metric for assessing the economic benefits of a harvest plan in EwE. The form of discounting is based on the intergenerational discounting (IG) approach (Sumaila, 2004; Sumaila and Walters, 2005). This approach considers a continuous interlacing of generations, where the discounting of future benefits is countered each year by the addition of 1/G stakeholders (Eq. 2; G is generation time = 20 years). These new entrants bring with them a renewed perspective on future earnings, partially resetting the discounting clock.

$$NPV = \begin{cases} \sum_{t=0}^{T} NB_t \left(d^t + \frac{d_{fg} \bullet d^{t-1}}{G} \left[\frac{1-\left(\frac{d_{fg}}{d}\right)^t}{1-\frac{d_{fg}}{d}} \right] \right) & \delta \neq \delta_{fg} \\ \sum_{t=0}^{T} \frac{NB_t}{(1+\delta)^t} \left(1 + \frac{t}{G}\right) & otherwise \end{cases} \tag{2}$$

where, NB is net benefits (gross income minus operating costs) accruing in year t; d is a standard annual discount factor ($d = 1/(1+\delta)$; is the annual discount rate; d_{fg} is a discount factor to evaluate benefits destined for future generations fg ($d_{fg} = 1/(1+\delta_{fg})$). With the IG approach, future benefits are less discounted and higher NPV is assigned to

Table 2
Landed value, total cost, and profitability of the 12 fishing gears in the Lake Victoria EwE model. These are estimated internally within EwE using input data on catch value for each species/functional group (US dollar/unit biomass, Supplementary Table S3) and the proportion of fixed and operational variable costs (Supplementary Table S4). Values in parentheses are used in the “penalty scenarios”, where the cost of fishing using illegal gears is increased until the profit from fishing is zero.

Gear	Landed value (US $ thousand km^{-2})	Total cost (US$ thousand km^{-2})	Profitability (%)
Longline (motorised)	1.52	1.04	34.0
Longline (paddled)	0.28	0.19	31.0
Gillnet (motorised)	1.43	0.83	42.0
Gillnet (paddled)	0.35	0.28	19.5
Small seine (motorised)	2.22	1.35	40.0
Small seine (paddled)	0.88	0.50	43.0
Beach seine	2.52	1.01 (2.52)	60.0 (0.0)
Monofilament (motorised)	0.64	0.22 (0.64)	64.0 (0.0)
Monofilament (paddled)	1.57	0.85 (1.57)	45.6 (0.0)
Cast net	0.19	0.07 (0.19)	62.5 (0.0)
Hand line	0.24	0.12	50.6
Scoop net	0.12	0.08 (0.12)	31.5 (0.0)

policies that spread out benefits over a long time (Sumaila, 2004).

Data are needed on the catch value for each species/functional group (Supplementary Table S3) and profitability for each gear (Supplementary Table S4). All optimizations were run with the default discount rate of 0.04/year; we also set the optimizations to the mode that allows the gears to operate under economic loss. The rationale is that although a gear can operate under unprofitable conditions, the sum of profits across all gears can compensate for those losses suffered by some gears (Christensen and Walters, 2004b)

2.3.3. Maximizing social benefits (employment)

EwE contains only simplified fishing fleet-dynamics, where fishery yield is used as a proxy for employment. Benefits are, therefore, assessed as the total number of jobs directly produced by the harvest plan, summed across each gear type and simulation year. The number of jobs is calculated as the sum product of catch value (calculated internally in Ecosim, Table 2) and jobs per-unit-catch-value (as input on the policy search form). Jobs-per-unit-catch-value for all gear types were set at 1, so that total employment was proportional to catch value (Christensen and Walters, 2004b; Ainsworth and Pitcher, 2010).

2.3.4. Maximizing ecosystem structure

The objective function maximizes the overall biomass to production ratio (B/P) by summing B/P values across functional groups and simulation years throughout the model and is based on one of Odum (1969) measures of ecosystem maturity. Ratios of production to biomass (P/B ≈ Z, year^{-1}) are available for all functional groupings as part of the standard Ecopath parameters (ESM Table S1). The inverse ratio, B/P, expresses average longevity (unit year), which is used as a biomass weighting factor for the optimization of “ecosystem structure”. Increasing this index leads to an increase in the biomass of long-lived species; hence, the index is used as an indicator of ecosystem health and structure (Christensen, 1995).

2.3.5. Maximizing biomass diversity

The objective function maximizes the Q90 biodiversity index of Ainsworth and Pitcher (2006). Under this criterion, biodiversity refers to organismal diversity at the level of ‘species’ functional groups. The Q90 index is a variant on Kempton’s Q index (Kempton and Taylor, 1976), and represents the interdecile slope of the cumulative species log-abundance curve (Eq. 3). Each functional group in the EwE model represents one ‘species’, and the functional group biomass substitutes for abundance (i.e., biomass serves as a proxy for the number of individuals in that species).

$$Q90 = \frac{0.8 \cdot S}{\log(\frac{R_2}{R_1})} \tag{3}$$

where, S is the total number of functional groups in the model, R_1 and R_2 are representative biomass values of the 10th and 90th percentiles in the cumulative abundance distribution (Eqs. 4 and 5).

$$\sum_{1}^{R_1-1} n_R < 0.1 \cdot S \le \sum_{1}^{R_1} n_R \tag{4}$$

$$\sum_{1}^{R_2-1} n_R < 0.9 \cdot S \le \sum_{1}^{R_2} n_R \tag{5}$$

where, n_R is the total number of functional groups with abundance R.

