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Biomass from Fisheries: Provisioning Services and Benefits

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Fisheries biomass forms the base for a range of potential ecosystem service flows and benefits, ranging across provisioning services (food for consumption), cultural services (fish catch for recreational enjoyment), and regulating services (influencing the biomass of other fish populations). Each of these service flows and benefits impacts different end users and therefore rely on different methods for their measurement and valuation. The focus of this paper is on the provisioning flows that arise from fish biomass and strategies available for measuring the physical size of this asset and its flows as well as valuing the flows that enter consumption and production processes. Valuation of recreational and existence uses of biomass is briefly discussed, but a separate issue paper will more fully assess the general area of recreational use valuation. Figure 1 provides a logic chain summarizing the overall flow from asset to human end user for this provisioning service of biomass.

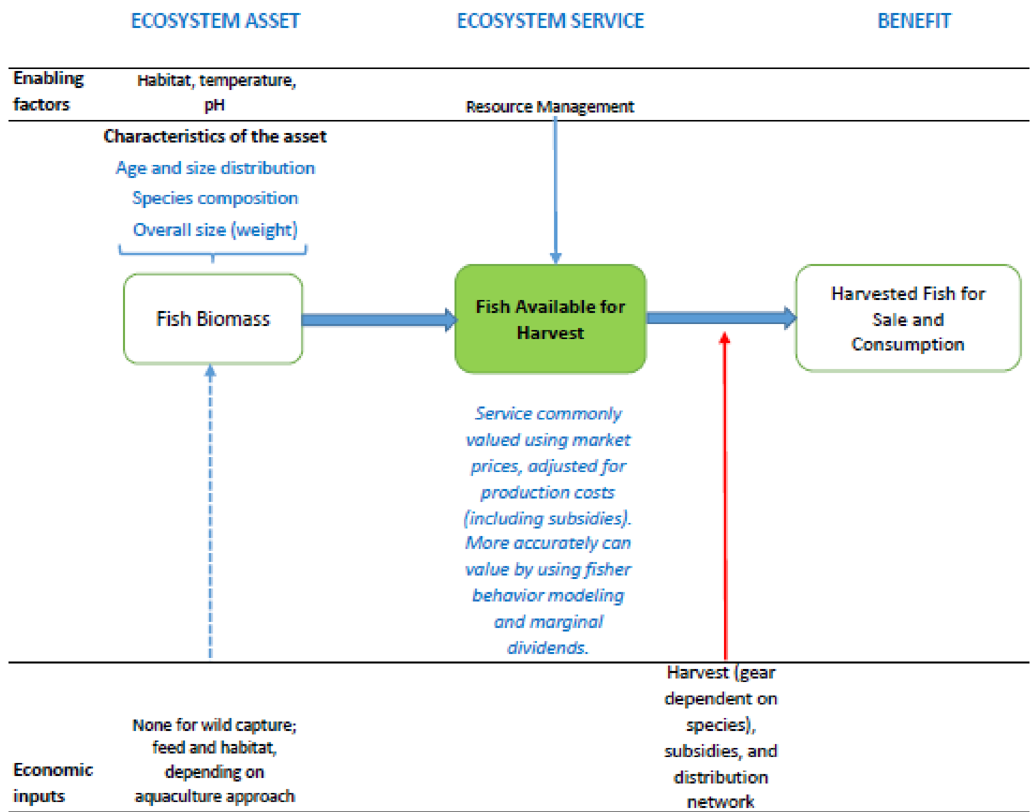

Figure 1. Logic chain for provisioning service of biomass from fisheries

Measurement of physical stocks and flows

A variety of approaches are available for quantifying the physical stocks and flows associated with the provisioning service of fisheries biomass. These include the use of catch statistics to quantify flows as well as the use of survey trawls, modelling approaches, satellites, and novel genetic techniques to estimate the size and distribution of the biomass stock. Each approach has its own embedded uncertainties and different costs associated with the collection and support of the data collection. These strategies are described in additional detail in the following sections. While physical collection of fish by scientific surveys of fishing fleets has been the primary means of data collection for a century, reductions in cost through alternative data sources (e.g., DNA) are opening up new avenues and further revolutions in data availability for both the developed and developing world will be likely over the next 1-2 decades. Currently, costs associated with many fisheries-independent data sources preclude their use for low value stocks or in nations with limited resources. Consequently, significant effort is being put into maximizing the information that can usefully be recovered from available data streams (e.g. Dowling et al 2016).

Role of catch statistics

Catch and aquaculture production data – the volume of biomass landed by fishers and farmers – would be the most direct measure of the flow of biomass out of the ocean and to beneficiaries.[1] However, this belies the significant investment that would be required to interact with the approximately 59.6 million people (as of 2016) engaged in the primary sector of capture fisheries and aquaculture (FAO 2018). The FAO aggregates the national scale reports provided by countries around the world, making the data available through its faostat portal (www.fao.org/faostat), similarly the OECD provides its own summary of fisheries catches via its data website (https://data.oecd.org/). However, as has been broadly reported elsewhere (e.g., Pauly and Zeller 2016), these data are subject to the idiosyncrasies of country reporting habits. While the FAO is actively trying to support improved national data collection systems and revise data found to be incorrect (e.g., Myanmar; FAO 2018), there is no denying that it is exceptionally hard to reliably capture the landings of the many small scale fisheries around the globe. The volume of material discarded is also uncertain, and it is not routinely or consistently recorded in all jurisdictions (in some jurisdictions it is not monitored at all). Until these gaps are filled by technological or other means, catch reconstruction provides a source of estimates for total landed catch and discards – typically available as spatial maps and time series; for example, those of Watson (2017) and Watson and Tidd (2018), or Pauly and Zeller (2016) and Zeller et al (2017) and the associated materials at the Sea Around Us website (www.seaaroundus.org). While such reconstructions are subject to their many assumptions and uncertainties, if undertaken in close collaboration with experts on a region they can be relatively reliable.

At a national or finer scale, directly approaching the fisheries operators or regulatory body for a region – the local fisheries management authority – is an option. Many commercial fisheries require sophisticated (and increasingly electronic at-sea) reporting of catch, effort and in some cases discards in near real time (or at the very least at or prior to landing). Satellite monitoring of at sea movements (via automatic identification systems or radar systems) is also allowing for understanding of fine-scale spatial and temporal behaviors of fishing vessels of all sizes – whether operating legally or not (Bean et al 2017; Dunn et al 2018; Ford et al 2018). However, despite the oceans being a public resource in many jurisdictions it is typically not easy to access anything but the grossest of catch statistics (e.g., annual time series of total catch or catch of major species). More finely resolved data (which might be derived from commercial fisheries log-books) is either not available (as data is only coming from aggregate landing points) or is subject to restricted access due to commercial in-confidence rules (this is the case in places such as the United States, Poland, and Australia). Similar issues bedevil harvest data for aquaculture. Evaluation of catch data, therefore, is often best done

[1] It is not clear that aquaculture stocks should be counted as natural capital any more than domestic cattle, which are usually considered produced capital.

in collaboration with industry and regulators who can facilitate access to these data streams.

