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Food for Thought

A critique of the balanced harvesting approach to fishing

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The approach to fisheries termed “balanced harvesting” (BH) calls for fishing across the widest possible range of species, stocks, and sizes in an ecosystem, in proportion to their natural productivity, so that the relative size and species composition is maintained. Such fishing is proposed to result in higher catches with less negative impact on exploited populations and ecosystems. This study examines the models and the empirical evidence put forward in support of BH. It finds that the models used unrealistic settings with regard to life history (peak of cohort biomass at small sizes), response to fishing (strong compensation of fishing mortality by reduced natural mortality), and economics (uniform high cost of fishing and same ex-vessel price for all species and sizes), and that empirical evidence of BH is scarce and questionable. It concludes that evolutionary theory, population dynamics theory, ecosystem models with realistic assumptions and settings, and widespread empirical evidence do not support the BH proposal. Rather, this body of evidence suggests that BH will not help but will hinder the policy changes needed for the rebuilding of ecosystems, healthy fish populations, and sustainable fisheries.

Keywords: balanced harvesting, ecosystem-based fisheries management, population dynamics theory, selectivity, size at first capture, size at maturity.

Introduction

A string of publications has proposed a new approach to fishing, called “balanced harvesting” (BH; [Kolding and van Zwieten, 2011, 2014](#); [Garcia et al., 2012, 2015](#); [Law et al., 2012, 2013, 2014](#)). The new approach is defined as “. . . distributing a moderate mortality from fishing across the widest possible range of species, stocks, and sizes in an ecosystem, in proportion to their natural productivity, so that the relative size and species composition is maintained” ([Garcia et al., 2012](#)). Moderate fishing refers here to fishing mortality rates about equal to the rate of natural mortality at the respective body size ([Caddy and Sharp, 1986](#); [Law et al., 2013](#)). Fishing is

proposed to include all trophic levels above primary producers, starting with organisms of ~1 g body weight (e.g. [Law et al., 2013](#)), corresponding to ~5 cm length in typical fish. Fishing mortality on juveniles is proposed to be compensated by reduced predation mortality. Such fishing is suggested to minimize the impact of exploitation on the structure of ecosystems and to lead to increased yields from larger stocks with less truncated size and age structure ([Law et al., 2013](#); [Kolding and van Zwieten, 2014](#)). Here, we examine these assertions by revisiting the empirical evidence and modelling assumptions put forward in support of BH. First, we try to clarify the meaning of selective fishing in the context of BH.

Then, we address the assertion that size-selective fishing, rather than excessive fishing pressure, has caused the historical decline in biomass and truncation of age structure visible in many commercially exploited populations. Subsequently, the assumptions behind the ecosystem- and size-spectrum models used as evidence of BH are examined. African inland water fisheries are contrasted with the description of these systems put forward in support of BH. Then, the status of real-world ecosystems with exploitation patterns similar to BH is explored. Finally, an assessment is made of who might benefit from implementing BH.

Unselective, selective, and super-selective fishing in proportion to productivity

In promoting the concept of BH, [Garcia et al. \(2012\)](#) compare what they call “selective” fishing with what they call “unselective fishing”. In fact, their definition of “selective” fishing is equivalent to “existing” fishing, and their definition of “unselective” is equivalent to “existing fishing with additional fisheries on lower trophic levels”. The term “selective” in the context of existing fisheries refers to the selection of certain species and size ranges for fishing. In contrast, the use of the term “unselective” in [Garcia et al. \(2012\)](#) refers to the fishing of the widest possible range of species, stocks, and sizes. However, fishing all species and sizes in proportion to their natural productivity would require knowledge of productivity-at-size for all species and a coherent policy of what proportion of productivity shall be fished in a given species at a given size. Even if such knowledge and policy existed, implementation would require a super-selectivity of gears and a regulatory effort far beyond current capabilities. [Garcia et al. \(2015, p. 9\)](#) address these contradictions as follows: “[...] as has been stressed many times, BH is not unselective fishing, but rather selective fishing that is based on productivity rather than value and catch rates. To achieve true BH, gear configurations and mixes of fishing methods may actually have to be much more selective than at present, with engineering challenges that are significant [...]”. In summary, there seems to be some confusion about the meaning of selectivity in the context of BH. In the remainder of this text, we strive to be explicit about what is meant when the term is used.

Disentangling the effects of minimum size limits and excessive fishing on size truncation

Publications in support of BH describe the widely observed truncation of natural age and size structures in exploited populations, i.e. the absence of large fish in many stocks and the tendency for a decrease in maximum size and age at maturation with possible undesirable fisheries-induced genetic selection. These effects are expected from evolutionary theory ([Roff, 1984](#); [Jensen, 1996](#); [Law, 2000, 2007](#); [Charnov et al., 2001](#); [Mangel et al., 2007](#); [Marty et al., 2014](#)), have considerable empirical support ([Heino and Dieckmann, 2008](#); [Conover et al., 2009](#)), and are also predicted from models that assume phenotypic plasticity even in the absence of natural selection effects ([Lester et al., 2014](#)).

Reductions in body size are likely to lead to long-term decreases in yield relative to predictions based on unfished growth and distribution patterns. Age-structured population models predict decreases of ~20–30% in long term yield due to reduced growth ([Zimmermann and Jorgensen, 2015](#)), whereas models based on expected changes in surplus production patterns predict less severe or even positive effects on production along with increased population resilience ([Heino et al., 2013](#)). Typical stock assessments use recent growth information in predicting short-term future

yields, and in doing so already account for the effects of selection in calculation of management targets and reference points.