2.3.6. Maximizing mixed objectives

The objective function maximizes, simultaneously, the weighted sum of NPV, catch value, B/P, and Q90 biodiversity index. Identifying ideal weights for a combined objective is not straight forward. We started by placing equal weights to all the objectives, i.e., “mixed objective with equal weights (1:1:1:1)” (Table 3). However, this criterion means that the relative improvement in harvest benefits over the baseline is equally weighted. Whereas this criterion can still give useful information on trade-off boundaries, there is no inherent comparability between the objectives (Christensen and Walters, 2004a). This weighting approach mostly affects the conservation criteria because it is easier to increase revenue or jobs than it is to restructure the ecosystem (Cameron Ainsworth, pers. com). Therefore, in addition to the mixed objective with equal weights, we included scenarios that tested, sequentially, the effect of higher weighting (in multiples of 10) on conservation objectives (B/P and Q90 index) (Table 3). This weighing approach on conservation objectives is comparable to those used in previous studies (Buchary et al., 2002; Mackinson, 2002; Zeller and Freire, 2002; Ainsworth and Pitcher, 2010). Also, because of the relative importance of Lake Victoria fisheries to employment and

Table 3
Weighting criteria for different objectives used in the optimal policy search (OPS). Objectives are arranged as NPV:catch value:B/P:Q90.

Objective metric	Weighting criteria	Description
Net present value (NPV)	1:0:0:0	Single objective optimization for economic value
Catch value (jobs)	0:1:0:0	Single objective optimization for social value
Biomass/Production (B/P)	0:0:1:0	Single objective optimization for ecosystem structure
Q90 index	0:0:0:1	Single objective optimization for biomass diversity
Mixed objective (NPV + B/P)	1:0:1:0	Combined optimization for economic value and ecosystem structure
Mixed objective (equal weighting)	1:1:1:1	Combined optimization for all objectives, weighted equally
Mixed objective (10•B/P + 10•Q90)	1:1:10:10	Combined optimization for all objectives; weight on B/P and Q90 index 10× higher than others
Mixed objective (100•B/P + 100•Q90)	1:1:100:100	Combined optimization for all objectives; weight on B/P and Q90 index 100× higher than others
Mixed objective (10•catch value)	1:10:1:1	Combined optimization for all objectives; weight on employment 10× higher than others
Mixed objective (100•catch value)	1:100:1:1	Combined optimization for all objectives; weight on employment 100× higher than others

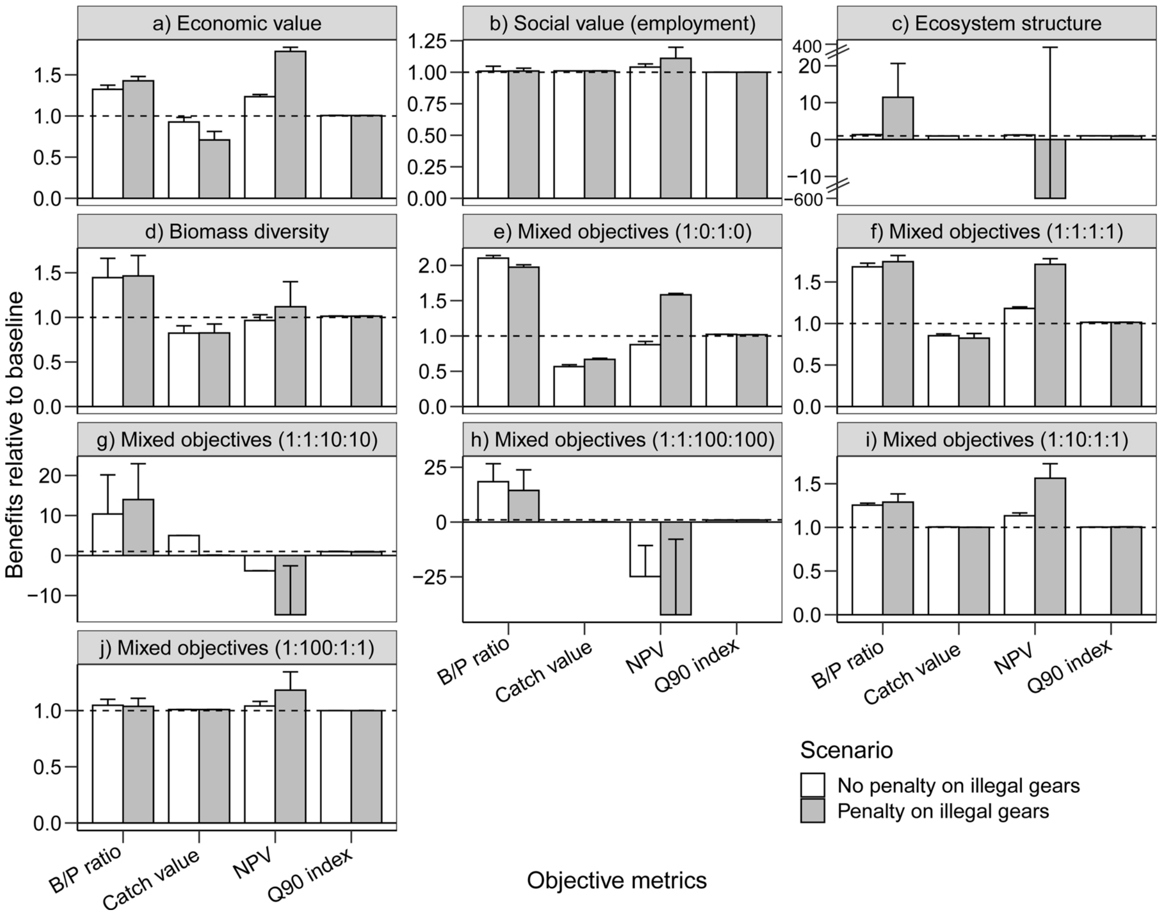

Fig. 4. Results from single-objective optimizations for economic, social and conservation benefits. Values are expressed relative to baseline values from Ecopath model; values below and above the dotted lines show decrease and increase from the baseline, respectively. NPV stands for net present value, while B/P is biomass to production ratio.

livelihoods (Johnson and Bakaaki, 2016), we tested the effect of higher weighting on social objectives (jobs) using the same approach as conservation objectives, i.e., using weights 10x and 100x higher on catch value than other objectives (Table 3).