In jurisdictions where log-books are not in use but there is representative sampling of landing sites (e.g. India) sound time series may stretch back decades – though the level of aggregation and method of collection may vary over the entire period and local expert guidance is needed to both access the different data sets and to make sure they are used consistently. The same is true for the output of periodic (or opportunistic) surveys of recreational landings in countries where this is a significant source of mortality for coastal fisheries resources (Eero et al 2015). Significant effort has been put into finding methods that exploit social networks to maximize data gathering for these fisheries (e.g. Griffiths et al 2010, Griffiths et al 2014, Williams et al 2015).

Estimating biomass stocks: Role of monitoring approaches

Apart from estimation of the flows of the fish out of the ocean, significant effort has been invested in determining the stock of biomass for various fish species, often with a focus on spawning stock biomass (SSB). When evaluating estimates presented as SSB, it is important to note that these do not represent the total standing biomass in the ocean, only those population members capable of reproducing. Understanding the biomass in the ocean is viewed as important for developing appropriate management strategies for the fish catch flows out of the ocean based on these stock assessments. There is a healthy debate in fisheries science on the value of the catch statistics described above, known as *fisheries dependent data*, as a means of estimating stock size. While they are often the only data available (Pauly 2013) – with the advent of smartphone-based apps facilitating reporting even by small scale and coastal fisheries (e.g., as done in Norway; Williams and Wathne 2018) – there are many confounding factors which make catch as the sole source of information potentially highly misleading (Hilborn and Branch 2013)[2]. Nevertheless, catch per unit effort from commercial harvest remains a key form of input data for many stock assessments (Bean et al 2017).

Where possible utilizing *fisheries independent* data is highly desirable, though these are not immune from selection bias issues. There now exist a broad range of monitoring approaches that provide data useful for estimating biomass, many of which are regularly conducted for specified species. The topic is such a large one that international conferences on fisheries observing and monitoring are regularly held, with the 9th held in Vigo Spain in June 2018, and bodies such as ICES have multiple working groups dedicated to the suite of available monitoring methods (http://www.ices.dk/community/groups/Pages/EOSG.aspx).

[2] Though statistical techniques such as those in Zhang and Smith 2011 can be helpful, they are rarely applied.

Trawl Surveys

Historically the most widely employed fisheries survey methods have been trawl surveys. Run from research vessels or chartered industry vessels, these surveys provide quantitative samples of species composition, along with biological information such as age, length, weight, maturity and gut contents. These data are routinely used in assessments to estimate abundance and distribution of target species. Many research cruises simultaneously collect oceanographic properties – such as water temperature, salinity, oxygen and plankton content – providing a more systemic view of marine ecosystems.

While there is as yet no publicly available database of trawl surveys conducted globally, efforts are underway to provide just such a resource, with an early precursor available in Worm et al (2009), which included surveys from 16 regions (primarily from the Northern hemisphere) in an assessment of global fish stock status. However, this compilation omits many long-term data sets held in research organizations in locations such as Chile, Thailand, India, South Africa, Russia and China (to name but a few). In addition, there are the UN Nansen surveys (http://www.fao.org/in-action/eaf-nansen/background/history-of-the-nansen-programme/), which have collected information on natural resources within EEZs and large marine ecosystems as a series of snapshots from 1975 until now. These can provide invaluable data sources in regions where no local sampling occurs, though they can only be accessed with agreement from the in-country partners.

Alternatives to Trawl Surveys

Trawl surveys are only one of many data collection methods available (Figure 2 provides an example of monitoring data streams available from multiple platforms in UK waters). The most frequently used supplements include other biological survey methods – using other fishing gears, ichthyology counts for daily egg production estimates (Stratoudakis et al 2006) or acoustic samplers. These other methods are often used to target species not well covered by trawls – with the various acoustics methods (for example) now able to target species throughout the water column (Trenkel et al 2016), once ground truthing of signal patterns has been completed (Bean et al 2017). Rapid technological development over the past decade has seen a growing number of potential sampling methods start to become available (with broader adoption likely within the next decade or so). Autonomous and remotely operated vehicles are greatly expanding potential sampling (Suberg et al 2014). While remotely operated vehicles typically operate as tethered sampling platforms (augmenting or replacing extractive sampling; Bean et al 2017), autonomous vessels have now advanced to the point where they can undertake missions ranging over thousands of kilometers (Cokelet et al 2015). Miniaturization and other sensor advances means that these platforms can mount cameras, video, acoustics, water and molecular samplers – they can collect standalone data or ground-truth other data streams (Bean et al 2017). Animal mounted sensors can be used in a similar way, though these have traditionally been limited to trackers, accelerometers, cameras or passive samplers for environmental properties (e.g. Volpov et al 2015). This use of animals as the sensor platform allows sampling of areas where traditional methods are often difficult to deploy.

Satellite-based observations are a growing method of providing synoptic estimates of production (Saba et al 2011), which have been bolstered by tailored algorithms that increase precision, particularly in coastal waters (e.g. Tilstone et al 2017). At the other extreme of scale, genetic material is also providing new opportunities for estimating abundance and distribution of marine species. Tag-recapture studies – using physical tags or photo identification – is a routine method used to estimate the abundance of a number of species (Pine et al 2003). As a result of advances in genetic methods, which now affordably and reliably identify parent–offspring pairs (or other close-kin relationships), genetic material can act as the tag – with genetic samples compared to see if any close-kin relationships are present and from that estimates of adult abundance can be made that are directly analogous to Lincoln–Petersen abundance estimates from standard mark-release recapture studies (Bravington et al 2016). Another revolution in genetic sampling has come in the form of eDNA (DNA collected from water and other samples and analysed using high-throughput sequencing methods). The use of these metagenomic samples on riverine and open ocean samples, to map occurrence (Carraro et al 2018) or create transects of biodiversity or relative abundance, has to date primarily been restricted to marine microbes (e.g. Raes et al 2018). However, a growing list of 'DNA barcoded' species and a greater familiarity with other biochemical tracers means these forms of data – collected from many sources, including gut contents – can now be used to inform ecosystem models (Pethybridge et al 2018) and provide simple estimates of relative abundance for invertebrate and vertebrate species too.

