# Stock assessment in inland fisheries: a foundation for sustainable use and conservation

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REVIEWS

# Stock assessment in inland fisheries: a foundation for sustainable use and conservation

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**Abstract** Fisheries stock assessments are essential for science-based fisheries management. Inland fisheries pose challenges, but also provide opportunities for biological assessments that differ from those encountered in large marine fisheries for which many of our assessment methods have been developed. These include the number and diversity of fisheries, high levels of ecological and environmental variation, and relative lack of institutional capacity for assessment. In addition, anthropogenic impacts on habitats, widespread presence of non-native species and the frequent use of enhancement and restoration measures such as stocking affect stock dynamics. This paper outlines various stock assessment and data collection approaches that can be adapted to a wide range of different inland fisheries and management challenges. Although this paper identifies challenges in assessment, it focuses on solutions that are practical, scalable and transferrable. A path forward is suggested in which biological assessment generates some of the

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critical information needed by fisheries managers to make effective decisions that benefit the resource and stakeholders.

**Keywords** Assessment tools · Fisheries management · Inland fisheries · Sustainable fisheries

## Introduction

Inland capture fisheries extract fish and other living organisms from surface waters inland of the coastline (Welcomme et al. 2010). Inland fisheries deliver nutritional security and income to hundreds of millions of rural households (Cooke et al. in press). They supply protein and essential macro-nutrients (Youn et al. 2014) and support livelihoods (Smith et al. 2005) through a diverse range of fisheries exploitation strategies (Welcomme et al. 2010). In 2008, inland capture fisheries produced an estimated 10 million tonnes of fish and crustaceans (FAO Fishstat 2010—see http://www.fao.org/fishery/statistics/software/fishstat/en). Inland fish resources also provide a wide range of other ecosystem services (Holmlund and Hammer 1999; Cowx and Portocarrero 2011; Lynch et al. 2016).

Inland fisheries are managed for a variety of objectives and by multiple means. The objectives are usually related to the types of use as well as socio-economic factors connected with the associated stakeholders. The means by which this is done include the management of exploitation (e.g., by catch, fishing effort or size limits), the management of fish habitat (e.g., river flow regulation, management of aquatic vegetation, etc.), and the use of fisheries enhancements (e.g., stocking of hatchery fish) (Welcomme et al. 2010; Arlinghaus et al. 2016). Effective management decision making requires knowledge of the state of the fishery relative to management targets or limits, and of the likely responses of the fishery to alternative management options. Providing such knowledge, typically in a quantitative form, is the aim of fisheries stock assessment (Hilborn and Walters 1992; Cowx 1996; Walters and Martell 2004). Use of assessments allows managers to be predictive rather than reactive with their actions, such as setting catch regulations and initiating enhancement or restoration measures. When assessment is conducted over long time scales (monitoring) such data can be used to identify factors related to changes in fishery productivity (e.g., environmental degradation or change, effort dynamics; Vaughan et al. 2001).

Perhaps surprisingly, many freshwater fisheries are managed without use of stock assessments, even where capacity for data collection and analysis exists. Management of such fisheries is likely to lose out on quantitative information that can support decision making on issues ranging from the sustainability of fisheries exploitation (Isaac and Ruffino 1996; Pitcher 2016), to the development of effective stocking and harvesting strategies for enhanced fisheries (Lorenzen et al. 1998a, b) or fisheries impacts of agricultural water management in river-floodplain systems (Shankar et al. 2005). Many inland fisheries scientists and managers are unaware of the power of stock assessment methods even in relatively data-limited situations and are unfamiliar with the methods, data requirements, and options for tailoring such methods to specific problems and contexts. Stock assessment experts, on the other hand, are often unaware of the particular issues involved in applying assessments in inland systems. Our aim with this paper is to break this impasse and inform inland fisheries scientist and managers about their options for stock assessment, and stock assessment scientists about the context of inland fisheries and its implications for stock assessment approaches.

The development and use of quantitative stock assessment methods has been spearheaded in large-scale, commercial marine fisheries that rely on extensive and long-term data collection, complex and skill-intensive analytical procedures, clear and uniform performance indicators, and frequent updating of assessments and adjustment of management measure to keep fishing within sustainable limits (Beverton and Holt 1957; Hilborn and Walters 1992). Even though most stock assessment methods are equally suited for use in inland and marine fisheries (Pitcher 2016), only a small proportion of

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inland fisheries are being systematically assessed. Reasons for this are likely to include the much smaller scale of most inland fisheries and the corresponding, relative scarcity of resources and expertise that can be applied to individual inland fisheries. However, inland fisheries are not simply equivalent to data- and capacity-poor marine fisheries, the assessment of which is a very active research area (Carruthers et al. 2014; Chrysafi and Kuparinen 2016). Rather, inland fisheries pose challenges, but also opportunities for assessment that are different from, and more diverse than those encountered in large-scale marine fisheries. These challenges relate to the nature and diversity of management objectives, anthropogenic impacts on freshwater habitats and fisheries, and tools for managing and enhancing fisheries. Traditional stock assessment methods may not address the range of management issues and approaches in fresh waters where fisheries exploitation is but one of many impacts (Cowx 1996; Welcomme et al. 2010; Cooke et al. 2014). Moreover, the small and confined nature of many inland stocks makes it possible to estimate their abundance effectively by mark-recapture or depletion studies, and the large number of individual fisheries means that comparative studies can be very effective means of quantifying management responses. This suggests that effective and efficient methods for assessment of inland fisheries can be developed if the characteristics of these fisheries are taken into account. In this review we focus on those characteristics that often make inland fisheries special, while also recognizing that some freshwater fisheries (e.g. in the Laurentian Great Lakes) are comparable to large marine fisheries and some small-scale marine fisheries share many of the characteristics identified here for freshwater fisheries.

We start out with a brief overview of stock assessment principles before outlining the particular characteristics of inland fisheries that often provide different context for assessment compared to large-scale marine systems. We then review and discuss the suitability of different stock assessment methods and approaches to data collection for inland fisheries. Finally, we outline approaches for selecting such methods and provide recommendations for the further development of such methods and their use in management.

## Stock assessment and management basics

### Stock assessment

In a nutshell, stock assessment entails the development of mathematical models of the fishery and their confrontation with data in order to determine how management practices (i.e., fishing input and output controls, stocking) and environmental forcing influence the state of fisheries. The role of models in fisheries assessment is threefold: they provide a clear conceptual and mathematical-statistical basis for synthesizing and interpreting information, allow the estimation of theory-based reference points such as Maximum Sustainable Yield (MSY) (which cannot be directly observed), and predict responses to different management options. Assessment models must be confronted with data in order to estimate stock-specific model parameters and management reference points. Essentially the confrontation of assessment model and data involves using the model to predict what data should be observed given the configuration and parameter values used in the model, comparing the predicted to observed data, and adjusting the parameter values and sometimes model configuration until a satisfactory fit is obtained. Modern 'integrated assessment' models can be used to predict and fit a wide range of observations for a single stock including time series of catch rates, age- and size-distributions. Other approaches may employ intermediate analyses to estimate some or all parameters external to the assessment model (e.g., growth parameters may be derived from biological sampling of the fish population and entered into a stock assessment model). Regardless of the details of assessment methods used, most large-scale commercial marine fisheries are assessed using complex assessment models and extensive, long-term data collection programs.

To illustrate the use of fisheries assessments and management reference points, consider the classic case of assessing exploitation of the fishery. An assessment would estimate sustainable yield and biomass curves: relationships between fishing effort (or fishing mortality), equilibrium yield, and equilibrium stock biomass (Fig. 1). Yield curves for single species stocks are dome-shaped while biomass curves decline monotonously with fishing effort. MSY, the maximum yield that can be harvested on an ongoing basis, is achieved by fishing at an effort level of $E_{MSY}$.

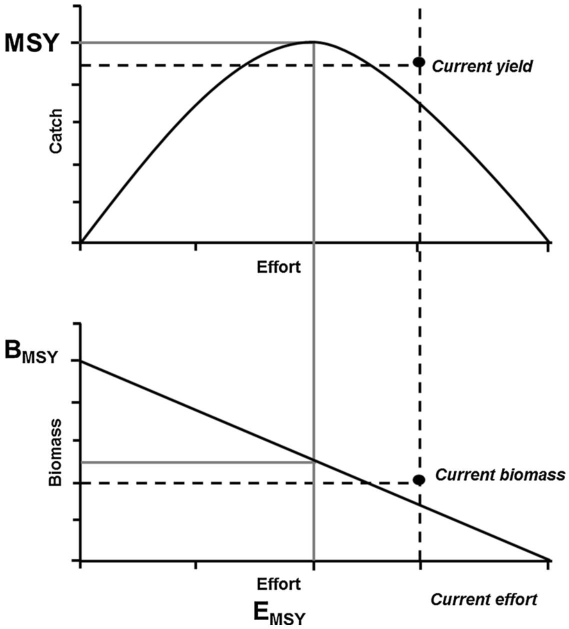



**Fig. 1** Relationships between fishing effort, yield and stock biomass in a single-species fishery. Equilibrium stock biomass (*lower panel*, *black solid line*) declines with increasing fishing effort while equilibrium yield (*upper panel*, *black solid line*) is maximised at intermediate levels of fishing effort. Maximum sustainable yield (MSY), the fishing effort at which MSY is achieved ($E_{MSY}$) and the corresponding stock biomass ($B_{MSY}$) are important management reference points in fisheries, indicated here by *grey solid lines*. The current status of the fishery is assessed relative to these reference points. In the example shown here, the current level of fishing effort is above $E_{MSY}$ and consequently, stock biomass is below $B_{MSY}$: the fishery is overfished and therefore the realized yield is less than MSY

The corresponding stock biomass is $B_{MSY}$. Both $E_{MSY}$ and $B_{MSY}$ are important fisheries management reference points, used to judge the status of a fishery by comparing actual effort and biomass to the levels that would achieve MSY. Actual catches and stock biomass may deviate from the equilibrium yield and biomass curves because, due to variation in stock productivity and fishing effort, fisheries are rarely at equilibrium. However, fishing at $E_{MSY}$ should return the stock to $B_{MSY}$ and catches to MSY in the long term. (Note that while $E_{MSY}$ can be used as an effort target regardless of the state of the fishery, MSY must not be used as a catch target when the fishery is overfished because in that situation, only catches substantially below MSY will allow the stock to recover to a biomass level where it can produce MSY).