2.3.7. Illegal gears

Gears that are banned from the lake (e.g., beach seine, monofilament gillnet, cast net) are the most profitable (Supplementary Table S4). These gears target high-value fisheries, particularly Nile perch and Nile tilapia (Luomba et al., 2018; Onyango, 2018). In the Kenyan part of the lake, for instance, the average catch per fisherman per day using beach seine is about 70 kg (equivalent to KES 5,487,680 or US$ 54,800 per fisherman per year) compared to 42 kg (equivalent to KES 1,394,722 or US$ 13,940 per fisherman per year) and 18 kg (equivalent to KES 3,024,756 or US$ 30,247 per fisherman per year) using the allowed longlines and gillnets, respectively (Onyango, 2018). The cost of fishing (both fixed costs and operational variable costs) are also lowest for beach seines compared the other gears.

Christensen and Walters (2004b) showed that it is not always the most profitable gear or the gear with the highest landed value that has its fishing effort increased after optimization. However, results from most case studies (e.g., Araújo et al., 2008; Cheung and Sumaila, 2008; Heymans et al., 2009) show that the optimization always decreases fishing effort for costly gears and increases fishing effort for more profitable gears, especially those targeting high-value fisheries. In the case of Lake Victoria, the optimization would likely increase the fishing effort of illegal gears, whose profitability ranges between 50 % and 65 %, and decrease fishing effort for the legal gears, where profitability is two times lower than the illegal gears (see Supplementary Table S4). To test this, we added a penalty to all the illegal gears by adjusting input data in Supplementary Table S4 such that the profitability of the illegal gears was zero (see Table 2). This scenario is analogous to fishing at a bioeconomic equilibrium (BE) point, where all the revenue is spent to cover the costs of fishing (Walters et al., 2005).

2.4. Simulation procedure and visualisation of outputs

Non-linear optimization methods can converge to local maximum (Fletcher, 1987). To avoid local maximum solutions, we began all optimizations from a random location on the response surface, where fishing mortality exerted by each gear type is randomly set, and repeated the optimization for each objective ten times. Results are presented as mean change in harvest benefits from the baseline and standard deviations, showing the range of all possible values.

After the OPS, both EwE and Atlantis were projected forward using fishing effort levels obtained for each objective (Supplementary Tables S5 and S6). Projections in Atlantis were conducted by changing the 2015 “end-state” fishing mortality (baseline) for each exploited fish group (Supplementary Tables S5 and S6) such that the relative change (either increase or decrease) in fishing effort from the 2015 baseline was proportional to the change in fishing effort for the same group in EwE after optimization.

EwE and Atlantis generate extensive outputs. To keep comparisons between models manageable, biomass predictions were aggregated into annual trends without spatial and size- and age-structure considerations. Changes in biomass of individual fish groups were analysed at the end of 20 years relative to the 2015 baseline values (Natugonza et al., 2019) according to the formula: $(B_{end}/B_{start}) - 1$, where B_{start} and B_{end} are biomass values at the start and end of the projection, respectively. A value of zero indicated no change in biomass relative to baseline. The criteria used in the interpretation of results is similar to that in Natugonza et al. (2019): models gave consistent qualitative results if the direction of change in biomass was the same; consistent quantitative results were indicated by predictions with similar direction and magnitude.

3. Results

3.1. Objective optimisations

When optimizing for economic value alone, an improvement was observed in economic benefits and ecosystem structure only, in the order of 30–80 %, and the benefits were seen to be higher when a penalty was applied to illegal gears compared to scenarios with no penalty on illegal gears (Fig. 4a). This objective showed that social benefits would reduce by 10–25 % from the baseline; the reduction would be highest where illegal gears are penalised. This trend was the same across single objective optimization for ecosystem structure and biomass diversity (Fig. 4c–d), and in all mixed objective optimisations, except where employment was given a higher weight than other objectives (Fig. 4e–j).

Single objective optimization for employment showed no change in all objective metrics, except for economic value, which slightly increased relative to the baseline (Fig. 4b). The increase in economic value was also slightly higher under the penalty scenario compared to the no-penalty scenario. This trend was generally the same across mixed objective optimizations with a higher weight on social criterion (Fig. 4i–j), except in the model run where the weight on the social objective was 10 times higher compared to other objectives. In this model run, results were comparable to those from single-objective optimization for economic value, except that there was no reduction in catch value from the baseline.

When optimizing for ecosystem structure, the addition of a penalty on illegal gears led to the greatest reduction in economic value (Fig. 4c). This scenario is associated with overfishing of main commercial fisheries, Nile perch and silver cyprinid (see below), as the fishing effort of illegal gears, after imposing a penalty, is redistributed to short-lived species. In the no-penalty scenarios, an improvement in the order of 20–30 % from the baseline was observed for both ecosystem structure and economic value. This trend was the same across mixed objective optimisations with a higher weight on ecosystem structure, except that the economic value decreased, while ecosystem structure improved, in both penalty and no-penalty scenarios (Fig. 4g–h).

3.2. Gear restructuring

Optimization for economic, social, or conservation objectives showed that the fishing gear and fishing effort would need restructuring to achieve a specific goal (Fig. 5). Generally, the fishing effort would be increased in most gears to maximize social benefits, and reduced in most gears to maximize economic and conservation objectives. With a few exceptions, the fishing effort increased in gears targeting high-value fisheries and decreased in gears either targeting low-value fishes (in terms of price per unit biomass) or where the operational costs for using such gears (Supplementary Table S4) were high. When optimizing for economic benefits, for instance, the fishing effort uniformly decreased in the small seine (motorised), scoop net, monofilament gillnet (both paddle and motorised), and multi-filament gillnet (paddle) (Fig. 5a). Except for the latter gear, which targets high-value species (Nile perch and Nile tilapia), the rest of the gear categories, despite being profitable, are dominated by the catches of low-value species (for example, silver cyprinid and haplochromines). The reduction in fishing effort of multi-filament gillnet (paddle) is associated with the high operational costs; note that this the least profitable gears (Supplementary Table S4).