Publications in support of BH assert that the negative effects associated with size truncation are caused by excessive fishing as well as minimum-size regulations.

- (i) “The outcome of heavy fishing with minimum-size regulations is major disruption to the structure and functioning of aquatic ecosystems” ([Law et al., 2013, p. 161](#)).
- (ii) “Even moderate fishing reduces the proportion of large and old fish in a population. Selectively fishing large individuals amplifies this effect, and although it does not provide the expected yield benefits ([Halliday and Pinhorn, 2002](#)), it results in ecological and evolutionary side effects” ([Garcia et al., 2012, p. 1045](#)).
- (iii) “Targeted removal of large adult fish will result in earlier maturation and decreased adult body size ([Law, 2000](#); [Conover and Munch, 2002](#); [Jørgensen et al., 2007](#); [Sharpe and Hendry, 2009](#); [van Wijk et al., 2013](#)), with unknown evolutionary consequences” ([Kolding and van Zwieten, 2014, p. 133](#)).
- (iv) “[Hsieh et al. \(2010\)](#) concluded from an analysis of many stocks that age and size truncation is widespread in size-selective fisheries” ([Law et al., 2013, p. 163](#)).

No attempt is made in any of these publications to disentangle the effects of fishing mortality and minimum-size limits on the size structure of an exploited population. Instead, minimum-size limits are suggested as the main cause of the observed negative effects (e.g. the last quote above). But, disentangling the effect of fishing mortality and minimum-size limitations on the age and size structure of populations is straight-forward ([Beverton and Holt, 1957](#)). Fish grow throughout their lives and the proportion of large fish in a population is determined by the number of fish that survive to the corresponding age. The number of survivors is a function of the mortality rate. If this rate is increased several-fold, as has been the case in most exploited populations during the past 50 years ([Froese and Proelß, 2010](#); [Pauly and Froese, 2012](#)), then the number of fish reaching old age and large size is drastically reduced. This effect is necessarily stronger if mortality is increased earlier, on smaller life stages. In contrast, with regulations that protect small fish from fishing, many more fish survive to older age and larger size. With moderate fishing of large fish, starting fishing at the size where the biomass of a cohort is maximum, a given catch will have the least impact on size structure and a given fishing mortality will produce close to the highest yields ([Beverton and Holt, 1957, 1966](#); [Froese et al., 2008](#)). This interplay of growth, mortality, size-selectivity, and yield is referred to as *basic population dynamics* in the remainder of this text.

There is widespread empirical evidence in support of the predictions from basic population dynamics. These rules are confirmed by trends in catch, biomass, and recruitment of hundreds of exploited stocks. They form the basis of modern dynamic stock assessment models ([Quinn and Deriso, 1999](#); [Quinn and Collie, 2005](#); [Methot and Wetzel, 2013](#)), which predict next year’s biomass as the basis for setting the next year’s catch quotas. These biomass predictions can be subsequently compared with observations and are, in the most cases, found to be sufficiently accurate to form the regular basis for advice to fisheries management (e.g. [Punt, 2006](#); [Punt et al., 2013](#)). Thus, the statement that “[t]he old single species Yield-per-Recruit models, upon which decades of management

advice, and volumes of simulation studies have been performed are basically not reflecting reality” (Kolding and van Zwieten, 2011, p. 8), is itself not reflecting reality.

For example, the analysis of fish bones found at archeological sites indicates that North Sea cod (*Gadus morhua*) was caught by mediaeval fishers mostly at sizes of 80–120 cm (Harland and Parks, 2008), which corresponds to the predicted size range of the peak in biomass (Froese et al., 2008). This targeted removal of large fish was sustained for hundreds of years, without a declining trend in size (Barrett et al., 1999; Harland and Parks, 2008), indicating a sustainable fishery. After the mid-1960s, fishing mortality was increased to $F > 0.6 \text{ year}^{-1}$, meaning that, on average, over 45% of the fish present during the year were removed by fishing. Also, cod were now targeted from 30–35 cm onwards (ICES, 2015). This fishing led to the collapse of the stock at the end of the 1990s (ICES, 2014a). Had the fishery only targeted fish above 80 cm long, smaller cod would not have been affected, and cod of 80–100 cm would still be present at about half of their pristine abundance. As predicted by basic population theory, but contrary to the predictions of BH (Law et al., 2013), it was the spread of high fishing mortality to all ages and sizes down to 30 cm that most likely led to the observed extirpation of large cod in the North Sea. As the recent stock assessment report for North Sea cod (ICES, 2014a, p. 804) puts it: “There is a need to reduce fishing induced mortality on North Sea cod further, particularly for younger ages, in order to allow more fish to reach maturity and increase the probability of good recruitment”.

As another example, Garcia and Demetropoulos (1986) present detailed evidence of a substantial increase in trawl catch in Cyprus after juveniles entering the fishing grounds were protected for 1 month before fishing was allowed to commence. This confirmation of the predictions of the basic population model was called the “Cyprus effect” and similar protection of juveniles was recommended for pilot projects in other areas (Garcia, 1986).