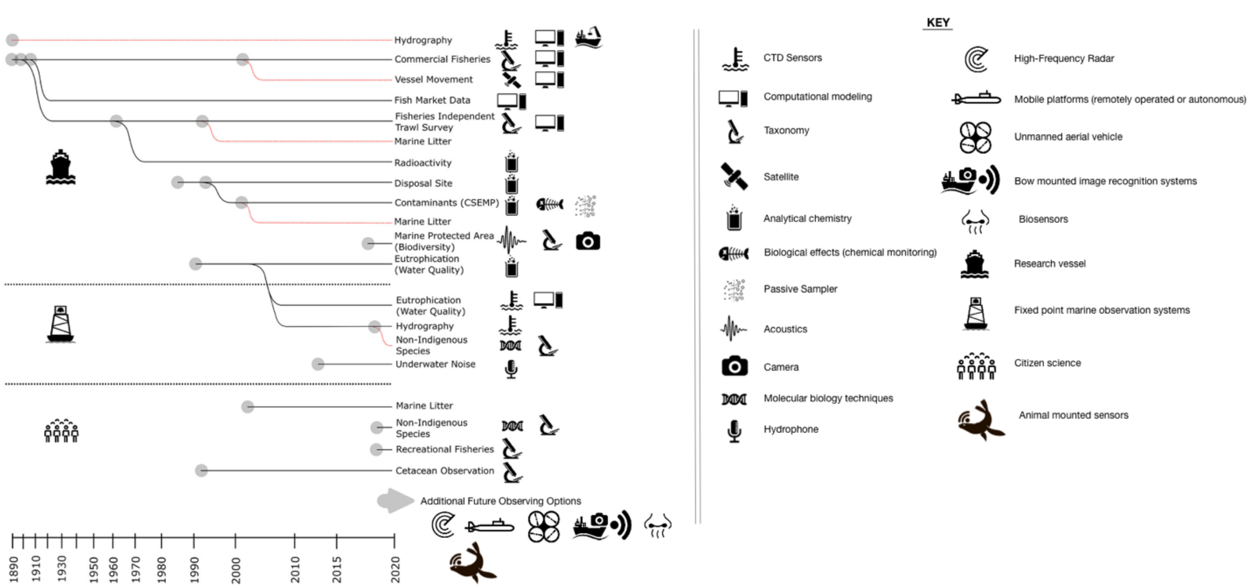

Figure 2: Examples of monitoring programs from multiple platforms (vessel, fixed point observing system, satellites, water samples, citizen science) available for UK waters. Gray dots mark the initialization of a new survey program and red lines denote opportunistic sampling. (Modified from Bean et al 2017).

No single sample method can cover the entire expanse of an ocean area. Moreover, oceanic sampling is stochastic and the target of interest (the fish species) are mobile, further complicating matters (Hammer 2018). This is why fisheries-based observer programs remain an important means of collecting supplementary biomass, abundance and distribution data (Mondragon et al 2018). For some of the data sets (e.g. discards) observer collected samples remain the primary data collection method. In addition, their understanding of at-sea behavior improves interpretation of both biological and catch and effort data collected from fishers (Hammer 2018).

The costs and complexities of human observer programs are a constraint in some jurisdictions. However, this is occurring at a time when lower cost and higher quality imaging technology is becoming available. While the volume of material to be reviewed can be staggering, the recent and considerable expansion in the capacity of automated image recognition algorithms (e.g. French et al 2015) is improving the feasibility of electronic monitoring in fisheries. Pattern recognition software is also being used in aquaculture to track in-pen abundance, size and animal health.

Citizen science is also providing a robust new means of collecting data at scales that have traditionally been infeasible for historical monitoring programs. Citizen scientists have assisted in the collection of tagging data, samples of size and age (Fairclough et al 2014) and information on species distributions (http://www.redmap.org.au). While data collected in this way must be handled with clear thought around potential biases and necessary corrections (Fairclough et al 2014) the method has good precedent – with bird enthusiasts responsible for the abundance and distribution estimates of many species worldwide (BirdLife International 2011).

Evaluating biomass stocks: Role of modelling approaches

The scales and difficulties involved in observing the ocean has long motivated the use of models to derive estimates of marine species abundance and to explore potential dynamics of these fluid environments. Over the last century these modelling methods have diversified into a suite of approaches, each with their own strengths and weaknesses. Entire university courses and textbooks have been written on the topic of population and ecosystem modeling and fish population models, specifically. A number of useful summaries and reviews exist (Quinn 1999, Plagányi 2007, Travers et al 2008, Fulton 2010, ICES 2012, Fulton and Link 2014, Chrysafi and Kuparinen 2016, Aeberhard et al 2018).

While this dynamic field (facilitated by expanding computational capacity and new data types) has new approaches and applications produced at a steady pace, the majority of these models fall into a few general classes. Amongst the simplest of these deals with estimation of the gross biomass based on energetic principles – the potential fish biomass available given levels of primary production and energy losses during transfers between trophic levels. Ryther (1969) and Pauly (1996) are two of the best known applications of these methods – and both estimated potential global fish production to be approximately 100 million t. Fogarty et al (2016) is one of the most recent examples, using a simple food web and spatial maps of size-structured phytoplankton production to allow for resolution of production into simple functional groups (including benthos, planktivores, benthivores and piscivores). This approach produced biomass and yield estimates per large marine ecosystem (LME), with aggregate total potential yield of approximately 140-180 million tonnes.

The most widely used means of estimating fish biomass are single species assessment methods. As early as 1922 catch-at-age data was being used to provide simple estimates of abundance (Derzhavin 1922). There has been a proliferation of methods since then, especially since the advent of many statistical procedures in the 1980s, with state space models that allow for process error in the population dynamics and observation error in the data used to estimate model parameters now used routinely to estimate abundance for some of the globally important fish stocks (Aeberhard et al 2018). Despite a diversity of approaches, single-species stock assessments can still be classified based on broad modelling approach and data needs (Punt et al 2013, Cadrin and Dickey-Collas 2015): (i) catch only models, (ii) time-series models, (iii) surplus production (or biomass dynamics) models, (iv) age- (or stage-) or size-structured models. All of these approaches have their strengths and weaknesses, and none should be applied blindly (Maunder and Piner 2015). A tension persists across these methods about what to do about a lack of a complete time series for many species – some have focused on more 'data limited' approaches, while others have put considerable effort into the development of integrated models, which use statistical frameworks to simultaneously employ all available data.