The influence of profitability on gear restructuring was more apparent in the illegal gears when optimizing for economic benefits. The model showed an increase in the fishing effort of beach seine, cast net, and monofilament gillnet when operated without a penalty. However, the fishing effort decreased substantially (i.e., by 80–100 % from the baseline) when a penalty was applied (Fig. 5a). Although fishing effort decreased in monofilament gillnet (paddle) and scoop net, which are also an illegal gear, in both scenarios, the magnitude was two times higher when a penalty was applied compared to the scenario where no penalty was applied.

Optimizations considering conservation benefits (ecosystem structure and biomass diversity) showed a reduction in fishing effort in most of the gears, especially when a penalty was applied to illegal gears (Fig. 5c–d). Exceptions were beach seine, scoop net, small seine longline (motorised), whose effort increased when optimizing for ecosystem structure. The decrease in the fishing effort was more pronounced when optimising for ecosystem structure in combination with economic objective (Fig. 5e–f). Mixed optimisations with a higher weight on ecosystem structure and biomass diversity than other objectives led to mixed results, but generally, the fishing effort increased in gear targeting main commercial fisheries, e.g., gillnet, longline, and beach seine, which target Nile perch, and small seine, which target silver cyprinid (Fig. 5g–h). Mixed objective optimisation with a slightly higher weight (ten) on social criterion produced results that were similar to the single objective optimization for economic value, with fishing effort decreasing in most gear (Fig. 5i). However, when the weight on social criterion was increased 100fold relative to other objectives, results were comparable to single-objective optimisation for social value, with fishing effort increasing in most gears, especially those targeting high-value fisheries (Fig. 5j).

3.3. Species-level trade-offs

The two ecosystem models generally gave contrasting biomass predictions, especially when no penalty was applied to the illegal gears (Fig. 6). The exceptions were the species/groups with strong diet interactions (e.g., Nile perch and haplochromines) and other demersal and benthopelagic groups (e.g., North African catfish, semutundu, Ningu, other tilapias, Rippon barbel, silver catfish, and squeakers), where biomass trends (direction of change) were relatively consistent, despite the differences in the magnitude of predictions. For these demersal and benthopelagic groups, biomass decreased in most optimisations, but the decrease was more pronounced in EwE than in Atlantis. For Nile perch and haplochromines, single objective optimizations resulted in about 25 % increase and 25–50 % reduction in biomass, respectively, in EwE, while Atlantis showed either small ($< 5\%$) or no change in biomass of these groups relative to the baseline (Fig. 6). However, with a few exceptions, especially where a higher weight was added to the social criterion, mixed objective optimizations resulted in consistent biomass trends for Nile perch, which increased, and haplochromines, which decreased, across models. For example, mixed optimisations with a higher weight on economic and conservation objectives resulted in $1-2$fold increase biomass of Nile perch, which ultimately collapsed the haplochromines. For the two other commercial fisheries (silver cyprinid and Nile tilapia), all the optimizations were associated with an increase in biomass in EwE and either a decrease (for silver cyprinid) or no change (for Nile tilapia) in Atlantis. Most of the native species in Atlantis were almost collapsed at the end of the historical reconstruction simulation, making them unresponsive to changes in fishing effort, except in scenarios where changes were substantially lower than baseline effort (Nyamweya et al., 2017).

When a penalty was applied to the illegal gears, biomass predictions

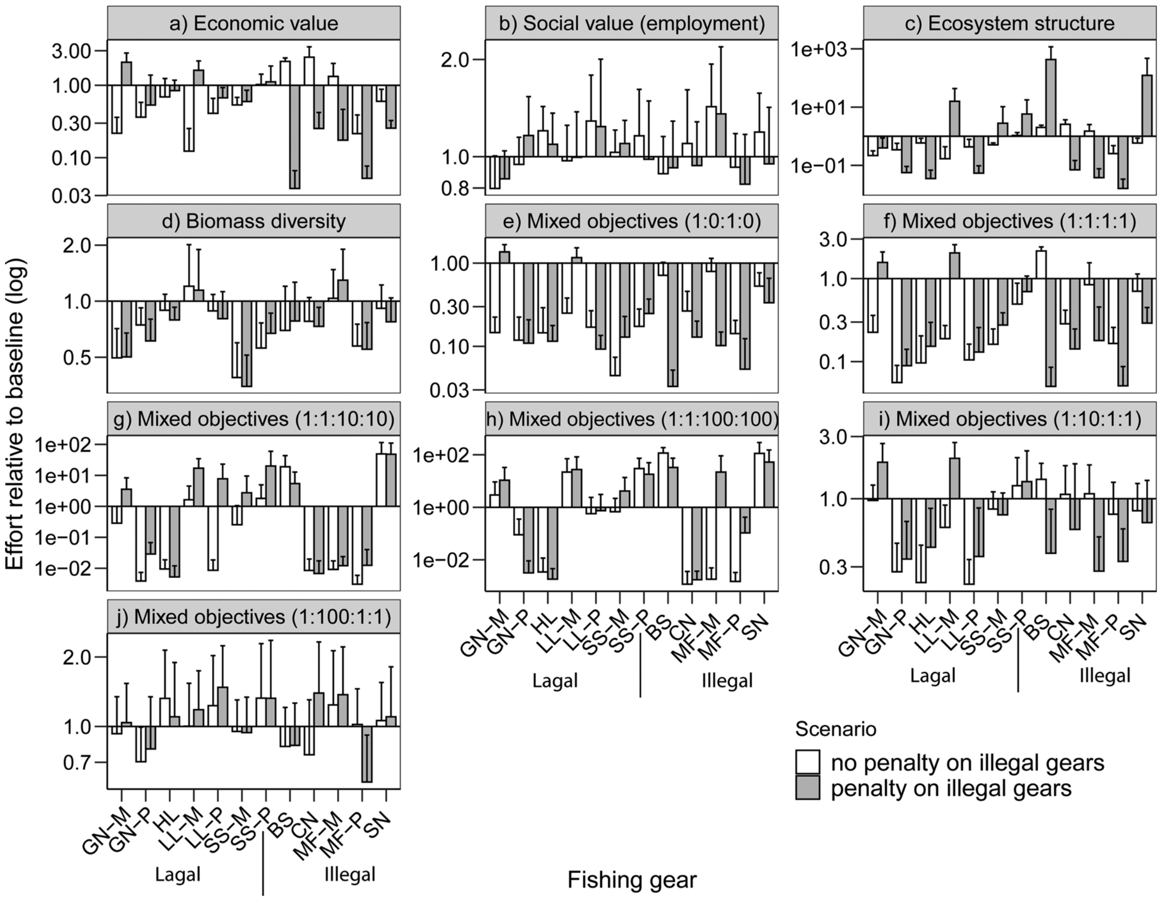

Fig. 5. Gear effort configurations required to maximize economic, social and conservation benefits. All values are expressed relative to the Ecopath baseline run; values below and above 1 show a decrease and increase in effort from the baseline, respectively. Error bars indicate the range of values obtained from 10 runs for each criterion.