Another study (Law and Grey, 1989) also provides strong empirical evidence of the predictions of the basic population model: large Arcto-Norwegian cod (*G. morhua*) have been fished sustainably for about 1000 years on their spawning grounds in the Lofoten Islands, contradicting the assertion of Garcia et al. (2012) that even moderate fishing of adults will result in deleterious evolutionary and ecological side effects. A new trawl fishery started on the feeding grounds around 1930, with fishing mortality three times the adult mortality rate exerted on juveniles and adults, i.e. the targeting of only large cod was abandoned and smaller size classes were exploited close to their higher rate of productivity, as called for by BH. As a result, biomass and yield declined strongly over the next decades and maturation shifted to earlier ages (Jørgensen et al., 2009). Such change is predicted by basic population theory and evolutionary theory (see above), but is contrary to the predictions of BH, where the fishing mortality of juveniles should have been partly compensated by strongly reduced predation from cannibalistic adult cod, leading to an overall increase in catch and biomass. Recent management of this stock has strongly reduced fishing mortality and implemented mesh size limitations, a minimum catching size, a maximum bycatch of undersized fish, closure of areas with high densities of juveniles, and other seasonal and area restrictions (ICES, 2014b). As a result of these management measures, and in accordance with basic population theory, current stock size and catches are close to the highest observed, though maturation schedule and size structure have not yet returned to historical values (ICES, 2014b).

Another example of long-term sustainable targeting of large fish is the Almadra trap fishery for Bluefin tuna (*Thunnus thynnus*) in the Mediterranean. For about 3000 years, this fishery has caught large tunas of ~200 kg on their spawning migration, yielding over 100 000 tons in 1950. The subsequent expansion of fishing pressure to other gears, areas, and body sizes down to 30 kg led to the near-collapse of the stock (EC, 2014).

In a final example, the effects of 6 years of strongly reduced fishing pressure in the Northeast Atlantic during World War II also confirmed the expectations from the basic population model, with two- to fivefold increases in stock densities, a shift in biomass towards larger fish, and a slight decrease in weight-at-age probably caused by density-dependent reduction in growth, that was however “insufficient to prevent the augmentation of the stock[s]” (Clark and Graham, 1948, p. 6).

In summary, basic population dynamics theory and empirical evidence leave little doubt that excessive fishing mortality and not minimum-size regulations caused truncation in age and size structure of exploited populations.

Lack of evidence of minimum size as a cause of age truncations and early maturation

In making the case for minimum-size regulations being the main cause of observed size and age truncations in exploited populations, 10 studies are cited in the examined BH publications (Rice and Gislason, 1996; Law, 2000; Conover and Munch, 2002; Halliday and Pinhorn, 2002; Blanchard et al., 2005; Jørgensen et al., 2007; Sharpe and Hendry, 2009; Guénette and Gascuel, 2012; Borrell, 2013; van Wijk et al., 2013). Only two studies (Conover and Munch, 2002; van Wijk et al., 2013) deal with the effects of different size-selectivity. These studies conducted experiments where, over several generations, only the smallest or the largest individuals of a cohort were allowed to reproduce. Not surprisingly, the offspring of the smallest individuals were smaller and matured earlier and the offspring of the largest individuals were larger and matured later than the respective controls with random mating. However, Conover and Munch (2002) used a semelparous species and van Wijk et al. (2013) sacrificed their specimens 30 days after maturation, i.e. there was no adult age structure that could be examined. Also, although both studies tie their results to fisheries management, their experimental harvesting strategies bear no resemblance to fishing. Conover and Munch (2002) harvested 90% of each cohort, letting only the 10% smallest or largest individuals survive and reproduce. van Wijk et al. (2013) harvested 80% of each cohort, letting only the 20% smallest or largest individuals survive and reproduce. Such precisely defined selectivity and massive mortality exerted only on the members of the same cohort in a single day can and should not be reproduced in real-world fisheries. Mortalities of 80–90% inflicted right before first reproduction would lead to a depleted spawning stock with high probability of impaired recruitment (Beddington and Cooke, 1983) and with severely reduced abundance of large fish (Beverton and Holt, 1957; Froese et al., 2008). Therefore, these experiments do not provide any support for the assertion that, in typical age-structured populations, moderate fishing of large fish will result in earlier maturation and decreased adult body size. Rather, as pointed out above, such fishing was sustainable for hundreds of years without evident impacts on ecology or population size structure (Law and Grey, 1989; Barrett et al., 1999; Harland and Parks, 2008; EC, 2014).

The remaining publications cited in support of negative impact of minimum size regulations did not examine the effects of different

size-selectivity and did not contain any evidence that minimum-size regulations caused size truncation or early maturation, or that fishing of all size classes, as proposed by BH, would increase the number of large fish. Rather, most studies identified excessive fishing pressure as reason for observed early maturation. For example, [Sharpe and Hendry \(2009\)](#) found that maturation indices declined “at a rate that was strongly correlated with the intensity of fishing[. . .]”. [Law \(2000\)](#) found that low age-at-entry into the fishery strongly selects for early-maturing fish and that “late-maturing fish are at a strong disadvantage relative to early-maturing fish under the current high levels of exploitation”. [Jørgensen et al. \(2007\)](#) concur that “[l]ife history theory predicts that increased mortality generally favours evolution towards earlier sexual maturation at smaller size and elevated reproductive effort”. [Blanchard et al. \(2005\)](#) identify not minimum-size regulations but the increase in overall fishing mortality as the main driver for the observed changes in the ecosystem size spectrum of the Celtic Sea. In summary, none of the studies cited in these BH publications supports the contention that minimum-size regulations cause size/age truncations and early maturation.