In many situations catch data is the only available source of information for exploited species. This has led to a number of proposed catch only methods (Froese and Kesner-Reyes 2002; Rosenberg et al 2017). However, such methods must be used with care as they are predicated on the assumption that catches are not biased (or inaccurate) and reflect the underlying exploitation and thereby abundance. Unfortunately, other factors can influence patterns of catch (Branch et al. 2011), and there can be mixed performance of methods across different life-history traits, initial levels of depletion, effort dynamics, and length of the catch time series (Rosenberg et al 2014). Apart from biology, institutional influences can also have an impact (Tewka et al. 2019). The use of a superensemble (as employed in weather and climate forecasting) may be a useful means of improving estimates of relative biomass, or at least relative stock status (Anderson et al 2017, Rosenberg et al 2017). However, important selection biases that depend on institutional arrangements and human behavior can remain (Ferraro et al. 2018).

Time series analysis can be used to infer population estimates, with catch-only methods as one example. Survey indices are another oft used biomass index. Other commonly available time series are catch length-composition data. These have also been used to provide estimates of relative biomass or stock status. Froese (2004) developed a set of 3 indicators using the proportions in the catch made up of individuals of specific lengths: (i) the proportion with length equal to or larger than the average length-at-maturity; (ii) the proportion of fish caught at optimum length (the length that leads to maximum yield and revenue, which is slightly larger than length at maturity); and (iii) the proportion of old, large fish in the catch (as these disproportionately contribute to reproduction and indicate a healthy age structure at a population level). Cope and Punt (2009) combined these 3 indices (by summing across them) and using the resulting values of the sum as well as the proportions of the three original indicators, created a decision tree that determined whether or not SSB is above the target reference point for the species (i.e. its stock status).

Surplus production models (e.g. Schaefer 1957, Pella and Tomlinson 1969) are one of the simplest assessment methods as they can be developed from catch time-series and an index of abundance (e.g. catch per unit effort). They were originally intended for application to individual species, but aggregate production models estimate total fish biomass by summing catch time series across all species and treating the result as one 'mega-species' (this obviously ignores the fate of individual species within the complex). While the equilibrium form of this method should not be used (due to perverse assumptions), approaches using regression and time series fitting to estimate parameter values can be effective (Polacheck et al 1993, Meyer and Millar 1999, Ono et al 2012); so long as attention is paid to the shape of the assumed production function (Maunder, 2003), impacts of transient age-structure are not substantial (Punt and Szuwalski, 2012), and the catch and effort data do not represent a "one way trip" (where catch per unit effort only shows continuous decline), as this means there is insufficient information content in the data to allow for reliable estimation of the key parameters (population growth rate, carrying capacity and catchability).

Age- and size-structured models are conceptually very similar. Indeed, both stem from what were originally statistical catch-at-age methods (Punt et al 2013). These structured models are elaborations of the basic concepts captured in delay-difference methods (which include at least two life stages, one each for pre- and post- recruitment to the fishable pool of the stock; ICES 2012). More elaborate models include more processes – such as recruitment – but equally require more data, such as catch, index of abundance, size or growth and natural mortality (ICES 2012). Key developments over the past 30 years of use have been in tailoring the models to the specifics of the species concerned (potentially including additional forcing factors such as environmental drivers or habitat; e.g. Plagányi et al 2013), parameter estimation and handling of uncertainty (Punt et al 2013). The number of parameters in these models means parameter confounding can be an issue (particularly for size-based models if not all processes are simply size-based or there is considerable variation in size-at-age). Inclusion of survey and tagging data in the assessment can alleviate these issues (Punt et al 2013). The most complicated forms of these models – integrated models – can be very complicated and highly data intensive (making them vulnerable to contradictory data and model-mis-specification), however they do allow for consistency in assumptions and propagation of uncertainty associated with data sources through to the final model outputs (Maunder and Punt 2013).

The most complex forms of models used to provide estimates and trajectories of biomass are multispecies and ecosystem models. The most straightforward forms are extension of single species approaches to multiple species – typically target species and their key predators, prey or competitors (e.g. Hall et al 2006, Gamble and Link 2009) – and their key environmental or anthropogenic drivers (Plagányi et al 2014). The concept behind these models is "intermediate complexity" or "minimum realistic" – i.e. what is the small set of included processes needed to represent the dynamics of the system of interest. While a similar philosophy can be taken to building food web and "end-to-end" (or whole of system) models, they do so at much broader scales, encompassing entire food webs or (typically regional) ecosystems. The earliest and most widely used forms of these models (e.g., Ecopath with Ecosim; Christensen and Walters 2004) focus on food web mediated interactions between biomass pools and require estimates of consumption, total mortality (fishing and natural mortality), biomass and diet composition as inputs. These models are structured around representation of the main functional groups making up a foodweb – sometimes also resolving key target species individually. The same is true of other ecosystem modelling approaches that incorporate explicit age (and size) structure and a broader suite of ecological processes – either via deterministic (Atlantis; Fulton et al 2011) or agent-based (OSMOSE; Shin and Cury 2004) frameworks. The input data requirements for these other approaches is substantially higher as input parameters are also required for each of the represented processes – growth, reproduction, movement, habitat mediation, etc. At the global scale such complex representations are typically beyond what is possible computationally and given available data (though exceptions exist, e.g., EcoOcean; Christensen et al 2015). Over the last decade this had led to the development of a number of basin to global scale models employing simpler computational concepts. These include multiple species distribution models (e.g. Cheung et al 2010), size-based models (Blanchard et al 2017; Galbraith et al 2017) and hybrid agent based models (e.g., APECOSM, Maury 2010; SEAPODYM, Lehodey 2005; Madingley, Harfoot et al 2014). These models are all built from fundamental metabolic or ecological concepts and, while model calibration and validation remains an important concern for these models, they are being used to produce estimates of potential fish production and how this may change into the future (Cheung et al 2016; Blanchard et al 2017).