Although both the biological basis of the MSY concept and its appropriateness as a management target have been subject to considerable controversy, MSY has proved an enduring cornerstone of fisheries assessment and management (Larkin 1977; Punt and Smith 2001).This is so because MSY estimates the fundamental biological potential (and upper limit to harvesting) of a stock and fishing at $E_{MSY}$ also tends to balance trade-offs between conflicting management objectives such as fishery access or employment and stock conservation (Hilborn 2007).

### Use of stock assessment in management

Management of most large-scale marine fisheries involves regular (in many cases annual) updating of assessments and catch limits (or other management limits or targets). Decisions about catch limits are made by deliberation in many fisheries, but may also be guided by formal 'harvest control rules'. Use of such rules has been associated with greater success in achieving fishery sustainability than the deliberative approaches (Walters and Martell 2004; Edwards et al. 2012). Overall fishery management strategies comprise the assessment methods, harvest control rules, and methods used to implement management measures (including enforcement; Punt et al. 2016).

### Ecosystem considerations

The ecosystem approach to (marine) fisheries management implies a need to account for environmental forcing on fish stocks and for all impacts of fishing activities (Pikitch et al. 2004; Rice 2011). This involves accounting for biological and technical interactions between the stock that is the focus of assessment and management and other stocks such as prey or predator species. Ecosystem considerations play an increasingly important role in marine fisheries management but have not, by and large, replaced single species approaches as the cornerstone of fisheries assessment and management.

## The context(s) of stock assessment in inland fisheries

Inland fisheries provide context(s) for stock assessment that often differ markedly from those of large-scale marine fisheries. These differences pertain to all dimensions of the fisheries systems: attributes of the resources, fishing techniques, enhancement measures, habitat and environment, stakeholders, markets and governance arrangements (Pido et al. 1996; Lorenzen 2008). Here we briefly outline these context(s) and their implications for fisheries assessment.

Inland fishery resources differ from marine fisheries in the scale and diversity of habitats and stocks and the number of distinct systems. The average inland fishery is about an order of magnitude smaller than the average marine fishery, and that translates into much lower levels of funding support that can be expended on monitoring, assessment and management of individual fisheries (Pitcher 2016). The collection of data sufficient for assessment is difficult for small-scale fisheries (Andrew et al. 2007) and it is recognised that assessment and management approaches need to be fundamentally different to those of large-scale commercial fisheries (Andrew et al. 2007; FAO and World Fish Center 2008). For one thing, small confined systems allow effective application of assessment techniques such as depletion sampling or mark-recapture methods for the estimation of abundance and biological parameters. Moreover, the replicated nature of inland fisheries (which often involve multiple similar lakes or streams) implies opportunities to gain efficiency through comparative studies and/or by focusing assessments on a smaller number of representative fisheries and transferring management advice to others in the replicated set.

Fishing techniques likewise differ, with a predominance of artisanal technologies that harvest relatively small numbers of fish per fisher or per boat. Many inland fishing techniques (traps, fyke nets, hook and line etc.) allow survival of fish that are captured but released, either for regulatory reasons or voluntarily. Catch-and-release recreational fishing is a predominant use of inland fisheries resources in many industrialized countries. Again, the use of artisanal methods implies that a large number of fishing units need to be sampled to obtain precise estimates of catches. Moreover, where release of captured fish is common, release rates and post-release survival must be estimated in order to assess removals by fishing.

Many inland fisheries are actively enhanced, for example through releases of hatchery fish, or improvements in habitat quality (e.g. control of aquatic vegetation) or connectivity (e.g. fishways). The level of enhancement activities in inland fisheries exceeds that in marine fisheries by a wide margin, reflecting the fact that technologies such as hatchery rearing of freshwater fish are well developed, the small scale and confined nature of many freshwater systems makes effective enhancement a realistic possibility, and governance control in freshwater fisheries is often high. Furthermore, fish and other freshwater organisms have been introduced widely beyond their native ranges, often establishing new fisheries and sometimes interacting strongly with native fish stocks and fisheries (Welcomme 1988; DIAS 2004; Gozlan et al. 2010). Where enhancements are common and may significantly impact on fisheries, data on the extent of these activities must be collected (e.g. stocking rates) and assessment methods must account for these interventions.

Fish populations and their productivity are inherently linked to the functioning and productivity of aquatic ecosystems (Lapointe et al. 2014). Freshwater habitats and ecosystems have been subjected to major anthropogenic changes including damming, habitat simplification for navigation, water extraction, acidification, eutrophication and pollution (Dudgeon et al. 2006; Strayer and Dudgeon 2010; Vörösmarty et al. 2010). As such, it is not surprising that freshwater fish are among the most imperiled taxa in the world (Richter et al. 1997; Dudgeon 2011). Assessing and possibly managing anthropogenic impacts on fisheries implies a need to collect data on such impacting factors and developing assessment models that can account for them. This is very different from large-scale marine fisheries in which environmental drivers are considered beyond management control and often accounted for simply as noise.

Stakeholders involved in inland fisheries management are often extremely diverse. Even direct users (primary stakeholders) may comprise subsistence, commercial and recreational fishers active in the same fishery. Compared to marine fisheries, subsistence and recreational use play a much larger role in inland fisheries. The importance of inland recreational fisheries grows and reliance on fisheries for food tends to decline with the regional level of economic development. These users may have very different, and often conflicting objectives for their use of the resource. Understanding these objectives and the related fisheries performance indicators is crucial to the design and assessment and management strategies that are relevant to the fisheries. In addition to direct users of fisheries resources, many inland fisheries have important 'secondary' stakeholders who influence habitat and environmental attributes and should be engaged in assessment and management activities where such influences are significant.

In the context of stock assessment, markets play a role primarily with respect to the extent to which caches are aggregated and can be sampled effectively at known locations. In many inland fisheries, landings and markets are spatially diffuse and often informal, making traditional data collection methods such as market sampling difficult to use.

Basic governance systems of inland fisheries include unmanaged open access (no management), managed open or only nominally restricted access, community management, and private use rights. Owing to the small-scale and confined nature of many inland fisheries, community-based and private rights-based management systems are more common in freshwater than in marine fisheries. The degree to which management systems use stock assessment information is highly variable. Few inland fisheries management systems are set up to adjust management measures frequently in response to assessments. Governance arrangements have implications for the spatial and temporal scales at which assessments are relevant to management.

As this brief survey has shown, multiple dimensions of inland fisheries systems have important implications for management objectives, management controls (the aspects that can be influenced by management), suitable performance indicators, assessment methods and options for data collection.

## Approaches to stock assessment of inland fisheries

A wide range of approaches are available for stock assessment in inland fisheries. Perhaps the central and most versatile approach is that of using single species population models in an integrated assessment framework. All other approaches are either simplifications of this approach (often informed by insights gained from analyses of population models) or extensions that include technical or biological interactions with other species and stocks. Here we provide an overview of approaches relevant to inland fisheries. Edwards et al. (2012) provide a more technical yet concise overview of many of these methods and full details and guidance for use can be found e.g., in Hilborn and Walters (1992), Hoggarth et al. (2006) and Haddon (2011). An overview of methods is provided in Table 1, while Table 2 gives examples of studies where alternative methods have been applied to different management questions.

### Indicators and reference points

An important but often overlooked aspect of selecting assessment methods is the consideration of performance indicators. Many discussions and theoretical arguments about fisheries management make reference to total catches and sustainable yields, and government and international agencies often collect catch data as the primary form of fisheries reporting. However, this does not mean that monitoring of catch levels and estimation of MSY are the holy grail of stock assessment (Walters and Martell 2002). Indeed, it is perfectly possible to assess the sustainability of fisheries exploitation without knowledge of total catches, for example by using size structure indices or comparing relative estimates of biomass under exploitation and unexploited reference biomass. Often, such indicators are far easier to measure with appropriate precision than total catches. Likewise, total catches are not necessarily required to assess the economic or social importance of fisheries. Contingent valuation approaches that measure willingness to pay for the recreational experience are the cornerstone of valuing recreational fisheries and do not require knowledge of catches. Likewise, the value of commercial fisheries can be deducted from sales records, and the importance of subsistence fisheries appraised from household consumption information.

As described earlier, many (but not all) assessment approaches involve the use of specific reference points to interpret information from indicators (e.g. stock biomass may be compared to stock biomass at MSY in order to judge whether a stock is overfished). Well defined and widely accepted reference points are used in many capture fisheries. However, fisheries enhanced by stocking of hatchery fish may require additional reference points in order to judge overall performance and impacts on the wild stock component (Lorenzen 2005). The issues become yet more complex in the context of habitat management or the

**Table 1** Stock assessment methods, their data requirements and outputs

| Method | Data requirements | Outputs |
|---|---|---|
| Single-species population models models | Time series of age and/or size composition data for the focal stock and fishery | Yield, biomass and other performance indicators for a wide range of management options |
| Single-species production or biomass dynamics models | Time series of catch and effort data for the focal fishery | Maximum sustainable yield (MSY), Effort at (MSY) |
| Multi-species population models | Time series of age and/or size composition data for the focal and interacting stocks and fisheries | Yield, biomass and other performance indicators for a wide range of management options accounting for technical and/or biological interactions |
| Ecosystem models | Life history and diet information for key species/groups | Ecosystem responses to fishing |
| Empirical yield and abundance models | Catch, effort, CPUE and trophic status indicators for multiple fisheries (representative sample) | Fisheries yield or fish abundance in relation to effort and/or trophic status |
| Observational studies | Catch, effort, CPUE and impacting factors, representative sample or specific study design | Effect of impacting factor on yield and abundance (may be biased due to confounding effects) |
| Management experiments | Catch, effort, CPUE and impacting factors, designed experiment | Effect of impacting factor on yield and abundance (unbiased if well designed) |
| *Simple indicators* | | |
| Trend analysis of indicators | Time series of catch, effort, CPUE for the focal fishery | Interpretation of catch, effort and abundance trends in terms of changing exploitation of environment |
| Catch size structure indices | Catch length composition for the focal fishery | Risk of overfishing, relative reduction in spawning per recruit |

See text for details and example applications

management of non-native species. Essentially, fisheries assessment theory has not grappled with the definition and evaluation of reference points for issues other than the sustainable exploitation of native species. This will be an important area for research and debate as stock assessments become more widely used in inland fisheries.