Lack of evidence that “more selective fishing neither maximizes production nor minimizes impacts”

One of the predictions of the basic population model is that, for a given F , increase of length at first capture towards an optimum value will increase yield ([Beverton and Holt, 1957, 1966](#)). In contrast, [Garcia et al. \(2012\)](#) state that “Increasing evidence suggests that more selective fishing neither maximizes production nor minimizes impacts (4–7)”. The numbers 4–7 refer to [Garcia et al. \(2011\)](#), [Misund et al. \(2002\)](#), [Bundy et al. \(2005\)](#), and [Zhou et al. \(2010\)](#). In the following, these publications are examined for evidence in support of the statement.

[Garcia et al. \(2011\)](#) is a report from a 2010 meeting in Nagoya, Japan, defining and discussing BH. The report contains several examples of multispecies fisheries where protection of juveniles led to unexpected side effects. But, there are also examples where expanding fishing to include juveniles substantially decreased yields (e.g. [Law and Grey, 1989](#)). The abstract of the report indicates a lack of conclusive evidence of BH: “The few attempts to verify the impacts predicted by [BH] models in real ecosystems with empirical data had limited success, indicating that such a demonstration might be a significant challenge” ([Garcia et al., 2011](#)).

[Misund et al. \(2002\)](#) propose unselective, multi-gear, small-scale fisheries as “the optimal exploitation pattern that exists”, without any supporting evidence with regard to relative yields and biomass or extinctions. [Bundy et al. \(2005\)](#) used ecosystem modelling to compare heavy exploitation of only top-level predators with heavy exploitation of only low-trophic level species, and with moderate exploitation of all trophic levels according to BH. They find that selective fishing for only the low-trophic level groups, and not BH, would maximize yield and minimized ecosystem disturbance. [Zhou et al. \(2010\)](#) is a “perspective” paper calling for a “balanced exploitation” approach similar to BH. It does not contain any data in support of the quoted statement.

Finally, the publication of [Halliday and Pinhorn \(2002\)](#) is a “mini review” of historical problems in fisheries management and in size-truncated fish populations, without data or simulations or a systematic meta-analysis of relevant studies to support the quoted statement. In summary, none of the five publications cited by [Garcia et al. \(2012\)](#) in support of their statement that increase in

size at first capture will neither increase yield nor minimize impact, actually contain data or modelling or a meta-analysis in support of that statement.

Ecosystem models do not support BH

At the heart of arguments about the benefits (for yield and ecosystem structure) of BH is a critical assumption about how fishing affects natural mortality rates of organisms via changes in trophic interactions (predation rates). BH asserts that harvesting of small creatures (including juveniles of larger fish) will result in substantial replacement of natural mortality rates with fishing mortality, with only a moderate increase in total mortality rates, so that large yields of small creatures can be taken while still taking close to maximum sustainable yields of larger creatures. In contrast, single-species models typically assume constant (though possibly size-dependent) natural mortality rates, so that fishing adds to these rates causing higher total mortality and, more particularly, high cumulative mortality for larger species before these species reach valuable sizes. But even if the BH argument is correct, the decreases in natural mortality rates, due to reduced predation mortality, must involve a substantial decrease in abundance of the larger predators. Even if natural mortality were indeed partly replaced by fishing mortality, the prey biomass taken by fishing is removed from the system rather than being propagated to higher trophic levels. The biomass of these higher levels will therefore shrink, even without fishing on these levels ([Smith et al., 2011](#)).

In most exploited ecosystems, high-value high-trophic level species have been overfished before low-value low-trophic level species were targeted ([Pauly et al., 1998](#)). In other words, even if mortality compensation can be demonstrated in simulations or experiments, such compensation is unlikely in real-world ecosystems where upper trophic level species have already been reduced by fishing. Rather, the rebuilding and sustainable exploitation of high-value species will require reduced exploitation of low-trophic levels ([Pikitch et al., 2012](#); [Essington et al., 2015](#)).

Because of the lack of empirical evidence supporting the notion that fishing at all trophic levels will be compensated by reduced predation at low-trophic levels, various trophic interaction and/or ecosystem models have been applied to make predictions about how the mortality rates at lower trophic levels might change if higher trophic levels are fished at rates close to their respective productivity. However, ecosystem models in general do not, and cannot, support the proposition that BH across sizes and trophic levels will somehow result in less disruption of ecosystem structure than moderate size-selective harvesting of selected species. It is a basic consequence of thermodynamic losses during trophic transfers of biomass and energy that appropriation of production of smaller creatures by fishing, i.e. the substantial removal of biomass from the ecosystem, necessarily causes severe and cumulative (over size and trophic level) negative impact on production available to support larger species ([Smith et al., 2011](#)), along with reduced prey densities and availability that further impact negatively on those larger species. The modelling exercises used in BH publications to support their opposite view are investigated in the following paragraphs.

[Garcia et al. \(2012\)](#) assert that Ecosim ([Walters et al., 1997](#)) policy optimization procedures support the use of BH policies for a wide range of ecosystems. In fact, Ecosim predicts that overall yields would be maximized by massive fishing of low-trophic levels, although this results in the elimination of larger species and has a severe negative impact on biodiversity ([Christensen and Walters,](#)

2005; Walters *et al.*, 2005; Cury *et al.*, 2012). Moreover, the assertions of Garcia *et al.* (2012) are misleading in at least four ways:

First, in the analysis of Garcia *et al.* (2012, Fig. 1), existing “selective” fisheries show maximum yields at moderate exploitation rates, with resulting biomass levels of about two-thirds of unexploited ecosystem biomass. Fishing of additional species that have not yet been considered profitable to pursue, increases yields while further reducing ecosystem biomass. But, the overall higher yields and higher biomass with fewer extirpations at higher exploitation rates predicted for unselective fishing result from including more, mainly small, abundant and productive species in the totals and proportions for BH, thus masking the negative effects of harvesting those small species on yield and value from existing fisheries (Walters *et al.*, 2005; see also similar evidence in African inland fisheries presented below).