Another approach to estimating the distribution (though less so the biomass) of fish species is to make use of the strong statistical relationships between fish occurrence and habitat or environmental variables (such as depth or temperature). For example, surveys of benthic biological data have been used to create predictive rank abundance distributions models based on oceanographic variables which are then applied along entire coastlines to produce maps of potential biodiversity (e.g. the east and west coasts of Australia; Dunstan and Foster 2011, Dunstan et al 2011). Gradient forest models can also be used in similar ways (e.g. Pitcher et al 2018). The success of these approaches is due to the strong relationships between many fish species and the habitats – either as nursery grounds or for adult habitat. There are observed correlations between area of habitat and fish catch (e.g., Robertson and Blaber 1992, Connolly 1994, Randall et al 1996, Husebø et al 2002, Unsworth et al 2018, and as summarized in Baran 1999, Manson et al. 2005). However, the importance of different habitats – be it mangroves, macrophytes, coral or indeed open ocean eddy structures – varies between species and life history stages. This is because habitats can provide a service to fish species in many ways – as refugia, sources of prey, reproduction sites etc. In addition, that relationship is not always direct (Blaber 2007, Sheaves et al 2017), making it quite challenging to provide unequivocal quantification of how a specific area of habitat will translate into a set amount of fish biomass. This could be because the importance of habitats is at larger scales – with the presence and connectivity across many habitat types ultimately being the crucial factor (Blaber 2007, Meynecke et al 2007). This could be why fragmentation and incremental habitat loss can erode fish biodiversity and production nonlinearly (Meynecke et al 2008). Despite the importance of habitats and expanding array of data collection methods (aerial photography, lidar, satellite imagery, acoustic sampling) routine monitoring is absent in many locations, with most habitat mapping still a largely opportunistic and research driven exercise. This may change as autonomous sampling becomes more established.

Future projections of biomass

In planning for the future and anticipating the likely consequences of management actions and development activities, there is high demand for means of projecting fish biomass. Information is required on short time frames of months to years – in which case species distribution and stock assessment models are often the method of choice – or on the order of decades, in which case models need to be forced by (or coupled to) trajectories of environmental change derived from general circulation models (e.g., changes in temperature, pH, salinity, oxygenation, current patterns or primary production) and scenarios of future human stressors (e.g., exploitation). In general, environmental conditioning is done via links to reproduction, growth, distribution of preferred habitat conditions and movement, while the human stressors act directly (via mortality) or indirectly (by further modifying availability of habitat or prey).

The further into the future and the broader the scale the more general the projection tends to be – especially as uncertainties and errors accumulate across scales. This is true for patterns of population change as well as the drivers used to force any model. The sources of uncertainty are many fold (Payne et al 2015): (i) parametric uncertainty (perhaps the most appreciated form of uncertainty, which is caused by sampling inadequacies and levels of natural variability); (ii) uncertainty to do with model initialization and internal variability (which reflects uncertainty pertaining to initial conditions and their ongoing influence on internal model interactions) (iii); structural uncertainty (regarding the model structure and assumptions); and (iv) scenario uncertainty (just what context setting decisions will be made – for instance what is the form of future emissions policies and practices). Borrowing lessons from the climate models used to generate the time series of future conditions that underlie the future projections suggests that uncertainties due to initial conditions and parameters are likely to dominate in the short term, model processes in the medium term and scenario details in the long term (IPCC 2013).

Additional data collection can help resolve some forms of uncertainty – particularly uncertainty around parameter values and initial conditions, and it can help resolve which of two models more appropriately represents dynamic evolution of a system. Considerable effort has been put into elucidating potential response functions – both in the field (Polosczanska et al 2016) and in laboratory settings (e.g. Doney et al 2009, Munday et al 2010), with appreciation for the need to consider multiple interacting stressors (e,.g Miller et al 2015, Boyd 2015) – although the sheer impossibility of the task of doing this for all species in realistic conditions (including acclimation and evolution; Munday et al 2013) means generalizations and approximations will be inevitable. Furthermore, simply gathering more data will not eradicate all uncertainty – models are system simplifications and may not contain all the processes important under future conditions; moreover, we do not know for sure what the future context will be (i.e. the exact scenario details). This is why the use of multiple modelling methods is strongly encouraged, particularly for projections beyond a few years. Even on the decadal scale – the scale most often of interest to resource managers - finely-detailed predictions remain elusive (though general patterns are more reliable), as much because physical models are still struggling to resolve the interplay of factors influencing climate variability on the decadal scale (Salinger et al 2016).

Valuing fish biomass

Measurement of physical stocks and flows is one element of the SEEA Experimental Ecosystem Accounting. The other key element is valuing the flows of ecosystem services to beneficiaries. The focus of this section is on valuing changes in fish biomass. However, a fish stock may be defined in multiple ways, and biomass should be interpreted as biomass of a type of fish, where type could mean species, market class, species-size combination, species-age combination, species-location combination etc. The change in fish biomass is a flow, while the either the standing or spawning fish biomass are stocks. Fenichel et al. (2018) outline three pathways that changes in a stock of natural capital, e.g., fish biomass, can generate real income flows or dividends.[3] These are as inputs to market production, inputs to household production that does not enter the market, and direct ecosystem income. The first two likely fall within the boundaries of the SNA, while the third may not. A fourth important pathway to consider is how changes in fish biomass impact value through ecological interactions such as food webs and through value chain interactions.

Inputs to Market Production

Fish biomass is valuable for its contribution to market-based products. A common approach to valuing fish for market production is *ex-vessel* prices, the gross revenue fishers receive when they sell their catch. Globally, the sum of sales is on the order of $80B (Munro; Arnason et al. 2009). This is useful starting point for valuing fish that enter the market. However, it is clearly an upper bound on the marginal value of increase in the standing stock of fish for two reasons. First, standing stock is not what is sold - captured dead fish are sold. Therefore, part of the *ex*-vessel price must include dividends from human capital (e.g., fishing knowledge), labor costs, dividends from produced capital (e.g., boats), and the costs of other variable inputs such as fuel. There is also a strong argument to made for subtracting out subsidies that the fishing industry may receive (Sumaila et al. 2008). Munro (2010) points out that when Arnason et al. adjust the global estimate of $80B by netting out production costs and subsidies, they arrive at -$5B, which is a net loss. This provides a quick accounting of the state of global fisheries, but locally some fisheries are doing well, producing positive net revenue, particularly those with property rights-based approaches (Lubchenco et al. 2016).

Ex-vessel prices are generally available where markets are well developed. Costs tend to be more difficult. First, when available, costs are often self-reported, and there may be reasons for inaccurate reporting of costs. Taxing authorities may be a reasonable source of cost data, but again there may be incentives for misreporting and such data are seldom publicly available. Second, costs are often considered private business information. In the United States the National Marine Fisheries Service has access to logbooks with cost data, but these data are proprietary. Similar situations exist in European countries, e.g., Poland. In some settings records simply may not be kept. Finally, there are substantial fixed costs in fishing. Allocating these costs to fish is a challenge.