## Single-species population models

Population models (also known and age-structured, analytical or dynamic pool models) are based on a detailed representation of life histories, population dynamics processes, and impacts of fishing and possibly other anthropogenic influences. Because of this detailed representation, population models can be confronted with a wide range of data that contain information on population dynamics including catch rates (catch per unit of effort, a measure of relative abundance), size and age compositions. Such models can also describe a wide variety of factors and processes influencing population dynamics and management measures, and predict a wide variety of performance indices including indices such as catch size structure that are generally not available from simpler models. Population models have been used widely since the seminal work by Beverton and Holt (1957) and today are among the most important tools for marine fisheries assessments.

Many large marine fisheries are assessed based on all these types of data, often using integrated assessment models that fit different data series in their 'raw' form in order to characterize model uncertainty for use in the provision of management advice (Methot and Wetzel 2013). While the integrated modeling philosophy is sound and equally applicable to inland fisheries in principle, very few inland fisheries are sufficiently data-rich to allow integrated modeling without the use of very strong a priori assumptions. Most inland fisheries are limited in the types as well as

**Table 2** Example applications of different stock assessment methods to management issues encountered in inland fisheries

| Method | Fishing | Habitat | Non-native species | Fisheries enhancement |
|---|---|---|---|---|
| Single-species population models | Isaac and Ruffino (1996)<br>Cox and Walters (2002) | Welcomme and Hagborg (1977)<br>Kareiva et al. (2000)<br>Halls and Welcomme (2004)<br>Scheuerell et al. (2006) | | Lorenzen (2005)<br>Van Poorten et al. (2011) |
| Single-species production or biomass dynamics models | Kolding et al. (2003) | | | |
| Multi-species population models | Brown et al. (2005) | | | Jones et al. (1993)<br>Tsehaye et al. (2014) |
| Ecosystem models | Cox and Kitchell (2004) | | Moreau et al. (1993)<br>Cox and Kitchell (2004) | Kitchell et al. (2000) |
| Empirical yield and abundance models | Lorenzen et al. 2006 | Leach et al. (1987)<br>Ranta et al. (1992) | | Hasan and Middendorp (1998)<br>Lorenzen et al. (1998b) |
| Observational studies | Lorenzen et al. (1998a)<br>Almeida et al. (2009) | Nguyen Khoa et al. (2005) | Arthur et al. (2010) | Lorenzen et al. (1998a) |
| Management experiments | Swingle (1950, 1956) | Olson et al. (1998) | Arthur et al. (2010) | Swingle (1950, 1956) |
| *Simple indicators* | | | | |
| Trend analysis of indicators | Gerdeaux and Janjua (2009) | | | |
| Catch size structure indices | Welcomme (1999)<br>Booth (2004) | | | |

the amount of data that are available or can be collected, and different approaches to estimating model parameters are suitable for different types of data and vice versa. The traditional approach in marine fisheries has been to estimate fishing mortality patterns from fisheries catch-at-age data using virtual population analysis or catch-at-age analysis, while estimating natural mortality from comparative life history information (Hilborn and Walters 1992; Shepherd and Pope 2002). Growth and maturity curves were often estimated from biological studies on sub-samples of catches. In inland fisheries, mark-recapture methods provide excellent opportunities for the precise estimation of mortality components and growth curves because often, a substantial proportion of the population can be tagged (see below for further discussion). This means that the famously ‘data-hungry’ population modeling approaches need not be out of reach for freshwater fisheries if data collection methods are chosen wisely.

Conventional fisheries population models used in marine systems are generally designed to assess exploitation of harvest-oriented fisheries, but not habitat or other environmental influences; the efficacy of fisheries enhancement or restoration measures, or the dynamics of catch-and-release oriented recreational fisheries. Such concerns are, however, often central to the management of freshwater fisheries.

Increasingly, conventional fisheries population models are being extended and modified for the assessment of such management issues, including interactions between hydrology and fisheries in river-floodplain systems (Halls and Welcomme 2004); impacts of stocking hatchery fish into natural populations (Lorenzen 2005); and the regulation of fisheries with large catch-and release components (Cox et al. 2002; Coggins et al. 2007; Dotson et al. 2013); or the control of invasive fish species (Brown and Walker 2004) . To achieve this, conventional process assumptions (e.g., that recruitment accounts for all environmental variability and density-dependence in the population; that growth and mortality patterns in the recruited stock are constant) need revising. For example, the dynamics of fish stocks in river-floodplain systems are characterized by extreme seasonal variation in mortality rates that affect both juvenile and recruited fish (Halls and Welcomme 2004). For many inland fisheries assessment problems, models must be able to integrate demographic processes with environmental influences and accommodate the many other processes deleterious to populations such as larval mortalities including fishery take (as a component of population mortality (see Koehn and Todd 2012).

In general, fish population models are designed to describe the dynamics of numerically large populations that are not extremely depleted. Populations that are numerically small (i.e., number less than a few hundred individuals or are depleted to a few percent of their unexploited abundance) are subject to particular threat processes that can prevent recovery and possibly lead to population extinction. Such processes include Allee effects (where reproduction is adversely affected by low abundance) and demographic stochasticity (where random mortality of the remaining individuals alone can lead to extinction when all happen do die within a short period of time), and other ecological and genetic processes. Population Viability Analysis models (see Burgman et al. 1993) allow for environmental and demographic stochasticity by incorporating variation in the survival and reproduction of individuals at each stage (Akçakaya 1991). An inland fisheries example of a PVA is Vincenzi et al.'s (2008) assessment of factors controlling the population viability of endangered marble trout populations.

The advantage of population models over empirical and production models is that they explicitly represent population age and/or size structure and key processes in stock dynamics and allow, at least in principle, the assessment of size-based regulations, the disentanglement of confounding impacts and assessment of management options that affect particular life stages. Specific aspects of population dynamics (e.g., reproduction, survival and growth) can be affected by anthropogenic stressors, such as habitat degradation, pollution and exploitation (Power 2007). Whilst these aspects can be included (Koehn and Todd 2012) in many instances accurate causal relationships between the stressor and its impact on the life stage are lacking and such effects are best assessed at the population scale. As an example, Power (2007) used stock-recruitment data for a hypothetical brown trout population to show that 10 % entrainment mortality resulted in a 21 % decline in the equilibrium population size. Significantly, the stock-recruitment model showed that the impact of entrainment on the population was more than proportional to the increase in mortality. Another assessment method, production foregone, measures the loss of production of biomass (rather than numbers) from entrainment mortality using instantaneous growth rate parameters (Power 2007). Finally, model development can also provide a mechanism by which up to date (including unpublished) local fishery and biological knowledge and beliefs can collaboratively be incorporated from both scientists and stakeholders, thus providing a measure of agreement and 'ownership' of the model and subsequently improving the likelihood of support for fishery management strategies (Koehn and Todd 2012).

### Single-species production or biomass dynamics models

Production models lump growth, mortality and reproduction/recruitment processes into an overall process of biomass production, thereby reducing model complexity and data requirements, but also the range of anthropogenic impacts, management measures and performance indices that can be evaluated. Production models are based on mathematical descriptions of the relationship between net rate of biomass production and abundance relative to carrying capacity in fish stocks. Carrying capacity is the equilibrium biomass that can be indefinitely supported by the resources available (mainly available food and space; Shuter 1990; Hayes et al. 1996). When stock abundance is reduced below carrying capacity by fishing, the rate of production increases due to reduced competition for limiting resources. Different production models are in use that assume different functional relationships between production and abundance, e.g., linear (Schaefer model), logarithmic (Fox model), or a more complex flexible form (Hilborn and Walters 1992). Net biomass production (i.e., production rate x biomass) tends to be highest when stock biomass is reduced to around 35–50 % of carrying capacity and it is the biomass production in this state that equals the MSY that can be harvested (Hilborn and Walters 1992).

Despite of the relative simplicity of production models, applications in inland fisheries have been limited. Examples include assessments of Lake Superior whitefish (Jensen 1976) and the small pelagic kapenta (Limnothrissa miodon) in Lake Kariba (Kolding et al. 2003). The most advanced application of production models is in the form of biomass dynamics fisheries models. These models are fitted to time series of catch and effort data to estimate parameters of the underlying stock-production relationships from which to derive management reference points such as MSY and effort at MSY. Such models typically require long time series of data (Hilborn and Walters 1992), but may be used in data-poor situations when combined with prior estimates of the maximum production rate and/or the catchability coefficient (Gedamke et al. 2007; Medley 2009); or applied in an explicitly hierarchical framework (see below).

## Replicated systems and the use of external information in population and production models

Both population and production models use biologically meaningful parameters that are not entirely stock specific but reflect features of species (e.g., growth, fecundity, mortality) or ecosystems (e.g., production ecology) and therefore can be at least partially generalized. There is a long history in freshwater fisheries assessment of using comparative generalizations, for example in empirical yield models based on relationships between primary production or morphoedaphic factors and fish production; or in population dynamics models using natural mortality estimates derived from comparison with unexploited stocks of the same species. The development of Bayesian statistical approaches to fisheries assessment has greatly expanded the scope and rigour of using external information (Parent and Rivot 2012). Hierarchical Bayesian models (also known as random effects models) facilitate transfer of information between different units (e.g., stocks of the same species in different rivers) and therefore allow data-poor units to 'borrow' information from data-rich units. For example, Wyatt (2002) used a Bayesian hierarchical model to estimate fish abundance in multiple sites sampled with varying intensity including single pass and multiple pass electrofishing. Single pass fishing provides only relative abundance while multiple-pass sampling provides absolute abundance estimates. Hierarchical modeling improved the precision of an overall regional abundance estimate and also provided estimates of absolute abundance for sites sampled only by single-pass electrofishing. In a more complex application, Prévost et al. (2003) estimated recruitment in data-poor Atlantic salmon stocks through hierarchical modeling of data from multiple stocks including a set of data-rich reference stocks, using latitude and riverine area accessible to salmon as covariates.