Second, Garcia *et al.* (2012) use the Ecosim profit maximization criterion rather than the yield maximization criterion, with default Ecosim prices (identical for all species and sizes) and default fishing costs (large proportion of base incomes). Because of the relative high cost of fishing, the profit maximization criterion with these default settings results in relatively low fishing efforts that are much less destructive of ecosystem structure than the efforts which would maximize ecosystem yield.

Third, even these economically optimum efforts commonly result in Ecosim predictions involving disappearance of larger species, and for that reason Ecosim includes a “mandated rebuilding” constraint option. Garcia *et al.* (2012, p. 4 in Supporting Online Material) set a rebuilding constraint of 40% of unfished biomass in all Ecosim models, thus generating a circular argument: they forced the goals of high value and maintenance of ecosystem structure (i.e. high biomass of long-lived ecosystem components) onto the optimization process, rather than letting these results emerge as an objective outcome from applying the BH approach to fishing.

Fourth, Garcia *et al.* (2012) assert that Ecosim predicts optimum selective “fishing mortality rates so low that yield is not economically sustainable”. This assertion is not correct. Ecosim catches are scaled relative to historical catches, which were certainly not “uneconomical”, and in fact often show how it has been economical to overfish stocks before they were depleted. Ecosim only calls for reduced fishing mortality rates when that would result in increased catches.

When reasonable economic inputs (prices and costs) are provided, Ecosim typically predicts a highest value from policies that involve selective harvesting across the ecosystem of both small and large organisms, targeting highly valued small and abundant creatures like shrimp, crabs, and bait fish for sport fishing. In other words, if realistic market prices rather than one uniform price for all species and sizes had been used in the Ecosim models in Garcia *et al.* (2012), then Ecosim optimization would have suggested highly selective fishing for valuable species as the superior policy. Note, however, that such selective fishing also contains risks. Ecosim models sometimes predict that economically selective harvesting may lead to unintended changes in ecosystem structure (Zhou *et al.*, 2010), in particular so-called cultivation–depensation and overcompensation effects (e.g. Walters and Kitchell, 2001; Choi *et al.*, 2004; Engelhard *et al.*, 2014; Gardmark *et al.*, 2015; Levin and Mollmann, 2015), where reduction in abundance of naturally dominant species results in increases in less-valued competitors and predators (e.g. small piscivores, or jellyfish) that then reduce productivity of the dominant species and create risk of the ecosystem being trapped in an unproductive equilibrium (an ecosystem

resilience issue). That this issue is not only of academic interest is illustrated by the northern Benguela upwelling ecosystem, where intense fishing pressure on the larger fish, which previously dominated that ecosystem, resulted in its flipping to a new, and apparently stable state dominated by jellyfish (Richardson *et al.*, 2009). Changes like this are an argument for reduction of fishing pressure on predators and competitors of the problematic species, or for selective fishing of the problematic species, not arguments for fishing everything in the system.

Size-spectrum models used to justify BH are highly unrealistic

Size-spectrum models have become a popular tool used to support BH (e.g. Rochet and Benoît, 2011; Law *et al.*, 2012, 2013, 2014; Jacobsen *et al.*, 2014). For example, Law *et al.* (2012) present an ecosystem consisting of a fixed plankton community and a single fish species. Fish start life as an egg of 1 mg. Fish larvae feed on plankton and grow, until they are large enough to feed exclusively on smaller individuals of their own species. Once fish reach maturity, a proportion of assimilated food (rather than body weight, as normally assumed) is allocated to the production of new eggs. As the density of adults increases, more juveniles are consumed, and thus recruitment declines with the biomass of adults (rather than being largely independent of spawner biomass above a certain minimum level, as empirically observed). To prevent a build-up in density of the largest fish, a very high “senescence” mortality is introduced, with $M = 5 \text{ year}^{-1}$ for fish above half of maximum weight of 1000 g, whereas the typical adult mortality for such fish would be less than one-tenth of that value. In contrast to real populations, this combination of poorly supported and/or inappropriate assumptions leads to a peak in cohort biomass at the smallest body size (Law *et al.*, 2012, Figure 1a). Curiously, in another study, Law *et al.* (2013) also assert a minimum impact of BH on a modelled population size spectrum, but in that analysis they use total mortality rates (fishing plus natural, see a survivorship plot in their Fig. 2) that approach zero for fish aged about 6 years and older. Other models assume a strong replacement of natural mortality rates of small creatures by fishing mortality (e.g. Engelhard *et al.*, 2014).