The approach of summing *ex*-vessel price multiplied by quantity and subtracting of production cost including subsides is a reasonable first approximation. However, this is not actually the value of a change in standing biomass. A more precise value would focus on the marginal dividends (Fenichel and Abbott 2014). This would be change in net real income from a change in the fish biomass. Importantly, the behavior of fishers may adjust to changing biomass either directly or through

[3] Also see Fenichel and Hashida (2019).

regulatory changes, which can lead to nonlinearities. Hutniczak (2014) and Zhang (2011) provide examples of modeling production decisions that can help identify cost functions. However, Reimer et al. (2017) illustrate that generally this is more challenging than it seems because of the many margins fishers can adjust on (also see Smith 2012). Arriving at the marginal dividend requires modeling the human behavioral responses (or feedback rules) that drive production decisions. These feedback rules are often called the economic program or resource allocation mechanism (Fenichel and Hashida 2019).

The marginal dividends approach measures the value of a change in the stock biomass at a point in time, but does not account for user cost of the stock.[4] That is, the marginal dividends calculation ignores the services of the stock in producing future fish stocks. This value should also be included to ensure that the value of the change in stock balances with real income flows from the stock (Fenichel and Abbott 2014; Fenichel et al. 2018). Arriving at the asset price for standing biomass requires including biophysical data along with socio-economic data (see Fenichel et al. 2018 for methodology). In cases where tradable permit (or catch-shares) programs exist, then using the share price (adjusted to appropriate units) is probably reasonable. The share price is comparable to the Fenichel and Abbott (2014) asset price, but is only a true asset value if all users and services from the stock participate in the market. Furthermore, the price observed in the exchange market is conditional on all existing regulations and market imperfections - just like any other observed market price. A combination of prices between periods 1 and 2 (e.g., an average) should be used to measure the value of the change in the stock (Arrow et al. 2003; Fenichel et al. 2016), which is equivalent to the flow of services. The prices are not appropriate for measuring total value (e.g., price times quantity is not a valid index). Rather the average of price 1 and 2 times the difference between stock 2 and 1 is a valid measure of change in value.

Most commercial harvest is associated with food production, there is also a valuable for-hire recreational sector (Abbott et al. 2018). In the for-hire recreation sector the challenge of developing the economic program is clearer. The good sold is not dead fish, but it is an experience, where the standing stock of fish biomass is an important input. Nevertheless, different species or qualities of fish may be good substitutes, so defining the stock may be challenging, changes in any one stock may have a small to non-detectable effect on recorded transactions, while changes in some stocks may have large impacts.

Inputs to Household Production

Fish are an important input into the recreational activity of angling and other forms of recreational fishing (Hyder et al. 2018). The consumption good in this case is a

[4] If one is interested in cost-based accounting, then the focus should be on the user cost not expenditure.

leisure time experience. However, except in the case of for-hire charters, the recreational fishing experience is produced chiefly through household production inputs. Fenichel et al. (2013) review approaches to modeling angle behavior and connected measures of value from recreational fishing. Because the consumption good is the service, which requires other inputs (Boyd and Banzhaf 2007), it is important to assess the marginal value of a greater standing stock of biomass on the value of the consumption good, which is often well approximated by a change in consumer surplus.

There are various techniques for measuring the willingness to pay for leisure experiences not traded in markets. The most common are travel cost-based methods. These include traditional travel costs, various discrete choice models (sometimes called Random Utility Models), and discrete-continuous choice models (Phaneuf and Requate 2018; Parsons 2003). Such modeling exercises try to explain trip taking behavior. In recreational fishing models catch rates are usually included as a determinant of trip taking and catch rate is often assumed proportional to stock. There are two paths to valuing a change in fish stock with these models. First, there is an assumed value of a trip, and it is the change in the number of trips with respect to a change in catch rate (assumed proportional to stock) that provides the "value of a change in biomass." Alternatively, if the marginal utility of income influences trip taking, then dividing the effect of change in trip taking with respect to the stock by a change in trip taking with respect to income, provides the value a change in the stock (the units being dollars per fish).

These approaches could be used to estimate marginal dividends, which would neglect the value of fish for future fish production. To recover the full value of a change in stock, recreationally derived marginal dividends combined with biophysical processes and the economic programs should be used to estimate the asset price of fish (Fenichel et al. 2018).

Other factors influence trip taking. Some may not be completely separable from the value of the biomass. For example, whether for-hire or not, fishing can lead to changes in fish behavior that alter the quality of the experience, which may also impact the value of a change in fish biomass (Arlinghaus et al. 2016). Further research is needed to evaluate the relationship between changes in fish behavior and subsequent impacts upon recreational fisher valuation of the targeted stocks.

Another important form of household production is subsistence fishing. Here the good in not the experience but is the protein and nutritional value provided by the seafood. Subsistence is certainly important in developing countries, but may also remain important in developed countries (Cooke et al. 2018). In most developed countries, subsistence fishers may fish under recreational regulations. However, the mode of fishing and what people would do if they were not fishing may be strikingly different for subsistence fishers compared with recreational fishers. Subsistence may be an important service, but by its very nature tends be very hard to measure. Replacement methods for valuing subsistence fishing are only valid if fisheries would actually make the substitutions. It is also not clear that potentially high price, commercially caught fish is the correct substitute. Moreover, it is likely that the subsistence service that fish biomass provides is likely an inferior good.

For recreation and subsistence, freshwater fish stocks might be particularly important, and these stocks maybe very localized relative to marine commercial stocks. Freshwater stocks will likely be harder to value than more unitary marine stocks. This is because it can be harder for fish to disperse between localized ponds with localized regulations, whereas in marine systems regulations may be broadly similar, and fish may be able to disperse. Indeed, fish in ponds are usually considered separate populations while fish in the ocean are often lumped into large marine ecosystem populations.

Non-consumptive direct services

The biomass of fish might also provide direct service, which may fall into two categories. First, direct services could imply that a change in fish biomass directly makes people better or worse off in the current period. For example, if the knowledge that there are more fish simply makes people better off. Such existence values are part of real income (Krutilla 1967), but are very hard to measure. The most common approach to measuring such direct services are stated preference methods. These methods have come along way and can produce credible valuation measure of willingness to pay and willingness to accept (Johnston 2017). The challenge with these methods for SEEA is not the state preference approach per se. It is mapping scenarios to real changes in biomass. Most state preference surveys are designed to answer a benefit-cost question of the form, “How much would you be willing to pay for a program with specific characteristics that generates 5% increase in fish biomass.” What is need is a series of these discrete questions that can be mapped into actual observed changes. Furthermore, we might be suspect if 5% change is realized through some other means. Finally, there is the issue of rights. A willingness to pay frame suggests the public must buy the change, while willingness to accept gives the public the rights to have more fish and asks them to sell the right to a 5% increase in the stock. There multiple reasons that willingness to pay and willingness to accept may not be the same for non-market goods. These include lack of substitutes (Shogren et al. 1994) and uncertainty, irreversibility, coupled with limited opportunity to learn (Zhao and Kling 2001). Some studies are now using a mix of revealed behavior and stated preference to assess the value of environmenal services that provide household production and direct services jointly (Phanuef et al. 2013).