An important implication of the availability of hierarchical and other comparative methods for the design of data collection programs is that it is often advantageous to concentrate scarce resources on sampling some representative units with high intensity while covering others with lower intensity and then transferring information, rather than covering all units with equal but more limited effort (as is often done in monitoring programs). Consideration of the landscape context of fisheries provides further avenues to greater assessment efficiency (Lester and Dunlop 2003; Lester et al. 2003; Hansen et al. 2010).

A huge amount of biological data on many fish species (and populations within a species) has been collated in databases such as FishBase (Froese and Pauly 2014); the RAM Legacy Stock Assessment Database (Ricard et al. 2012); the Multi-state Aquatic Resources Information System (MARIS) (Beard et al. 1998) and Fisheriesstandardsampling.org (Bonar et al. 2015). This means that, even in the absence of biological information on a particular species and location, it is often possible and easy to obtain biological information for the same or a closely related species in other locations and to use such information for initial assessments. Moreover, meta-analyses of large amounts of biological data have led to many useful generalisations about fish life history parameters such as natural mortality rates (Pauly 1980; Hoenig 1983), the level of recruitment compensation or "steepness" of the stock recruitment relationship (Myers 2001) and life history strategies that affect population variability (Winemiller 1989; Winemiller and Rose 1992; Rose et al. 2001; Olden et al. 2006).

## Multi-species and ecosystem models

Whilst the single-species approach remains central to the assessment and management of most fisheries that are systematically assessed at all, there has been an increasing emphasis in fisheries research and policy on ecosystem-based fishery management approaches (e.g., Link 2002; Pikitch et al. 2004; Rice 2011; FAO 2012). This emphasis arises because many fisheries impact on species other than their primary target, and this can lead to community and ecosystem-level impacts such as the cascading effects often observed in temperate and boreal lake fisheries (Carpenter and Kitchell 1996). Moreover, in species-rich tropical systems, many fisheries harvest multiple species quite indiscriminately such that it is impossible even in principle to identify single 'target species' (Welcomme 1999). An additional reason for taking a broader approach is that fishing is only one of many ecosystem services being provided by inland waters. Fishing may impact or compete with many other services requiring the use of water (e.g., maintaining biodiversity, hydropower generation, agricultural irrigation; Cowx and Portocarrero 2011; Koehn 2015). For these reasons, ecosystem-based fisheries management calls for indicators and models that measure impacts on the ecosystem, in addition to indicators directly related to harvested stocks (Rochet and Trenkel 2003).

It should be recognized that ecosystem-based fisheries management can be supported by a variety of assessment approaches and does not necessarily require full ecosystem models. For example, population models can include environmental covariates in the stock-recruitment function (Zhao et al. 2013) or be set up to account for concurrent harvesting of other species by the same gear (Jensen 1991). Indeed it has been argued that models of intermediate complexity (for example, those representing only a sub-set of species and interactions) often hit the 'sweet spot' where models are complex enough to capture important interactions, yet simple enough not be subject to great parameter uncertainty (Collie et al. 2014).

For the purpose of assessment and management, it is useful to distinguish between technical and ecological interactions in multi-species fisheries. Technical interactions that arise from harvesting of multiple species (exploited species as well as 'bycatch' species which may be fully protected) with the same fishing gear are important in many fisheries. Technical interactions can be assessed by modeling the joint harvesting of multiple, biologically independent stocks with interconnected (dependent) harvest rates (Hilborn 1976; Jensen 1991; Hoggarth and Kirkwood 1996). Such assessments provide information on trade-offs involved in joint harvesting, including species at risk from overfishing as bycatch.

Ecological interactions that arise from predator-prey interactions of harvested species with other, harvested or non-harvested species are assessed using food web modeling (Pauly *et al*. 2000; de Roos and Persson 2013). Theoretical and empirical studies suggest that complex, cascading interactions between harvested species and their predators and prey can arise particularly in relatively simple food webs with few interacting species (such as those found in boreal lakes), but are not commonly observed in complex webs with many interacting species (such as those found in tropical systems) (Walters and Martell 2004; Frank et al. 2007). Many different food web models have been proposed and used in primary research, but only a few have been widely applied in fisheries management contexts. The most widely used are the Ecopath model and its companion models to predict stock status changes in time (Ecosim) and space (Ecospace) (Christensen and Pauly 1992; Pauly et al. 2000). Matsuishi et al. (2006) provide an example in Lake Victoria fisheries where the potential impact of increased fishing effort on the Nile perch stocks of the lake are derived. The Atlantis model (Fulton et al. 2011) has also been widely used in marine systems. In contrast to Ecopath and its derivative models, Atlantis accounts for the effects of nutrient cycling on primary production, a feature that may facilitate assessments of environmental change in freshwater systems. Both Ecopath/Ecosim and Atlantis require a host of detailed biological information which even in well studied, large marine systems can be challenging to assemble and will rarely be available in freshwater systems. One way of dealing with this issue is to 'populate' the model using comparative information from databases in addition to system-specific information (Christensen et al. 2009). Another approach is to build food web models on the basis of size-based interactions independent of species identity, as implemented in the Osmose model (Shin and Cury 2001).

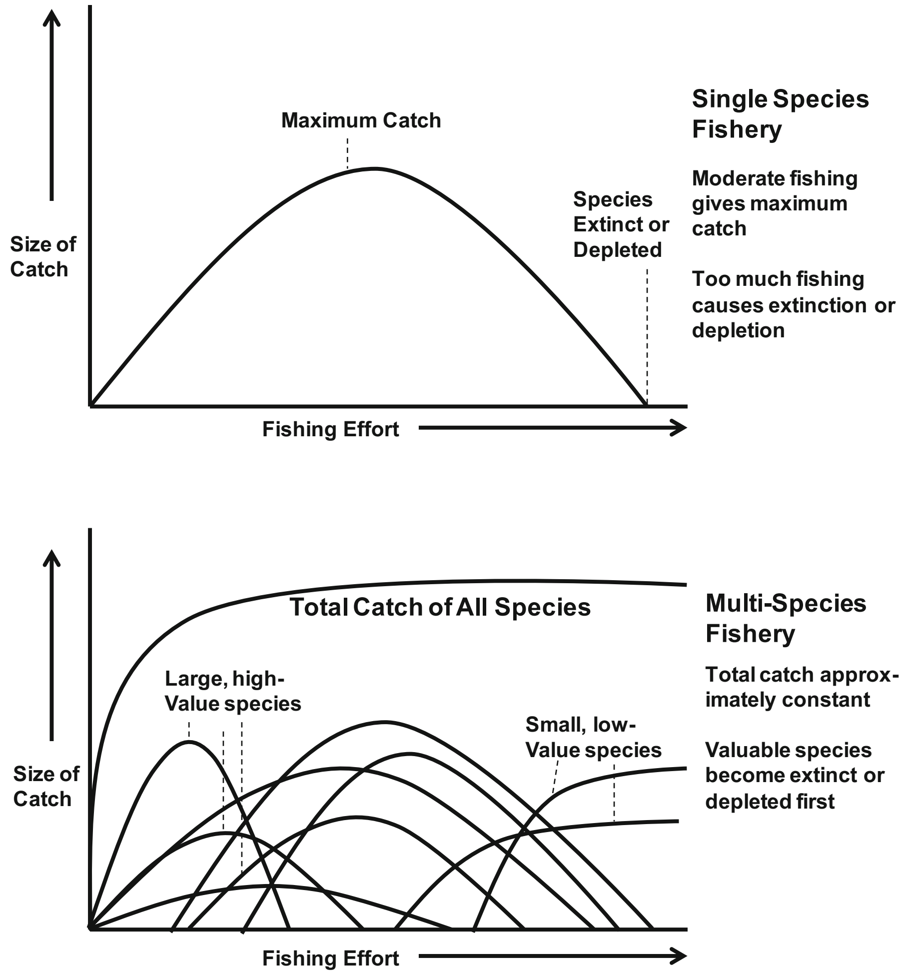
In inherently multi-species fisheries, which are typically small-scale in developing countries, assessments need to account for the inherently multi-species and multi-gear nature of fisheries (Welcomme 1999, Kolding and van Zwieten 2014). Assessments of such fisheries need not be more complex than single-species assessments when aggregated catches and simple community indicators are considered (e.g., Welcomme 1999; Lorenzen et al. 2006). In such fisheries, as fishing effort increases, shifts in the size and species composition of the harvest occur for several reasons. Stocks of large, slow growing species become heavily fished and their contribution declines even at moderate levels of fishing effort, fishers may target smaller fish as the large fish are depleted, and the productivity of stocks of smaller species may increase as they are released from predation by larger species. Despite the complexity of these 'fishing down' processes, they typically give rise to a fairly simple, asymptotic relationship between species-aggregated yield and fishing effort in multi-species fisheries, rising rapidly when effort is low, but then reaching a plateau that is sustained over high levels of fishing effort (Welcomme 1999; Lorenzen et al. 2006). The high yields at high effort levels are a result of shifting to the smaller more productive species that are lower in the trophic web (Fig. 2) and adaption of fishing methods. This decline in mean size (and hence mean trophic level) of the harvest has been interpreted as a negative signal (Welcomme 1985, 1999; Pauly et al. 1998). A decline in mean size of harvested fish is expected while the fishery develops (Tweddle et al. 2015). Nonetheless, fisheries yields tend to be sustained because the systems (especially floodplain systems) tend to be resilient and the fish assemblages, and thus fisheries, adapt to the changing conditions (Tweddle et al. 2015). Recently, Kolding and van Zwieten (2014) proposed that structural changes could be avoided if all species were harvested in proportion to their productivity – an approach known as 'balanced harvesting', the realism and utility of which is however subject to great controversy (Froese et al. 2015).