Size-spectrum models that have predicted higher yields from individual species by using BH (Law *et al.*, 2013, 2014; Jacobsen *et al.*, 2014) have assumed mortality rates high enough to cause decreasing biomass with size over a wide range of sizes within each species, as if decrease in biomass with size for communities as a whole also applies to each species within the community size spectrum. However, species contributions to the biomass spectrum are typically dome-shaped [see, e.g., Duplisea *et al.* (1997) and Kolding and van Zwieten (2014, Fig. 7a)] and there is lumpiness in size distributions even at the scale of trophic levels (Boudreau and Dickie, 1992; Yurista *et al.*, 2014), implying higher yield by selectively fishing each species at near the size where its cohort biomass is maximum. Beyond lacking empirical support, the assumption that each species exhibit maximum biomass at small body size does not make evolutionary sense. Were such a situation to develop naturally, the evolutionary effect would be strong selection for reduction in body sizes at maturity (Roff, 1984; Jensen, 1996; Law, 2000, 2007; Charnov *et al.*, 2001; Mangel *et al.*, 2007; Lester *et al.*, 2014) due to there being no selective advantage (no increase in lifetime reproduction) from delaying maturity so as to increase body size and fecundity. In summary, the size-spectrum models presented in support of BH make a number of unrealistic and even contradictory

assumptions, which call into question the validity of their support for BH.

African inland fisheries are not a desirable role model for future global fisheries

The Lake Kariba fishery is probably the most prominent case study used as supportive empirical evidence of BH (Garcia *et al.*, 2011, 2012; Kolding and van Zwieten, 2011; Law *et al.*, 2012; Kolding and van Zwieten, 2014; Garcia *et al.*, 2015). Between 1980 and 2000, fisheries in the Zimbabwean part of Lake Kariba were actively managed with limited access, closed areas, and enforced gear and mesh size restrictions, whereas the fishery on the Zambian side was an unregulated open access fishery. Over that period, catch per unit effort (cpue in kg set⁻¹) in a fishery-independent survey was about seven times lower in the open access fishery (Kolding *et al.*, 2003), suggesting a severe reduction in community biomass. BH, however, argues that reduced cpue is a normal response to fishing, and that similar diversity and community size structure in the two fisheries indicate that the open access fishery was not overfished (Kolding *et al.*, 2003; Kolding and van Zwieten, 2011, 2014). Tweddle *et al.* (2015) question this interpretation because the community analysis included wide-ranging species that may have come from the regulated area and which, therefore, may have masked local overfishing (e.g. Økland *et al.*, 2005). Furthermore, Tweddle *et al.* (2015) point out that, in the open access fishery, valuable cichlid species appear to have been replaced by very low-value small catfish (*Synodontis* species), which now constitute ~70% of the gillnet catches.

Perhaps, the best African example of the consequences of unselective, unregulated fishing is from Lake Malombe, a shallow lake (390 km²), which is situated 12 km downstream from the outflow from Lake Malawi (Tweddle *et al.*, 2015). Here, the introduction in the 1980s of a small-meshed purse seines (nkacha net) to harvest previously unexploited small haplochromine cichlids caused the collapse of a highly valuable fishery for chambo (Weyl *et al.*, 2004), which is a collective name for three large tilapia species comprising the *Oreochromis* “Nyasalapia” species flock (Trewavas, 1983). The nkacha net catches a wide range of species (>50 taxa) at all life history stages (Weyl *et al.*, 2004). Its introduction initially increased yields threefold (from 4000 to 12 000 t year⁻¹), with additional catches of small haplochromine cichlids and juvenile chambo (Weyl *et al.*, 2004), in line with the prescriptions of BH theory. But, Tweddle *et al.* (1995) assessed the fishery in 1992 and warned that the reliance on tiny, immature fish could lead to a collapse. Subsequently, catches decreased to levels close to the 4000 t year⁻¹ before the introduction of the nkacha net, but now consisting mainly of small, low-value species, because of recruitment and growth overfishing of the high-value chambo species (FAO, 1993; Tweddle *et al.*, 2015).

While Kolding and van Zwieten (2011) argue that most fisheries will reach some form of economic break-even point long before the extinction of the target species, they ignore that in Africa, where economic alternatives to fishing are often absent, fishers have no choice but to stay in the fishery (Allison and Ellis, 2001). In Lake Malombe, this has resulted in dwindling economic returns per fisher from about 11 USD per day during the peak of the chambo fishery to about 2 USD per day after the collapse of the fishery (Tweddle *et al.*, 2015). Contrary to the suggestions by Kolding and van Zwieten (2011, 2014), unregulated fishing still leads to the tragedy of the commons (Hardin, 1968) and “rhymes with poverty” (Béné, 2003).

The open access African inland fisheries presented in BH publications as “closest example of the optimal exploitation pattern that exists” leading to ecosystems where “everything is less abundant, [but] the relative fish community structure remains largely unchanged” (Kolding and van Zwieten, 2011, p. 17–18) are in reality the remnants of previously valuable fisheries, with fish communities now dominated by small, low-value species (Weyl *et al.*, 2010; Mkumbo and Marshall, 2015; Tweddle *et al.*, 2015). The proclaimed preference for such fish is instead often the result of lack of choice due to large-scale collapses of truly preferred species such as large cichlids (Tweddle *et al.*, 2015).

The Lake Malombe scenario is not unique and the loss of high-value target species due to unregulated effort and selectivity is a problem in many African inland fisheries (Tweddle *et al.*, 2015), including Lake Malawi (Weyl *et al.*, 2010) and Lake Victoria (Mkumbo and Marshall, 2015). Many fisheries have relatively stable total yields, but catches are now characterized by low-value species and decreased individual catch and income.

In summary, it is very questionable whether the examples of African inland fisheries that follow BH principles are indeed a desirable role model for future global fisheries.