The second form of direct services may not actually be direct services at all, but simply services where the production is too complex to analyze. Particularly, for ecosystems this a challenge because biomass of a prey species may provide an intermediate service to biomass production of another species. Yun et al. (2017) show how ecological interactions can influence value of a change in fish biomass because of predator-prey relationships. In such cases, ignoring such relationships tends to overvalue increases in predator biomass and undervalue increases in prey biomass. Such interactions are also likely true for habitat interactions and fish biomass production (Bond 2017). Because these are thought of as intermediate goods, they may not show up in national accounts. However, there will be changes in asset values, which should balance changes in income. Importantly, change in prey biomass does not just impact the change in a predator through direct biophysical interactions, there is also a price effect (Yun et al. 2017; Fenichel 2018). Interestingly, Yun et al. (2017) find that increase in prey (predator) biomass can increases the value of a change in the biomass of predator (prey) creating a complementarity relationship, and the Yun et al. approach can measure the size of this effect.

Implications for SEEA Accounting

A range of approaches exist for estimating the physical stocks and flows of biomass from fisheries as well as for valuing those stocks and flows. As such, there is a need to provide a flexible approach based on individual country capabilities and data availability when providing guidance on the structuring of the accounts for this service flow. For example, many countries may have catch data, fewer may conduct regular trawl surveys, and even fewer may have habitat and ecosystem data needed to populate models relating habitat and ecosystem condition to biomass. On the valuation side, while *ex vessel* prices may be readily available, cost estimates may be less accessible, and models of fisher behavior even scarcer. This may require categorization of approaches from "least preferred" to "most preferred" based on the uncertainties inherent in either the physical flow estimation or the monetary valuation.

The production boundary issue remains an important consideration for this service as harvesting of fish, as noted above, requires inputs of physical capital and labor. Moreover, in the case of aquaculture, the specific nature of the aquaculture system in place will determine how analogous farmed fish are to other agricultural products. Separating out the ecosystem contribution for both wild caught and farmed fish stocks remains important for highlighting the value of ecosystem contributions to the final ecosystem service; in both cases, however, a focus on the final end product produced (e.g., fish for sale) would not be expected to alter current SNA estimates of value up to the unpriced production of wild fish that incur a real user cost.

While not the focus of this paper, the role of recreational and existence values in accounting will be an important issue to continue to address for fisheries. Both can be significant contributors to the value placed on fisheries biomass by people. SNA-based estimates solely using expenditures on fishing trips thereby may neglect an important source of wealth for coastal communities. Valuing the complex ecosystem dynamics where the presence of one species interacts with and affects the viability of other fish stocks will also be important for better highlighting the ecosystem contributions to the final produced product. Combining more refined ecological models with economic behavioral models can assist in this regard.

Addendum: Input Received From Expert Meeting

As discussed at the Expert meeting, we have prepared below an addendum with feedback received/discussion points raised at the Expert meeting, initial responding comments, and suggestions for future investigation where appropriate. Note that some comments refer to the revised logic chain shown below, which itself was a modification of the original logic chain to better reflect the notion that the spatial coastal/marine area would be the spatial ecosystem asset that supports maintenance of stock of fish biomass, which then generates flows of fish for various purposes.

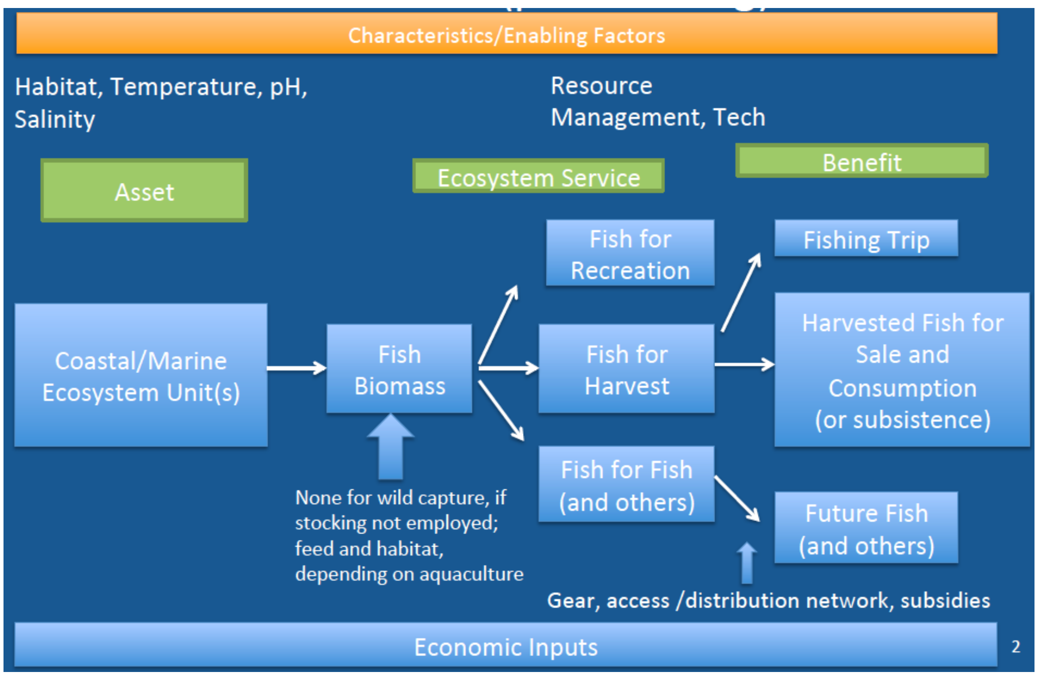

Figure A1. Revised logic chain

Paper is straightforward and focuses substantially on methods and approaches for data collection, modeling, and valuation. Connections to policy would be beneficial to consider as well.

Policy implications are important to consider in developing the ecosystem accounting approach for fisheries. To best understand the potential policy implications, there is a need to better delineate who the expected end users are for the ecosystem accounts and what indicators they would respond to for making policy decisions. This will require conversations with individuals in target agencies/ministries. For example, in countries where stock assessment is currently managed by existing commissions or panels (e.g., the Atlantic States Marine Fisheries Commission in the United States) on a regular basis, will the accounts affect policy decisions in a meaningful way? What additional indicators will come out of an ecosystem accounting approach that are different and useful beyond indicators already used in stock assessment decisions (e.g., SSB)? This question extends beyond fisheries and will likely be informed by planned work (noted at the Expert meeting) looking at actual integration of the accounts into decision making processes.