for Nile perch and haplochromines were consistent across models even in single-objective optimisation (Fig. 7). Both models showed that maximizing economic value would result in an approximately 50 % increase in the biomass of Nile perch relative to the baseline, which is two times higher compared to the no-penalty scenario, and a corresponding decrease in the biomass of haplochromines, although the magnitude of change was two times higher in EwE than Atlantis. Optimization for social benefits led to variable results, but generally, biomass decreased for most fish groups. Optimization for ecosystem structure led to the largest change in the main commercial fisheries, resulting in a collapse of Nile perch and silver cyprinid in both EwE and Atlantis. This objective was associated with the recovery of most demersal and benthopelagic groups (marbled lungfish, North African catfish, semutundu, Rippon barbell, silver catfish, Robbers, squeakers, and snout fishes), especially in EwE (Fig. 7), majority of which were in a collapsed state at the start of the simulation (Natugonza et al., 2020). Optimization for biomass diversity also resulted in a consistent increase and decrease in the biomass of Nile perch and haplochromines, respectively, in both models. The trends for other major fisheries, such as silver cyprinid and Nile tilapia, however, were different, with biomass generally increasing in EwE and decreasing in Atlantis. Mixed objective optimization, especially those emphasizing economic value and ecosystem structure, resulted in biomass changes comparable to those from single-objective optimization for economic value.

4. Discussion

We analysed trade-offs between management objectives for a fishery with limited alternative livelihood options. The study relied on two structurally-distinct ecosystem modelling frameworks, EwE and Atlantis. Illegal gears were penalised in one of the scenarios by increasing fishing costs until fishing was no longer profitable. This scenario was based on the assumption that in the real world, managers can intensify enforcement and increase fines on the impounded illegal gear to a level where fishers make no profit from fishing illegal gears (Mboya, 2013). Results showed profit maximization to be more compatible with conservation objectives than is the maximization of catch value (or employment). Penalising illegal gears resulted in the highest economic benefits and biomass of main commercial fisheries, but not when the objective was to exclusively maximize ecosystem structure. However, to maximize economic value and maintain ecosystem structure, fishing effort (and potential yield) would be reduced in almost every fishing sector/fleet. These findings are consistent with those from both empirical and modelling studies that have compared actual and potential outcomes of fisheries management objectives (e.g., Christensen and Walters, 2004b; Hilborn et al., 2004; Araujo et al., 2008; Cheung and Sumaila, 2008; Heymans et al., 2009; Hilborn, 2010; Ainsworth and Pitcher, 2010; Andersen et al., 2015; Forrest et al., 2015).

4.1. Trade-offs among objectives

Fisheries managers are more often interested in the stock status and fishing rates that can maintain harvested populations at or above target biomass levels, while avoiding overfishing thresholds (Smith et al., 2007). However, the work presented here is also important as it helps to gain valuable insight into the risks and benefits associated with alternative policies. This study has revealed that trade-offs exist between