Real-world fisheries do not support the predictions of BH

While consistent evidence in support of the predictions of BH is lacking, there is no lack of evidence to the contrary, i.e. of ecosystems with exploitation of nearly all species at nearly all size levels as proposed by BH, but with stocks exhibiting truncated size structures, early maturity, severely depleted biomass and with reduced overall catches from the system. For example, fishing in the North Sea affects most of the species, either as target or as bycatch or as collateral damage from bottom trawling. Excavators are used in fisheries on mussel beds and shrimp trawls operate in shallow water nursery grounds with 16 mm mesh size. A variety of trawls, traps, hooks, gill-nets, and other gears are employed throughout the North Sea, without any major marine-protected area closed to fishing. Minimum-size regulations for species are typically set below the size at maturity (Froese *et al.*, 2015), and there is a substantial catch of small individuals and non-target species that have traditionally been discarded at sea, but which is to be partially landed and used as fishmeal under the new fisheries policy (CFP, 2013). Exploitation rates have been reduced towards natural productivity rates for many stocks in recent years and some of these stocks have already increased in biomass. Thus, many of the prescriptions of BH are already in place; yet, most stocks show distorted size structures and are, with few exceptions, below levels that can produce maximum sustainable yields (Froese *et al.*, 2015).

A strict implementation of BH in the North Sea would require increased exploitation of former bycatch species and of very small size classes, including the early juveniles of all commercial fish. Current stock assessment documents (e.g. ICES, 2014a) reflect the dynamics of the exploited populations and their interactions through recruitment, growth, and natural mortality within the North Sea ecosystem. These assessments leave no doubt that full exploitation of early juveniles will decrease recruitment, spawning-stock biomass, and yield of the stocks that currently provide most of the catch. No compensation of fishing mortality by reduced predation mortality of juveniles is to be expected, because most predator stocks are already at low levels. Rather, intensified extraction of biomass from lower trophic levels is likely to further reduce biomass

of the upper trophic levels (Smith *et al.*, 2011). Such fishing may result in the relative size composition in the ecosystem being similar to one with higher overall biomass, as desired by BH (Garcia *et al.*, 2012), but it would basically turn the North Sea into a system optimized for the production of fishmeal, with severe negative consequences for biodiversity and far from the good environmental status aimed for in the Marine Strategy Framework Directive (MSFD, 2008) and the Common Fisheries Policy (CFP, 2013) of the European Union.

Similarly, nearly 400 species of fish, crustacean, and molluscs are exploited in the Mediterranean according to international catch statistics (FAO, 2015). Minimum-size regulations are set mostly below the size at maturity and are weakly enforced, resulting in a large proportion of juveniles in the catch (Tsikliras and Stergiou, 2014). Bottom trawls typically operate with a mesh size of 20 mm, which can be as low as 13 mm in non-EU countries, resulting in (discarded) catch of undersized or unwanted species that may reach up to 65% of the landings and an average of 18.6% across all gears (Tsagarakis *et al.*, 2014). Thus, three key requirements of BH are already in place: the catching of all species (i) at all sizes (ii) with highest biomass extraction from small species and juveniles at low-trophic levels (iii) (Figure 1). Exploitation rates of adults typically exceed maximum sustainable levels (Tsikliras *et al.*, 2015), but these rates may match the higher productivity of early juveniles and smaller species in the bycatch, thus fulfilling another requirement of BH, at least in part. Yet, contrary to the predictions of BH, the vast majority of Mediterranean stocks are outside of safe biological limits (Colloca *et al.*, 2013), their body lengths are shrinking (Damalas *et al.*, 2015), and catches are declining (Tsikliras *et al.*, 2013; FAO, 2015). A strict implementation of BH would reduce exploitation rates of adults while further increasing the exploitation of small species and juveniles of all larger species that are currently part of the bycatch. This would open new intensive fisheries on even smaller size levels. But, compensation of fishing mortality at low-trophic levels as a result of reduced predation is not to be

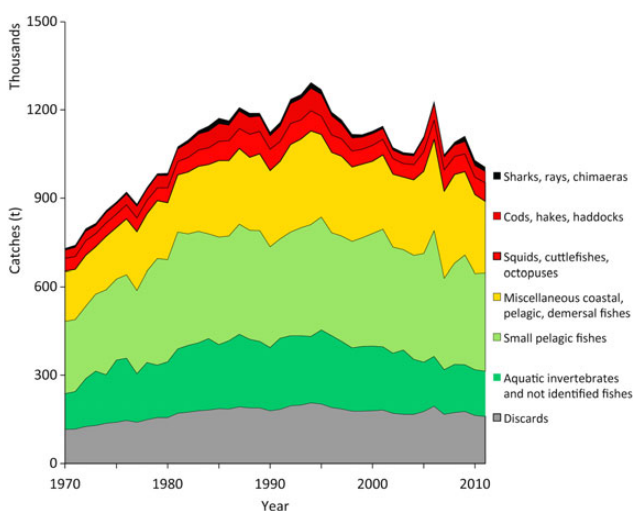


Figure 1. Landings and discards reported for the Mediterranean, by major groups of organisms ranked by the mean trophic level. Note that catches include a wide range (>400) of species and decline strongly with an increase in the trophic level, as demanded by BH. Yet, overall catches are declining and most Mediterranean stocks are outside of safe biological limits.

expected because abundance of predators is already low. In contrast, most recent studies conclude that strong reduction of fishing mortality is the only way to recover Mediterranean stocks (Tsikliras, 2014; Damalas *et al.*, 2015; Osio *et al.*, 2015; Tsikliras *et al.*, 2015), sometimes with an explicit emphasis on shifting size of first capture towards optimal length (Colloca *et al.*, 2013) or limiting juvenile exploitation (Vasilakopoulos *et al.*, 2014).