"Fish for fish" is an important service flow associated with fish biomass and should be considered separately.

This relationship can be further highlighted in work on the SEEA Revision related to fisheries. The ecosystem-based fisheries models included in this paper implicitly include the role of "fish for fish" (e.g., as a part of food web dynamics, in the modeling of population growth over time); it may be appropriate (to respond to the comment) to take the relevant models and highlight the flows that are the contribution of "fish for fish." As noted above in the discussion of policy implications, it will be important to determine who the likely SEEA end users for fisheries are to determine how much each ecosystem linkage should be disaggregated; for example, if all relevant fish stocks and their changes are tracked in a statistical accounting framework, it may fall to outside researchers (rather than within the accounting system itself) to investigate (using the models discussed) how changes in "fish for fish" may have impacted ecosystem dynamics.

There may be a role for using averages as indicators given the fluctuating nature of the fish stocks.

The specific "headline" indicators that come out of the accounts are important to consider. This again returns to the question of who will be the end users for the products of the accounts and the level of detail that will be needed for them to make decisions. Quarterly or annual reports are used for many existing economic indicators that are part of the statistical compilation for the System of National Accounts. Whether averages are appropriate will depend on whether or not the variability/volatility itself may be an important indicator for management decisions. Use of means versus medians versus tails of risk distributions has also become a point of debate, with current policy interest in extreme events related to climate.

Other impacts on the resource (e.g., degradation) could also be discussed.

Additional material on threats to the habitat associated with fish biomass would certainly be relevant to provide context for any changes noted in statistical accounting of ecosystem condition or stocks/flows of fish. For example, if fish harvests are seen to change, researchers may find it useful to also have a statistically rigorous collection of aquatic habitat types and their extent to better evaluate any potential correlation/causation. The challenge of including threats in a broad document such as this paper is that the threats are likely location specific. A better approach may be to compile the fisheries statistics and ecosystem condition/extent data rigorously, note changes in the data, and then facilitate localized investigation of potential causal mechanisms for the observed statistical changes on a case-by-case basis.

"Fish" should be differentiated as people will care more/less about certain fish stocks.

The paper focused on generally addressing the collection of data and assignment of value for fish biomass. Data availability will vary across species (which may, itself, represent relative interest by people/governments across species). The extent to which people "care" about certain fish stocks should also be reflected in valuation of the flows of fish to market or non-market uses.

There may be a benefit to breaking out more detail about the different industries involved in the beneficiary side of the supply chain.

Yes, the beneficiary supply chain should be traced to provide clear linkages to existing industries and products and the broader national economy. The NESCS approach developed by the US EPA may prove useful in this regard for connecting ecosystem accounts with existing national accounting structures and classification systems (e.g., NAICS in the United States).

Additional papers may be good to include as references such as recent work on wild seafood provisioning service indicators in Europe (Piet et al. 2017, Ecological Indicators) and work on small scale fisheries by the International Institute for Environment and Development (https://www.iied.org/no-hidden-catch-mainstreaming-small-scale-fisheries-national-accounts).

These have been noted.

The point made in the document about the differences across countries is an important one and should be kept in mind when developing guidance for the ecosystem accounts. How do we deal with lack of data or fisheries that are poorly managed or overexploited? Uncertainty in data collected is important to consider related to its implications for national account estimates that would be produced.

This is a key challenge in developing the ecosystem accounts and a compilation of case studies of the accounts (from data poor to data rich regions) should assist in evaluating challenges encountered and creating more streamlined approaches that can be used as a starting point in data-limited locations. Uncertainty, which is also present in data collected for standard national accounts, remains important to consider for ecosystem accounting as well; discussion with potential users of the data can elucidate thresholds and tolerance for uncertainties in the data.

The spatial aspect of the fish stock (where it is harvested versus its nursery area versus where it feeds, etc.) and how to spatially assign values associated with the fish stock asset and its flows is important to consider further. Associated with this topic, there *are likely "terrestrial" habitat types (e.g., marsh/mangroves) that are an important spatial element in the support of biomass stock and subsequent fish flows.*

This spatial aspect should likely be discussed as a joint effort by those looking at delineation of spatial units, sourcing of data for population of ecosystem accounts, and valuation of those accounts. The location of assignment of value could have important implications for prioritization of management and conservation areas. In coastal areas, the ecosystem accounting unit may likely incorporate adjacent terrestrial areas.

Fisheries biomass should be elaborated beyond its role in generating provisioning service flows (e.g., recreation, existence).

The original remit of the paper was to focus on provisioning services, but we have modified the logic chain to take the broader set of service flows into account. These service flows could form the basis of a separate document or a future expansion of the current paper.

Subsistence fisheries as a topic should be elaborated in greater detail.

We agree that further work is needed in determining how to treat subsistence fisheries in relation to ecosystem accounts and national accounting more broadly. There are existing approaches for production of goods for own consumption, particularly related to agriculture, that can be supportive in creating a crosswalk for this ecosystem service across the ecosystem accounting and national accounting spaces.

Future projections of fish stocks (and associated flows) and their intersection with valuation is an important area to consider and the discussion of this in the paper is helpful.

The role of future projections for fisheries-related ecosystem accounts and their application by end users is agreed to be an important issue to continue to investigate. Whether or not such projections are important in setting up the initial statistical accounts for ecosystems is an open question, but discussions with potential national end users could determine how in depth the SEEA Revision should be about providing guidelines for future projections. An alternative, similar to what is noted above, is to set up the statistical system and allow researchers to use the system to investigate potential future trends based on historical data and other relevant external factors.

Overlap with Central Framework can be more explicitly addressed.

The crosswalk between what is currently recommended and practiced for fishery accounts from a Central Framework perspective versus what would be needed for an ecosystem accounts perspective would be a valuable area for investigation. This could be a short summary paper highlighting what Central Framework accounts require for data, how those data overlap with data needed for ecosystem accounts, and gaps between data required for ecosystem accounts versus those required for Central Framework accounts. The spatial nature of ecosystem accounting for fisheries and its linkage to habitat (nursery, feeding, etc.) would be key distinguishing characteristics.

Discarded fish could be something to evaluate more in the future.

Discarded fish are a challenge both in estimating their quantity and in determining how to value these discarded fish as a flow from (and back to) the ecosystem (e.g., do they live or die?). More work is needed to determine how these would specifically be tracked in ecosystem accounting tables.

Challenges in using resource rents as a measure of value could be elaborated in greater detail. Total value added could be considered as an approach.

Resource rents may be problematic for indicating the true value of the resource because of their intersection with the management regime. Further thinking is needed to determine whether total value added is appropriate for compilation of an ecosystem account for fisheries.

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