### Empirical yield or abundance models

Empirical yield or abundance models represent statistical relationships between fisheries response variables (e.g., harvest, abundance) and explanatory variables such as fishing effort, primary productivity or the morphoedaphic index and provide models of fish production and potential yield. Typically these models are developed using data for a representative sample of fisheries. Empirical models can be developed for single species or aggregated multi-species complexes and may be used to predict regional, national or global yields from models developed for representative samples of fisheries (Cowx 1996). This empirical approach plays a much greater role in understanding key drivers in inland fisheries relative to marine ones, owing to the large number of "replicated" inland systems and a high level of variation in explanatory variables among them. These features are often seen as challenges from the perspective of applying assessment methods that require detailed information for each system, but they also provide opportunities for using comparative empirical approaches that may be equally informative. Empirical models have been used to describe how fisheries yield or fish abundance responds to variation in fishing effort (Bayley 1988; Lorenzen et al. 2006), morphometric and nutrient or chlorophyll concentrations (Ryder 1965; Ryder et al. 1974, Henderson and Welcomme 1974; Ryder 1982; Schlesinger and Regier 1982; Bachman et al. 1996; Shuter et al. 1998; Lester et al. 2004; Kolding and van Zwieten 2014; and other environmental factors (Hasan and Middendorp 1998; Lorenzen et al. 1998a, b). Most empirical modeling approaches use statistical models, but some use alternative methods such as neuronal networks (Laë et al. 1999).

These empirical models are especially useful because they predict potential yield from lake/river characteristics that are easily measured (e.g., river length; Welcomme 1976), derived from existing maps, or available satellite imagery (see Fig. 3). GIS methods allow for calculation of the surface area of lakes and rivers within specified regions. Maps of topography and surficial geology can be used to approximate water depth, nutrient levels and temperature regime. Thus, sufficient data are available for approximating potential fish production in any region of the world (e.g., Minns 2009). Model predictions may not be so precise at small scales (e.g., individual

**Fig. 2** Yield-effort relationship for species-aggregated yield in multi-species fisheries, compared to single-species fisheries. Multi-species fisheries are composed of multiple stocks that differ in the level of fishing effort they can withstand. As overall fishing effort increases, the contribution from long-lived slow-growing species declines and the contribution of short-lived fast-growing species (which can withstand higher levels of fishing) increase. Aggregated yield from such fisheries tends to be insensitive to fishing effort while species composition changes and overall diversity of species harvested tends to be highest at intermediate effort levels (Modified from Hoggarth et al. 1999)




lakes), but precision improves when individual estimates are aggregated at larger scales. Although more research would be beneficial to validate and improve these models, they can be useful to obtain initial estimates of potential fish yield in a region or large water body. This information provides a basis for estimating the potential economic value of inland fisheries if ground-truthed against actual species catch statistics, and evaluating trade-offs with other ecosystem services.

Empirical yield models can be used to quantify the effects of management measures only of information on such measures is included in the explanatory variables (and there is sufficient contrast in the levels of such variables to allow estimation of the relevant coefficients). Where such management variables are not included in the model, deviations of individual fisheries from the mean relationship are sometimes interpreted qualitatively to identify fisheries that are "underperforming" and may benefit from greater management efforts (Leach et al. 1987).

Development of simple proxies to estimate fisheries production began many decades ago [e.g., morphoedaphic index (MEI), Ryder 1965; Schlesinger and Regier 1982] and there is renewed interest in further refining such tools. There is also promise for adopting landscape approaches to assessment (which would interface with landscape approaches to management) where there are simply too many water bodies to assess using traditional on-the-ground fish sampling strategies. There is continued need for both fisheries-dependent and fisheries-independent data streams as well as ways to combine the two approaches to yield more comprehensive understanding. In addition, methods to improve the accuracy and consistency of catch data along the lines of the Big

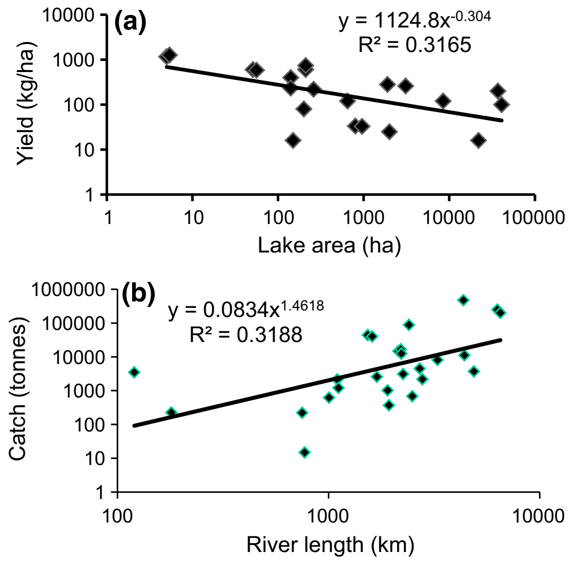



**Fig. 3** Models of fisheries yield per area in relation to lake area (**a**) and total yield in relation to river length (**b**) in African rivers. Fisheries yield per area positively correlate with basin area and river length

Numbers project (http://www.worldfishcenter.org/resource_centre/Big_Numbers_Project_Preliminary_Report.pdf) are required. This attempts to fill the gaps in information sources and improve on the quality of the data.

## Observational studies

Observational studies differ from empirical yield or abundance models in that they tend to be designed deliberately to answer certain very specific questions. Observational studies may involve the deliberate selection of impacted and otherwise similar control fisheries, for example, rather than observations at a randomly selected sample of fisheries (Eberhardt and Thomas 1991).

Observational studies can be applied to virtually any assessment question (e.g., sustainable harvest levels, impacts of management input/output control models, effectiveness of enhancement or restoration measures) An obvious drawback of observational studies is that they can only be used to assess the impact of factors for which empirical data exist (or can be collected). Similar to empirical models in many respects, observational studies have been used to assess impacts of specific anthropogenic habitat modifications or fisheries management measures (e.g., Nguyen Khoa et al. 2005), in stream habitat enhancements on riverine fish populations in the UK (Pretty et al. 2003), or co-management agreements on floodplain lake fisheries in the Amazon (Almeida et al. 2009). Such studies may be viewed as applications of empirical modelling with binary explanatory variables (presence/absence of an anthropogenic impact or management measure). Multiple observational studies can also be combined in meta-analyses of management measures, as demonstrated in Wilde's (1997) analysis of the effectiveness of length limits in bass fisheries management.

Observational studies (and empirical models) are based on observational rather than experimental data, which means that the observed levels of explanatory variables in the different systems may have been influenced by human behaviour or natural processes related to the response variable rather than being allocated at random as would be the case in designed experiments. This can lead to biased assessments when response and explanatory variables are confounded. For example, if no-take zones are placed in areas where fish abundance and diversity are already high, then a subsequent observational study could incorrectly attribute higher abundance and diversity to the establishment of no-fishing zones. Likewise, fishing effort in different waters may be related to differences in innate productivity, or enhancement measures may be taken in more degraded habitats. It is important to consider such possible confounding factors and where possible account for them in empirical studies.

## Experimental management

The replicated nature and relative ease at which particularly small-scale freshwater fisheries can be manipulated means that experimental management is far more feasible in freshwater than in marine fisheries. Experimental approaches to fisheries assessment and management were pioneered for small impoundments by Swingle (1950, 1956). More recent examples of experimental approaches include the experimental harvesting of lake trout populations to estimate biological responses (Healey 1980) and assessment of impacts of stocking of non-native species in small water bodies in the Mekong region (Arthur et al. 2010). There may also be the opportunity to develop relationships between habitat factors and the fishery for some species (e.g., macrophyte removal and fish growth; Carpenter et al. 1995; Olson et al. 1998). Experimental management differs from observational studies underlying many empirical models (see above) in that treatments are randomly allocated to study units minimising biases that can arise from observational studies. An excellent example of the power of experimental management in small-scale fisheries is provided in Castilla and Defeo (2001). Although focused on coastal shellfish, many aspects of the approach are readily transferred to inland fisheries. Design considerations for management experiments are outlined in McAllister and Peterman (1992) while MacGregor et al. (2002) provide a detailed example of experimental design evaluation.

## Simple indicators for rapid appraisal

It is often possible to obtain at least a rough indication of the status of a fishery from the use of simple indicators, even when lack of data, time, or assessment skills preclude the use of mode complex approaches. By "simple" we mean that the indicators can be used quickly and easily in field situations, not that their scientific basis is necessarily simple. Simple descriptive variables that are commonly available from fisheries-dependent or independent data, such as catch, effort and catch-per-unit-effort (CPUE), species diversity, population structure (*e.g.,* proportional stock density), mean fish length, and fish condition can be used as surrogate measures for biological assessment of the fish community (Rochet and Trenkel 2003). Mean length in the catch is a well-established correlate of fishing mortality (exploitation level) at the population level (Beverton and Holt 1957; Ricker 1963). Such indicators can be developed at the population (single species) or community level and they typically rely on either, inter-temporal comparisons of aggregated indices such as total catch or CPUE, or on information about the size structure of catches (or both). Population level indicators are useful where fisheries targeted at individual species can be identified and managed. Community-level indicators are the only realistic option where many species are harvested and their management cannot be separated (*e.g.,* in inherently multi-species tropical fisheries) and may also be used to provide additional community-level information in fisheries where management is single-species focused.

Where time series of CPUE data (an index of relative abundance) are available, decreases in CPUE of individual species may indicate that catches are exerting an influence on stock biomass, and a declining trend is of particular concern if catches are stable or increasing (Rochet and Trenkel 2003). At the community level, differential trends in CPUE of individual species or guilds with different life history characteristics can be indicative of exploitation-induced changes in assemblage structure, but species-aggregated CPUE tends to be relatively inert and may respond to exploitation in a non-monotonous manner (Lorenzen et al. 2006).