Who would benefit from BH?

It is reasonable to ask who would benefit if the BH approach to fishing was to find wide acceptance among fisheries managers. Fishers may like it in anticipation that gear restrictions and many regulations on minimum size, bycatch, or discard avoidance would disappear, with all sizes of all species becoming regular catch. Also, catches would increase initially as more species and sizes are fished. However, fishers are apparently not aware of the likely consequences for future catches, which will consist mostly of low-value species and small fish, which even BH sees as problematic “[...] in markets where small fish have low value compared with large fish” (Law *et al.*, 2013). Since much of the new catch would go into fishmeal, the aquaculture industry would benefit from the increased supply and presumably lower price of pellets. And hunters of seabirds [see Grémillet *et al.* (2015)], seals [see Pannoizzo (2013)], and whales (Komatsu and Misaki, 2001) would also benefit, because exploitation of the widest possible range of species under BH demands that these groups become part of the regular harvest, with fished groups including “all groups historically fished (including whaling, sealing, etc.)” (Garcia *et al.*, 2012, p. 2 in Supporting Online Material). In other words, instead of avoiding impact of fishing on non-target species, the targeted killing of seabirds, whales, dolphins, seals, and other non-fish groups becomes a scientific requirement of ecosystem-based fisheries management (EBFM) under BH.

Moderate fishing and maximum sustainable yields

BH aims for exploitation rates in proportion to natural productivity of the respective species and size group (Garcia *et al.*, 2012). Fishing mortality equal to size-dependent natural mortality has been proposed as the default exploitation rate under BH (Law *et al.*, 2013), although much higher exploitation rates are also deemed compatible with BH as long as the size spectrum is maintained (Kolding and van Zwieten, 2011, 2014; Garcia *et al.*, 2012). However, the rate of natural mortality is a widely accepted proxy for the maximum sustainable rate of fishing [see review in Froese *et al.* (2015)], i.e. such fishing pressure is not moderate but the maximum that a given stock can tolerate in the long term. The acceptance of BH with such fishing pressure would therefore lead to the abandonment of one of the few currently uncontested tenets of EBFM that maximum sustainable yields cannot be achieved simultaneously for all exploited stocks in an ecosystem: if we wish big fish, we must leave prey for them to feed on (Pikitch *et al.*, 2012; Essington *et al.*, 2015). Also, and this is more ominous, BH would provide a justification for the further development of industrial fisheries for zooplankton. Artisanal fisheries for zooplankton currently exist, e.g. in the form of fisheries for Akami paste shrimp (*Acetes* spp.; Omori, 1975) and jellyfish (Brotz, in press). However, industrial fishing for zooplankton is currently conducted only in Antarctica (for krill, *Euphausia superba*; Olsen *et al.*, 2006) and in the Norwegian Arctic (for “redfeed”, i.e. copepods, mainly *Calanus finmarchicus*; Tiller, 2008), with both fisheries producing feed for aquaculture. We question whether these fisheries should be emulated in other parts of the world, and doubt that they would

increase the availability of fish to the poor people in developing countries, one of the reasons advanced in support of BH.

Practical and conceptual problems of BH

Practical and conceptual problems of BH were also raised at the recent BH workshop in Rome. The executive summary from that workshop (Garcia *et al.*, 2015, p. 11) notes the following: “In relation to how BH could be considered as a strategy for implementation of the Ecosystem Approach to Fisheries, it was recognized that BH only addresses the objective of food production and maintaining ecosystem structure and functioning. Other ecological objectives (e.g. those related to minimizing impacts on habitats), as well as social and economic objectives, are not explicitly covered by BH [...]. Furthermore, there is also the issue as to what extent excluding taxonomic groups (e.g. charismatic species) and sizes (e.g. juveniles, adults) from the BH equation would lead to desirable outcomes”.

These recognized deficits of BH are, in fact, addressed by existing fisheries policies (e.g. Restrepo *et al.*, 1998; DAFF, 2007; MSFD, 2008; CFP, 2013) aiming for targeted exploitation of resilient and economically valuable species at levels below the maximum sustainable yield and at sizes that ensure growth and successful reproduction, while minimizing the impact of fishing on the ecosystem and the environment.

Conclusions

BH calls for fishing across the widest possible range of species, stocks, and sizes in an ecosystem, in proportion to their natural productivity (Garcia *et al.*, 2012). Such fishing is suggested to minimize the impact of exploitation on the structure of ecosystems and to lead to increased yields from larger stocks with less truncated size and age structure. We examined the models and the empirical evidence put forward in support of BH. We found that the models used unrealistic assumptions and settings, and that conclusive empirical evidence of BH is lacking. Instead, moderate harvesting of resilient species for human consumption, with least possible impact on stocks and ecosystems, is still the most promising approach for sustainable use of the living ocean. Nearly 70 years after the International Overfishing Conference in London (UN, 1946), which aimed at ending overfishing, and over 30 years after the binding regulations of the Law of the Sea on the sustainable use of living marine resources (UNCLOS, 1982), it is high time that sustainable, profitable, low-impact fishing is finally implemented on a global scale. The BH proposal for fishing all species at all sizes is more likely to hinder these efforts than to help them.

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