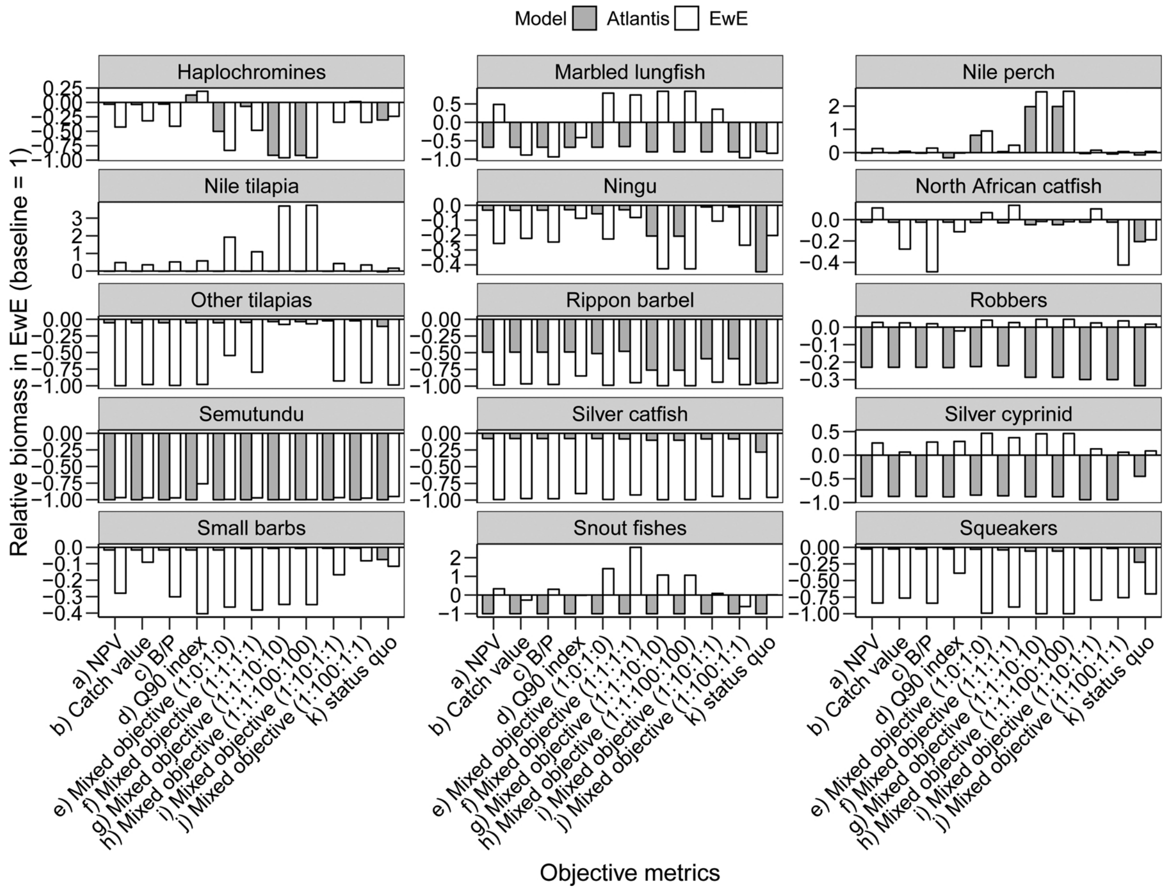

Fig. 6. Biomass change (2034 relative to 2014 baseline) for the different functional groups as predicted by EwE and Atlantis when no penalty is applied to the illegal gears during the optimal fishing policy search. Bars below and above the zero lines indicate a decrease and increase in biomass from the baseline, respectively. Bars on the same side of the zero lines (either positive or negative) indicate consistent qualitative results across models. Results of the 2015 status quo scenario are included for comparison because the ecosystem would be expected to change under any level of fishing, including the baseline fishing rates.

socio-economic and conservation objectives and that the magnitude and severity of the trade-offs vary, depending on the weight given to each objective. For example, a higher weight on employment would possibly collapse the main commercial fisheries in the long term due to the associated increase in fishing effort of most gears. A higher weight on the conservation objective would lead to little, more profitable catch due to the reduction in fishing effort of most gears, but at the expense of the livelihood of riparian communities. These findings underscore the importance of i) identifying management objectives before choosing fleet capacity, ii) considering trade-offs when choosing between multiple ecosystem-level goals, and iii) stakeholder involvement in defining management objectives and evaluating trade-offs.

The benefits from conservation-oriented objectives are known: the risk of depletion is reduced, while biodiversity and ecosystem maturity are improved by protecting the intrinsically vulnerable ecological groups (Christensen, 1998). For Lake Victoria, model outputs show that achieving this objective would require a reduction in fishing effort of gears targeting mostly the demersal groups (marbled lungfish, Ningu, most catfishes and snout fish), especially when a penalty is applied to illegal gears. These fisheries are less dominant in catches, making up less than 15 % of the total landings, and yet the objective still requires a reduction in catch. The trade-off here seems to emanate from the mixed nature of the fishery, where one single gear harvests more than one fish group. The gears that harvest the less productive demersal species (e.g., gillnet, longline, and handline), for instance, are the same for the highly productive and high catch-value species (Nile perch and Nile tilapia). The catches of highly productive fisheries would, therefore, be sacrificed to protect the vulnerable groups, which has been a hindrance to single-stock MSY targets in multi-species fisheries (Walters et al., 2005).

The reduction in catches of Nile perch and Nile tilapia when maximizing conservation objectives, however, does not affect the net economic value. Instead, the two objectives (economic value and ecosystem structure) can be achieved concurrently by fishing less. This result is in agreement with the principle of “pretty good yield” Hilborn (2010). In practice, however, fishing of haplochromines, the main prey for Nile perch, would need to be reduced to avoid drastic fluctuations when the fishing pressure on Nile perch is reduced (Nyamweya et al., 2017; Natugonza et al., 2020).

Nile perch supports the largest commercial fishery in Lake Victoria in terms of catch value. By 2015, Nile perch catches constituted only 18 % of the total annual fish landings (LVFO, 2016a). However, Nile perch landings were worth the US$ 307 million, i.e., 52 % of the total revenues from the entire fishery, from direct sales at landing sites (LVFO, 2016a). This species is also an important export commodity, with exports mainly to Europe worth the US$ 300 million per year (LVFO, 2016b). The trade-off associated with maximizing ecosystem structure, involving the collapse of Nile perch stock, especially when a penalty is applied to the illegal gears, seems to be severe and would have far-reaching socio-economic implications (Johnson and Bakaaki, 2016).

The prevailing socio-economic set-up around Lake Victoria, where most fishers survive on daily fishing and constantly migrate to new fishing villages when catches decline (Nunan, 2010; Johnson and Bakaaki, 2016), demand that employment creation is one of the objectives for management. Between 2000 and 2014, the total number of fishers increased from 60,000 to > 210,000. The total number of fishing crafts using outboard engine increased from 4000 to 21,600