Data on size composition of catches are easily collected and can provide qualitative and quantitative information on exploitation levels without a need for time series, as long as basic life history information (growth, size at maturity, natural mortality) for the species is known. Froese (2004) proposed three intuitive, size-based indicators that can be used to provide a semi-quantitative indication of the risk of overfishing: (1) percentage of mature fish in catch, with 100 % as target; (2) percent of specimens with optimum length (i.e., the length that maximizes yield per recruit) in catch, with 100 % as target; and (3) percentage of large or old highly fecund female "mega-spawners" in catch, with 0 % as target and 30–40 % as representative of reasonable stock structure when no upper size limit exists. Hordyk et al. (2015a, b) provide a method for estimating the Spawning Potential Ratio (SPR), a measure of relative spawning stock depletion, solely from length composition data and basic life history information. Life history parameters required for use of size composition -based indices should normally be estimated for the populations being assessed, but comparative information on other populations of the same or closely related species may be used to provide rough estimates when stock-specific information is not available. Fishbase (Froese and Pauly 2014), the world's largest database on fish, contains life history data for several thousand fish species and also calculates the corresponding optimum length for exploitation required for use of Froese's (2004) index. Per-recruit models have been widely used for freshwater assessments. The outputs are for example SPR (40 % unfished) and Fsb40 (the corresponding fishing mortality to maintain 40 % spawning biomass per recruit). The latter is widely used as proxy for $F_{MSY}$.

Per-recruit models are also useful to evaluate size limits, slot limits or optimal mesh sizes in gillnet and seine fisheries. Inland fisheries examples can be found in the work by (Booth 2004; Weyl et al. 2005). At the community level, species-aggregated length composition or mean length can likewise be used as an indicator of exploitation, but no corresponding community-level reference points have been defined (Welcomme 1999).

In smaller freshwater systems it is relatively easy to obtain absolute estimates of biomass from depletion or mark-recapture methods (see below). A rough estimate of potential yield can then be obtained by multiplying the estimated stock abundance with the production-biomass (P/B) ratio. The P/B ratio is equivalent to the maximum sustainable rate of exploitation of a fish stock or community, hence estimates of the P/B ratio and the standing stock can be used to make a rough determination of sustainable catch limits (Bagenal 1978; Downing et al. 1990; Mertz and Myers 1998).

There are many other approaches to assessing fisheries in 'data poor' situations that have been developed with marine fisheries in mind but can also be applied in freshwaters. Recent reviews include Carruthers et al. (2014), Newman et al. (2014) and Chrysafi and Kuparinen (2016).

## Data for stock assessment

Data collected from fisheries for assessment purposes may include time series of species-specific or aggregated catch and fishing effort as well as length and possibly age composition. Such data may be collected from the operational fishery (fishery-dependent data) or from a research fishery designed for scientific data collection (fishery-independent data). In designing data collection and monitoring programs for inland fisheries, it is important to consider the types of assessment models that may be used and the assessment information they can provide; the data and information needs of these approaches; and the opportunities for meeting these needs through direct data collection and the use of comparative approaches. An overview of data collection methods and their advantages and disadvantages is given in Table 3.

### Fisheries dependent sampling

Appropriate methods for fisheries dependent sampling depend strongly on the nature of the fishery. Commercial fisheries with major landing sites, for example, can be sampled well with port/landing site surveys while diffuse subsistence fisheries may require household surveys. In general, it will be possible to develop appropriate surveys for almost any fishery as long as its specific conditions are taken into account. A good overview is provided in SEAFDEC (2005).

#### *Logbooks*

Logbooks are fishing trip or day records provided by fishermen, typically as a part of formal reporting requirements (but see voluntary reporting below). Logbooks are a good way of obtaining detailed and specific data on fishing activities and catches, but rely on the fishermen's willingness and ability to provide such records.

#### *Surveys at landing sites*

Landings, or landing sites, refer to areas or structures where catch may be unloaded and/or temporarily stored prior to being taken to market or being processed (Medina Pizzali 1988). Data acquired at landing sites are particularly valuable in lakes or large rivers where fish are sold through a limited number of sites immediately following harvest. Potential data variables include the quantity, species, weight, length and maturity of fish landed by the fishery (for example, see Jiddawi and Öhman 2002). Landing sites also present opportunities for cataloging gear types in use. Landing statistics are less applicable in fisheries where landings and sales are dispersed across the landscape, rather than being channeled to discrete locations, and where subsistence fishing and/or illegal fishing, especially of undersized fish, is prevalent. In these situations landing data may be incomplete because they do not account for the considerable amount of fish consumed by the fishers, their families and associated communities.

In commercial and artisanal fisheries, fishers typically are requested to report their harvest and effort or submit to observation/inspection at various stages through the market chain, usually at first point of sale.

**Table 3** Data collection methods for stock assessment in inland fisheries

| Data collection method | Output | Pros (+) and cons (−) |
|---|---|---|
| *Fisheries dependent sampling* | | |
| Logbooks | Catch and effort data for individual fishermen/boats or households | (+) detailed trip information<br>(−) relies on fisher's willingness to complete detailed records |
| Surveys at landing sites | Catch, effort and biological data for sample of fishermen/boats | (+) direct sampling by observers gives high quality data<br>(−) Expensive and limited in coverage |
| Household surveys | Household level catch and effort data | (+) Captures subsistence fishing that bypasses landing sites and markets, including fishing by women and children |
| Market surveys | Total landings | |
| Voluntary reporting (traditional and mobile technologies) | Catches, effort, CPUE, species and size information | (+) Potentially easy to collect in large quantities<br>(−) Relies on individual motivation, biased towards avid anglers and high catches, low uptake and continuity |
| Automatic recording by cameras etc. | Fishing effort, catch, bycatch | (+) Can partially replace direct observation by people, cost effective<br>(−) Still requires visual analysis of recordings |
| *Fisheries independent sampling* | | |
| Standard sampling | Relative abundance, population structure | (+) Rapid sampling with gear of known selectivity and to a scientific design<br>(−) Provides only relative abundance estimates |
| Depletion sampling | Abundance (absolute), population structure | (+) Provides absolute abundance estimates immediately<br>(−) Requires intensive sampling |
| Mark-recapture studies | Abundance (absolute), growth, mortality components | (+) Provides estimates of absolute abundance, growth and mortality components, movement<br>(−) Requires sampling over extended period |
| Hydro-acoustics | Biomass, size structure, distribution | |
| Remote sensing | Habitat characteristics, fishing effort | |
| eDNA | Indirect estimate of biomass index | (+) Rapid in the field, suitable for habitats that are difficult to sample<br>(−) Provides only rough estimate of biomass |

Note that the methods are not mutually exclusive: e.g., biological sampling and mark-recapture studies can be integrated with various fisheries-dependent and fisheries-independent studies

These catches are then up-scaled to account for the whole fishing effort through frame surveys (Bazigos 1974; Caddy and Bazigos 1985). Catch statistical surveys typically involve observers visiting a high proportion of the fishing villages or landing sites on a regular or *ad hoc* basis and collecting as much information on the number of fishing boats, fishing gears and fishermen as possible. These data are usually collected by direct enumeration or random subsampling. Although this procedure gives a summary picture of the fishery, the data collected are often based on a single sample and may not be truly representative of the pattern over space and time. In addition, the manpower and financial resources required to implement such frame surveys on large fisheries are generally excessive and often lead to inadequate coverage of the fishery, particularly when it is diverse with regard to the fishing methods used and species exploited. These limitations preclude the reliability and applicability of simple frame surveys for assessing the overall status of (large) fisheries and warrant the use of subsampling or stratified sampling procedures (Bazigos 1974). Subsampling procedures involve measuring catch and effort data for a representative sample or samples and determining the total population statistics by proportion. The procedure is dependent on the subsample being representative of the whole fishery; a situation which is difficult to assess but rarely achieved because of variability in many parameters associated with the fishery. Many of the problems are overcome by stratified sampling procedures, engaging with fishers to collect appropriate information using logbooks, or through comanagement arrangements with the fishing communities.

*Household and angler surveys*

Landings data can be difficult to generate in inland fisheries, as many inland fisheries are small, highly dispersed, and often subsistence or recreational in nature (meaning that catches do not pass through lending centers or markets). In many inland fisheries, there are no formal licensing or reporting requirements and even where these exist, such requirements as well as fishing regulations may be very difficult to enforce. Therefore, catch statistics for many inland fisheries must be collected using approaches such as household surveys of subsistence fishers or telephone surveys of anglers. Even appropriate survey designs, however, remain vulnerable to the effects of underreporting of legal and illegal catches. (e.g., Kang et al. 2009; FAO and World Fish Centre 2008).

For dispersed, subsistence or recreational fisheries, targeted surveys are required to obtain the catch information. Survey methods include: (1) on-site surveys in which fishers are contacted during or shortly after they complete a fishing trip; and (2) off-site surveys in which fishers are contacted by mail, phone or other methods and asked to recall their fishing experience during a prior period (Pollock et al. 1994). On-site surveys are expected to deliver more precise and less-biased estimates of effort and harvest because they rely less on angler recall (Malvestuto 1983), but this method is not practical when monitoring large numbers of inland lakes and rivers. Off-site survey methods are simpler when the target population of fishers is known. This is often the case for recreational fisheries in countries where anglers must obtain a license. Surveying the population of license holders can provide estimates of fishing harvest and effort for the entire population or for specified sectors of the population. Spatially explicit estimates are available if the survey obtains information about where anglers fished over a relatively coarse temporal and large spatial scale.

In Canada, a mail survey method has been used since 1975 to monitor recreational fisheries. The survey is conducted at 5-year intervals and provides useful statistics for measuring the size of the fisheries in each province and tracking changes through time. These statistics include fishing effort, as well as the catch and harvest by species. In the province of Ontario, the mail survey data have been used since 2005 to estimate fishery statistics in each of 20 fisheries management zones [Hogg et al. 2010a; Ontario Ministry of Natural Resources (OMNR) 2005]. In addition, potential bias has been evaluated by comparing mail survey results to on-site survey estimates. In Ontario the mail survey estimates tend to overestimate fishing effort and harvest by about twofold (Hogg et al. 2010b).

These surveys can be used to quantify harvest from fisheries where a substantial share of the catch is neither marketed nor landed at defined landing sites, for example subsistence fisheries in small or larger water bodies that are accessed in a diffuse manner (SEAFDEC 2005). Specific goals of household surveys are varied, but may include estimating food consumption, household income, food production decisions, contribution of fisheries products to livelihoods, time and capital investment in fishing activities (Beaman and Dillon 2012). Household surveys can also be used to collect detailed data suitable for use in fisheries assessments, for example on catches from different water bodies or habitats, species composition, seasonal change in catch composition and use of fishing gears (SEAFDEC 2005; Nguyen Khoa et al. 2005; Almeida et al. 2009). Household surveys have been successfully implemented in the Mekong region (IFReDI 2013) and elsewhere (Almeida et al. 2009) to study inland fisheries. Inland fisheries information generated through household surveys is mainly used for generating awareness on the size and importance of inland fisheries in terms of abundance, diversity, contribution to income, livelihoods, and food and nutrition security (IFReDI 2013). Household surveys are occasionally complemented by collecting data through focus group discussions (Hortle and Suntornratana 2008).

Household surveys may also be conducted communally as part of community-based fisheries management (CBFM) or co-management efforts. For example, CBFM can provide support for fisheries assessment through household surveys and community-based fisheries approaches collecting quantitative data (yield, fish catch data), quantitative indices

(CPUE, production methods, aquaculture) and relate these to the habitats involved (floodplain, mainstream, rice fields, coastal). As mentioned in *Simple indicators*, estimates of total harvest and effort may be difficult to achieve, but simple proxy indicators (*e.g.,* CPUE and mean size of fish) are very useful for monitoring status of a fishery. A good example of such a mixed approach is the cooperative assessment developed for arapaima *Arapaima gigas* in the Amazon (Castello et al. 2009) or assessment of catches in the lower Mekong Basin (Hortle and Suntornratana 2008). Interaction with fishers can elicit local knowledge of the time and place fish are caught and information on catch rate, which may reflect changes in abundance. Such knowledge should be sought and used, and may be combined with formal scientific approaches (Hind 2014).

### *Voluntary reporting by anglers*

Voluntary reporting of catches and fishing activities is sometimes used to complement formal surveys, particularly in recreational fisheries. Examples include angler diaries (Pollock et al. 1994; Cooke et al. 2000) and various web or mobile device-based programs. Such approaches are often popular and can generate useful additional data, but generally cannot replace formal surveys. The reason for this is that sampling is not random (who submits information and for what events is strongly influenced by personal characteristics including avidity and education level), unsuccessful trips are rarely reported, and few participants remain in such programs for extended periods of time (Cooke et al. 2000). These factors limit data quality and generally lead to biases that are difficult to quantify (Pollock et al. 1994; Bray and Schramm 2001). However, the quality of such reporting procedures can be enhanced by targeting fishing competitions and angling associations that want to keep records of catches (e.g., Cowx and Broughton 1986).

Cell phones, smartphones, tablet computers and other hand-held devices are emerging as important fisheries data collection tools (Gutowsky et al. 2013). These technologies are attractive to researchers and agencies because they (1) often combine global positioning, wireless capabilities, accelerometers, gyroscopes, and high resolution cameras in a single unit (that can accommodate additional attachments); (2) can quickly capture, store and upload a great deal of information; and (3) reduce data recording and transcription errors. Examples of researchers and government agencies 'going digital' include the government of Ireland using a mobile application (app) to identify and georeference invasive species fished in inland waters (Inland Fisheries Ireland 2012), and the National Oceanic and Atmospheric Administration using an app to monitor a commercial groundfish fishery (E. Kupcha personal communication).

Perhaps the greatest potential for mobile technologies is in their ability to put fisheries dependent data collection in the hands of fishers from all inland fisheries sectors. This potential is starting to be realised in well-developed recreational and small-scale commercial sectors. For example, recreational anglers that use the iAngler app generated data for common snook, *Centropomus undecimalis*, stock assessments in Florida, USA (Muller and Taylor 2013), and charter boat captains that use the iSnapper app generated harvest data in support of red snapper, *Lutjanus campechanus*, management in Texas, USA (Stunz et al. 2014). Other examples include iFishWatcher for European anglers (Abou-Tair et al. 2013), a mobile system for small-vessel, commercial salmon fishers in Oregon, USA (Lavrakas et al. 2012), and a text-based data collection system for coastal recreational anglers in North Carolina, USA (Baker and Oeschger 2009). Text-based collection may be a viable option in cell-phone rich (but smartphone poor) areas such as Africa (Bratton 2013). Even angler apps that aren't developed with research or management in mind can provide informative data. For example, data from the iFish Alberta app revealed province-wide, seasonal patterns of lake popularity that were consistent with conventional data, and described lake connectivity via anthropogenic usage, a novel concept not been widely described in Alberta or elsewhere (Papenfuss et al. 2015). The technology is also used to inform fishers of the sale price of fish in different seasons and locations and thus provides key information to determine the economic value of the fisheries (e.g., in South India; Jensen 2007).

These and other examples of user-generated assessment data indicate that mobile technologies can provide relatively inexpensive, high-resolution, real-time harvest and first sale price data to inform and engage fishers (e.g., via cooperative assessment, citizen science). By contrast, conventional approaches to fisheries dependent data are costly, limited by diminishing resources, fairly restricted in time or space, can suffer from recall bias and are not appropriate for all inland fisheries or fisheries sectors. User-generated assessment data are particularly valuable to highly distributed fisheries for which traditional approaches to data collection are impractical (e.g., many water bodies). Mobile technologies are also conducive to proactive and predictive management because they allow for early detection (either through direct reporting or real-time analyses), and have the potential to re-distribute fishers through access to real-time updates relative to reference points (e.g., estimated yield relative to sustainable yield). An additional benefit is the capacity for fishers to report 'non-fisheries' observations such as pollution events, changes in water levels or invasive species.

*Automatic recording using cameras*

Automatic recording of fishing activities, catches and fish populations using cameras as an alternative to human observers has been tested in recent years. For example, boat ramp cameras have been used in British Columbia, Canada to quantify recreational fishing effort on lakes (van Poorten et al. 2015). Cameras mounted on board of fishing vessels are increasingly used to monitor catch, bycatch and interactions of fishing with protected species (Evans and Molony 2011). Stationary, towed or remotely operated underwater cameras are widely used in surveys of reef fish stocks and may well have useful applications in inland systems (Schobernd et al. 2013).

## Fisheries independent sampling

Fisheries independent sampling involves the collection of data independent of fishery. Typically, fisheries independent sampling is used to estimate fish abundance and other attributes of the targeted fish populations that may not be easily derived from fisher catch data. Often, the methods used will sample many species, not solely the species of interest for the fishery. The gear types used for fisheries independent sampling can be diverse and include those that require physical capture of the fish (e.g., netting, electric fishing) or those that rely on visual or acoustic observations (e.g., snorkel counts, hydro-acoustics, underwater video). They can include commercial gears that can be adapted to catch the full size range of fish, for example, by reducing mesh sizes, or use of gears that are banned, e.g., micro-mesh seine nets and trawling. These techniques have various strengths and weaknesses and their specific details are beyond the scope of this paper. Readers are encouraged to consult some of the many "how to" fisheries techniques-type resources on various gear types (e.g., Bonar et al. 2009; Zale et al. 2012). It is important to understand the biases associated with different gear types and carefully consider how and when they are deployed (see Pope and Willis 1996). Knowledge of the basic spatial ecology of fish (e.g., through telemetry studies) can be useful for developing effective assessment strategies and understanding their biases.

*Standard sampling*

Standardization of fish sampling techniques offers many advantages (Bonar and Hubert 2002; Bonar et al. 2009). Standard data collection methods give biologists a much improved ability to compare data across regions or time. This meets increasing needs for larger regional or global assessments necessary for setting broad-scale regulations, identifying effects of global climate change, and evaluating the adequacy of global food supplies. Furthermore, data collection standards help biologists communicate and share data across political boundaries. Standardization of fisheries assessment techniques helps control bias associated with different fish collection techniques. For example, collecting fish using nets of different non-standard mesh sizes obscures evaluations of whether differences in fish catches over time or among water bodies were due to actual shifts in the fish community or simply differences in the species or size groups that the nets caught. When assessment does occur it often is done haphazardly and without standardized gear types or methods such that among-system, cross-jurisdictional or between-year comparisons are nearly impossible—something that must change and has now occurred at least on continental scales in North American (Bonar et al. 2009) and European inland waters [European Committee for Standardization (CEN) 2003, 2005; and others].

When fisheries biologists have used standard assessment techniques benefits have been striking. For example, Swingle (1950, 1956) developed early standard techniques to study fish populations in Southeastern US ponds. The information gained was instrumental in understanding the basic biology of fishes, and how to manage them successfully in ponds for food and sport. Standardization of fisheries assessment techniques has progressively encompassed larger regions. For example, the American Fisheries Society recently recommended standard techniques for sampling North American freshwater fish populations, a process involving 284 biologists from over 100 North American organizations (Bonar et al. 2009). The European Community has a continuing program to develop fish sampling standards involving many European countries [e.g., European Committee for Standardization (CEN) 2003, 2005; and others]. At international meetings (e.g., World Fisheries Congress, American Fisheries Society), standardization on even larger, global scales has been discussed—an increasingly important issue with advances in worldwide communication and global threats to inland fisheries.

There are many, primarily social, barriers to standardization, including differences in governance strategies (protocols, policies, laws) surrounding gear use. However, promoting institutionalised change in technique(s) currently being used to adopt a universal standard is challenging, often because of potential interference with long-term data sets, political rivalry among agencies and countries, and tradition (Bonar and Hubert 2002). One gear type can to be compared to another using developed gear calibration and comparison techniques (Peterson and Paukert 2009) or ground-truthing for comparison. Further compliance with standardized procedures may be encouraged by continuing to state the advantages of using standardized methods; including many varied parties in standards development; and, depending on the situation, either not requiring/requiring methods to be adopted.

Standardized fish sampling is integral to the Sustainable Rivers Audit that monitors environmental condition of rivers across the Murray-Darling Basin (1 million $km^2$) in south-eastern Australia (Davies et al. 2010, 2012). It uses environmental metrics derived from field samples and/or modelling and combined as indicators of condition in five themes (hydrology, fish, macroinvertebrates, vegetation, and physical form). Fish sampling provides data on the identity, origin and condition of individual fish, their abundance and the composition of communities. Fish sampling is undertaken under low-flow conditions in spring–summer–autumn, using standardised methods (electric fishing, bait-trapping), which allows for assessment of most species across the fish community. A minimum of 18 sites per river valley are sampled on a rotational basis every 3 years. Main habitat types in the channel at each site sampled in proportion to their relative extent and a range of covariates are also measured. Such repeated sampling over time allows for assessment of the condition of populations and trends in relation to environmental conditions or restoration measures (see Koehn and Lintermans 2012) to be made. Reference condition, an estimate of condition had there been no significant human intervention in the landscape, provides a benchmark for comparisons and target setting.