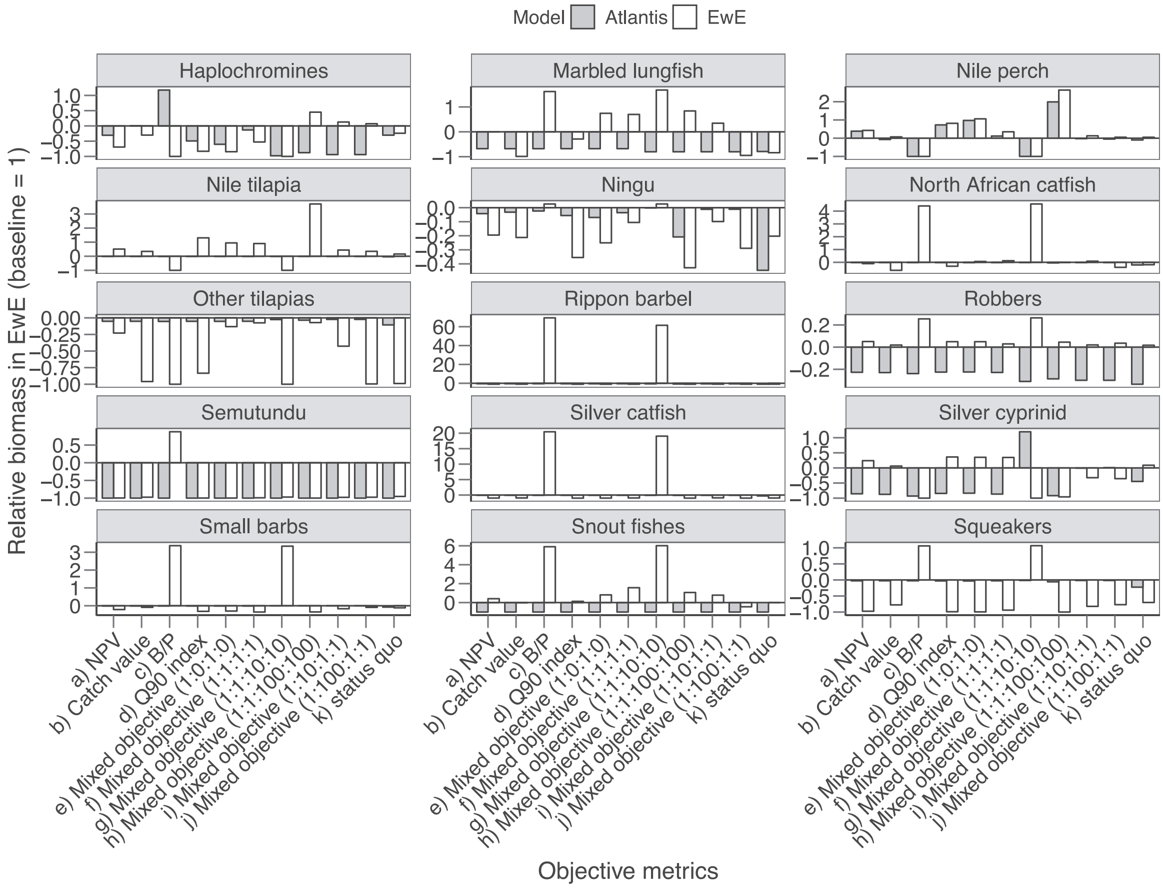

Fig. 7. As for Fig. 6, but a penalty is applied to the illegal gears during the optimal fishing policy search.

boats. While fishing crafts using paddles remained unchanged, the total number of gillnets doubled and the number of longline hooks increased fourfold (i.e., from 3.5 to 14 million) (LVFO, 2015). The increase in the use of outboard engines suggests fishers are extending to deeper offshore waters. In reality, the social objective already has a higher weight. Although a higher weight on social criterion did not negatively affect economic value and ecosystem structure, the continuous increase in fishing capacity still poses a danger of overexploiting fisheries (Njiru et al., 2010; Mkumbo and Marshall, 2015; Mangeni-Sande et al., 2018). The observed decline in catch rates over the past decade may be an indication that fishing capacity has reached the upper limit (LVFO, 2016a).

Whereas economic and conservation objectives can be achieved concurrently, reducing fishing effort in a fishery with more than 200,000 fishers, along a 7142 km shoreline, would not be straightforward. Several attempts have been made in the past, particularly the measures in the Regional Plan of Action on fishing capacity (RPOA-2), which was adopted in 2007 (LVFO, 2007). However, these measures have not been successful as fishing capacity continued to soar (Mulanda et al., 2020). This continuous increase in fishing capacity has been attributed to the limited resources for effective policing (Mkumbo and Marshal, 2015; Mulanda et al., 2020), but this trend could also be related to the social cost of effectively curtailing fishing effort. On the Ugandan side of Lake Victoria, authorities have replaced beach management units (BMUs) with the military to restrict access and eliminate illegal gears (Glaser, 2018). The intervention seems to be working as seen from the improvement in harvested sizes of Nile perch (Taabu-Munyaho pers. comm.), but it could be short-lived. Elsewhere, studies have shown that a sole policing approach, without an alternative livelihood program, is never effective for fisheries that are already in a Malthusian overfishing state (Teh and Sumaila, 2007). Therefore, alternative livelihood plans are still needed for the affected fishers on Lake Victoria; otherwise, new versions of illegal fishing may emerge.

Applying a penalty to the illegal gears also has trade-offs, both positive and negative. The model showed that net economic benefits would be higher when a penalty is applied to illegal gear than a scenario where the profitability of illegal gears is unchecked. The assumption is that fishers do not continue fishing at a bioeconomic equilibrium point. The increase in net economic value is associated with the reduction in effort of beach seine, monofilament gillnets (motorised) and cast net, which releases pressure on Nile perch and its prey (haplochromines) and Nile tilapia. However, when the objective is set to maximize ecosystem structure, the penalty on illegal gears leads to the largest reduction in net economic and social benefits. The penalty causes the effort of longline (motorised) and beach seine (both targeting Nile perch) and small seine (targeting silver cyprinid) to be increased, while attempting to maximize biomass of long-lived species, which drives the two groups to extinction. The implications of the outcome for management are two-fold. First, the effort of illegal gear may grow out of proportion, when a penalty is introduced to limit their profitability, since fishers may not be willing to work with less profit. Second, if management succeeds in eliminating illegal gears, the fishing effort should not necessarily be redirected to the legal gears as this can cause overfishing of the new target fishery. Because the primary target species differ in each gear, an unplanned increase in effort of the legal gears could also lead to overfishing of the target species. The current enforcement on the Ugandan side of the lake seems to be focusing more on undersized gears, and not on the excess numbers of the large gear. This strategy needs to be reconsidered because the life-history changes that may arise as evolutionary responses to intensive, size-selective fishing could rapidly and continuously destabilize and degrade the ecosystem (Kuparinen et al., 2016).

Application of these findings in a management context should take cognisance of the assumptions underlying the OPS in EwE as well as study-specific assumptions. First, the formulation of the multi-criterion objective function in EwE (Eq. 1) requires that all fishing sectors/gears cooperate to maximize the overall benefits of the fisheries. In some instances, however, the benefits of some fishing gears can be maximized at the expense of other gears, thereby affecting the resulting fleet-effort configuration, although the overall trade-off relationships are expected to remain valid. Second, when maximizing the social criterion, we assumed jobs-per-unit-catch-value for all gear types in the model to equal 1, so that total employment was proportional to catch value. EwE coarsely handles employment estimates in and provides only a rough estimate of employment benefits. Future investigations should consider using better estimates of employment rates per gear type, which may require surveys and interviews to consider both fisheries and other supporting sectors. Third, in the mixed objective optimisations, we chose a simple benchmark to assign relative weights to the objective metrics; in one scenario, for example, a weight of 1 was assigned to each of the economic, social, and ecological objectives. This weighting scheme, however, means that a scenario that doubles NPV, for example, would be worth the same to the overall objective score as a scenario that doubles the number of jobs or doubles the B/P ratio of the ecosystem. Yet, the units used to express benefits are different and there is no inherent comparability between NPV, catch value, and the ecosystem maturity index. For example, a doubling of the B/P ratio would represent a dramatic change in ecosystem structure, whereas doubling NPV might be as simple as redistributing fishing effort to more profitable gear. To ensure an "even" mix between these objectives, we also ran optimisations using higher weights on ecological and social objectives. However, the choice of the weights was random, and a larger part of trade-off space remains unexplored. In a management context, the parameter space will need to be fully explored, and the ideal weightings to use on these objectives is a matter of societal values.

4.2. Model comparisons

The structural and functional differences between EwE and Atlantis are very large. The models differ in their degree of complexity, incorporation of heterogeneity in lake habitats, physics (temperature, depth, light), and algorithms of biomass elaboration and feeding (Christensen and Walters, 2004a; Audzijonyte et al., 2017a, b). The calibration process also differs between the models; Atlantis tracks the age structure and weight-at-age of fish groups, whereas calibration in EwE is mainly achieved through adjustment of foraging arena parameters. Nonetheless, these modelling frameworks have been shown to give consistent qualitative policy evaluations (Forrest et al., 2015; Pope et al., 2019; Natugonza et al., 2019). This study has further shown that structurally-distinct ecosystem models can provide consistent qualitative advice for strategic management, which underscores the importance of structured modelling in addressing uncertainty in complex ecosystem models (Espinoza-Tenorio et al., 2012). The observed differences in qualitative results can be attributed to the modelling approach, where the optimal fishing effort resulting from EwE's OPS may not necessarily be optimal in Atlantis.

While the predicted biomass trends were relatively comparable across models, quantitative results differed substantially (Figs. 6 and 7). These differences can be attributed to both model structure and biases in individual models by the modeller. For example, Forrest et al. (2015) found the dynamics of several functional groups in Atlantis to be more influenced by bottom-up processes (fluctuations in primary production, driven by the oceanographic components in the physical sub-model) than in EwE, where there were no explicit primary productivity drivers, which contributed to major deviations in the magnitude of predictions between the models. This study also used EwE model of Lake Victoria in its simplest, non-spatial form, which could also be the main reason for the differences in the magnitude of predictions. Both empirical and modelling studies have shown seasonal variation in Lake Victoria's physical processes and spatial heterogeneity in nutrient concentrations, which are all positively correlated with fish species abundance (Hecky et al., 2010; Nyamweya et al., 2016a,b). Future studies may need to compare the models that are standardised as much as possible. Environmental and non-fishing anthropogenic factors can be incorporated in the EwE model using appropriate forcing functions (Christensen et al., 2008), while spatial effects may be considered using the Ecospace routine of EwE (Christensen and Walters, 2004a).

5. Conclusions

This paper analysed socio-ecological trade-offs between management objectives for Lake Victoria. The OPS routine of EwE was used to search for long-term, gear-specific fishing effort (optimal fishing effort) that can maximize benefits from a defined management objective. The optimal fishing effort was then used in both EwE and Atlantis models to predict future changes in the biomass of exploited fish groups. Results showed that economic benefits can be optimised while maintaining ecosystem structure, but the fishing effort (and potential yield) would be reduced in almost every fishing sector/fleet. This trade-off seemed to be severe for a fishery with limited alternative livelihood options. The prevailing socio-economic set-up around Lake Victoria, where approximately 1000 new fishers enter the fishery per year (LVFO, 2015), demands that employment creation is one of the priorities for management. However, since the lake is already in a state characteristic of "Malthusian overfishing", where reduction of fishing effort is inevitable, planning and developing alternative livelihoods for the fishery-dependent communities will be necessary.

The importance of this study is in weighing the relative risks against the benefits of different management objectives, which will enable stakeholders and the public to conduct informed discussions on future management policies. The study particularly underscores the importance of developing clear policy objectives in consultation with different fishery stakeholders. Stakeholders must decide which of the objectives to prioritise, and what they would be prepared to sacrifice to achieve the desired goal. We have only shown the trade-off boundaries. However, it is unlikely that stakeholders will choose an extreme end that favours, for instance, the ecosystem. More likely, it will be some combination of weights, not necessarily the combinations we have tested in this study. Part of the stakeholder consultation process should involve completely sampling the trade-off space to understand not only the potential costs and benefits, but the nature of the different strategies employed with regards to what fishing sectors are favoured, and what functional groups bear the greatest levels of exploitation.

The optimum fishing effort may vary with different parameter combinations in Ecopath, but the extent of variation has not been determined in this study. The results described in this paper may be interpreted and applied in qualitative terms, i.e., as an aid to discussions on long-term strategic management policies. The EwE and Atlantis models of Lake Victoria were parameterised and calibrated independently, but rigorously using the best available data and following the best practices documented in the literature. The differences in predictions across models do not mean that the models are faulty. Instead, the findings point to areas where different model considerations lead to varying predictions. This modelling approach is necessary as inconsistencies can be incorporated in the policy recommendations.

Authorship statement

VN conceived the idea; VN, CA, CN developed the methodology and created models; ES, GS supervised the investigation process leading to the manuscript; VN, CA, ES, LM, ROO, TT, GS wrote the manuscript; TT provided financial support for the project leading to this publication.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.fishres.2020.105593.

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