### *Depletion sampling*

Depletion and mark-recapture methods are commonly used in inland fisheries to obtain estimates of absolute population abundance and possibly other parameters (*e.g.,* catchability of gear). Depletion sampling is used to estimate abundance by removing a known amount of fish and monitoring the resulting change in relative abundance indicators such as catch-per-unit effort and thereby establishing what proportion of stock abundance the removed fish represent. Depletion methods are well suited for use in smaller freshwater systems where it is feasible to remove a substantial proportion of the population and can provide precise estimates of absolute abundance as long as key assumptions are met (e.g., Seber 1982; Rosenberger and Dunham 2005).

### *Mark-recapture methods to estimate abundance and vital rates*

Mark-recapture methods are powerful means for obtaining absolute abundance, growth, mortality and movement information at high precision (Francis 1988; Pine et al. 2003; Coggins et al. 2006). The methods can be applied regardless of whether fish can be aged and are as suitable for tropical as for temperate systems. Mark-recapture methods are well suited for abundance estimation in smaller freshwater systems where it is feasible to mark a substantial proportion of the population. The methods are also well suited for cooperative and community based research programs. Amilhat and Lorenzen (2005) for example used a cooperative tagging program with rural communities in Northeast Thailand to quantify growth, mortality, habitat use and movement of snakehead (*Channa striata*) in a rainfed rice farming landscape.

### *Biological sampling*

Biological sampling to obtain basic life history data on growth, maturation, fecundity, mortality rates, etc. is important to all assessment methods that use size- or age-based information (Ricker 1975). Biological sampling typically includes aging of fish from hard parts (otoliths, scales, or vertebrae), staging for maturity, and measurement of fecundity. Age information is crucial for estimating growth (from size-at-age data) and mortality rates. Some methods are available for estimating growth and mortality rates solely from length frequency data when fish cannot be aged (Pauly 1987; Gulland and Rosenberg 1992). The resulting input parameters are crucial for many of the above assessment techniques.

### *Hydro-acoustics*

Hydro-acoustic technology provides numerous benefits as an un-invasive sampling method, including being suitable for studies at the population scale (MacLennan 1990) although it does not work well in all situations (e.g., where there is high levels of entrained air, or high levels of in-water structure such as vegetation). These methods use the propagation of sound waves to detect the presence of individual or schools of fish, and other aquatic organisms such as plankton. Bezerra-Neto et al. (2012), for example, used hydro-acoustic technology to evaluate diel movements in fish communities in response to daily migration patterns of *Chaoborus* spp. in Brazilian Neotropical lakes. Hydro-acoustic technology can largely be categorised into approaches using passive sound detection or active emission and reflection of sound beams. Active approaches can further be broken down into those using vertical or horizontal beams, and either method may be applied using a fixed (i.e., on shore) or mobile (i.e., from a boat) station (Lucas and Baras 2000). Early applications relied on single beams, which yielded information on the location of targets, but not on directionality or movement. Subsequent generations modified transducers to apply dual beams (determine target distance, direction, and orientation) such as used in dual-frequency identification sonar (DIDSON) systems (Holmes et al. 2006) and split beam transducers, which use pre-programmed algorithms to measure target distance, direction, orientation and can distinguish between individual organisms.

Having been first employed in abundance assessments in the 1960s; technological advancements have greatly improved the suitability of hydro-acoustic techniques for freshwater systems (Maxwell 2007). Fixed location horizontal beaming has been applied in numerous studies for counting migrating salmonids (e.g., see Dunbar 2001; Pfisterer 2002) and other groups (e.g., cyprinids, Rakowitz et al. 2008), but this technique is limited to systems dominated by a single or few migrating species travelling uni-directionally (Winfield et al. 2007). Vertical beaming, while inappropriate for shallow freshwater systems, is increasingly being used in deeper lake environments to evaluate fish abundance, fish distribution and behaviour, either alone or in combination with horizontal beams (e.g., in Loch Ness; George and Winfield 2000 or Lake Victoria; Getabu et al. 2003; Tumwebaze et al. 2007). Incorporating both beam types in shallow lake or riverine environments allows for detections at shallower depths (see Djemali et al. 2009, where differences in population size and density in fish communities of Tunisian lakes were differentiated by photoperiod). Despite these adjustments, the applicability of the hydro-acoustic techniques remains constrained by environmental and morphological considerations. Count studies (e.g., single beam or horizontal beam) should be restricted to non-turbulent areas of river and lakes, as noisy environments dampen signal strength, a key feature of the technology (Lucas and Baras 2000). Further, both lake and river substrates should be largely free of large boulders or other features that could impede the flow of the acoustic signal (Maxwell 2007). As such, hydro-acoustic systems need to be validated at each site prior to implementation.

### *Remote sensing and GIS*

Scientists and managers are increasingly exploiting remotely sensed information to fill knowledge gaps for fish production or habitat, often through development of predictive models. For this paper, remote sensing is defined as data derived from satellites, geographic information systems (GIS) or Google Earth; the motivation is to estimate key fisheries parameters offsite, given that millions of freshwater system occur on Earth and many occur in remote locations or in countries where on-site biological assessment is not feasible. At the same time, it is acknowledged that for more local applications, remote sensing could be more inclusively defined to include new technologies deployed underwater (e.g., arrays of receivers to record fish migration, cabled sensors to provide near continuous abiotic data, hydro-acoustic receivers that estimate fish densities) or airplane-derived LiDAR to generate high resolution data on bathymetry or substrate, or drones or videography to provide localised assessment of habitat type.

Satellites are a key remote sensing tool for estimating potential fish productivity. Satellites have been used to estimate chlorophyll a (a surrogate for phytoplankton biomass) in oceans for decades, but over recent years algorithms have been developed for estimating chlorophyll *a* in fresh water (e.g., Gons et al. 2002; Lesht et al. 2013), which can sometimes be problematic owing to high levels of suspended sediments. Because positive linkages have been demonstrated between chlorophyll and fish productivity (e.g., Bachman et al. 1996; Ware and Thomson 2005), a satellite derived estimate of chlorophyll could be used to remotely predict fish production. Despite this promising application for freshwater systems, there are important caveats. First, freshwater systems with high levels of suspended sediments or productivity (i.e., $>50\ \mu g\ L^{-1}$) still require calibration with in situ estimation and validation. Second, many freshwater systems are either too small (i.e., $<1\ km^2$) or have too irregular shape (i.e., dendritic and narrow, even though total surface area may exceed $1\ km^2$) to be detected by satellites. Nonetheless, the use of satellites to estimate chlorophyll *a* and ultimately fish production is a promising tool to improve knowledge in regions of the world where biological assessment on the ground is unlikely or impractical.

A related tool for biological assessment of inland fisheries is GIS. Fisher (2013) provided an excellent summary of the primary applications of GIS for inland fisheries, including relating the distribution, abundance or movement of fishes to their habitat or watershed. The number of available GIS layers increases each year, and includes several variables that could influence fisheries production, including land cover, human population density and climate variables. Although GIS layers cannot directly assess fish productivity, models can be developed that estimate fish productivity based on habitats revealed in GIS coverages. For example, land cover within the watershed can predict the productivity of fish in reservoirs (De Silva et al. 2001; Vanni et al. 2005) or rivers (Creque et al. 2005).

Google Earth is a third potential tool to inform effort in inland fisheries when no other monitoring exists. For example, Al-Abdulrazzak and Pauly (2014) used Google Earth images to count the number of fishing weirs in the Persian Gulf (Arabian Gulf) and then made assumptions about the mean harvest per weir to argue that fisheries yield reported to FAO was biased down up to a factor of six fold. In another study, Trujillo et al. (2012) used Google Earth to count the number of aquaculture pens in the Mediterranean Sea and determined that the production estimate reported to FAO was entirely plausible. Hence, Google Earth offers a new tool to estimate fishing effort under the right circumstances, such as when large fishing gears are used, although caution must be taken over the age of the Google Earth images, especially in more remote areas.

### *eDNA*

Although physically capturing and enumerating fish is a fundamental component of biological assessment it can be laborious and inefficient for some species (e.g., those that are cryptic or rare). Over the past decade there has been a number of innovations such that it is possible to use an organism's DNA in the environment (eDNA) to determine presence/absence of a given species (Lodge et al. 2012). Presence-absence is particularly relevant in the context of detecting the distribution of endangered species (Jerde et al. 2011), assessing biodiversity (Lodge et al. 2012; Thomsen et al. 2012) or detecting invaders (Jerde et al. 2011). Since the first field applications for evaluating fish composition (e.g., Jerde et al. 2011; Minamoto et al. 2011), there have been many advances in quantitative real-time PCR and next-generation sequencing such that it may be possible to move beyond simply documenting presence to actually being able to yield a relative index of abundance (Lodge et al. 2012). This frontier is still in its infancy but the approach shows promise. For example, Takahara et al. (2012) revealed that the concentration of eDNA was positively correlated with common carp biomass in controlled laboratory environments and experimental ponds. The authors then extended their work to the field to estimate the biomass and distribution of carp in a natural freshwater lagoon. Similarly Hamilton et al. (2014) were able to determine the effects of oestrogenic effluents on genetic structure and effective population sizes of roach populations in rivers in southern England using a suite of DNA microsatellites. In general, once fully quantitative eDNA methods are developed they should offer a non-invasive, simple, and rapid method for estimating biomass (Takahara et al. 2012), which would be particularly relevant for water bodies that are difficult to sample or for which the capacity for sampling is limited. One could envision the elegance and simplicity of simply having water samples collected from around the globe and sent to central processing facilities where "detections" would be screened against a genetic library and then evaluated quantitatively to provide information on fish community structure and species-specific biomass.

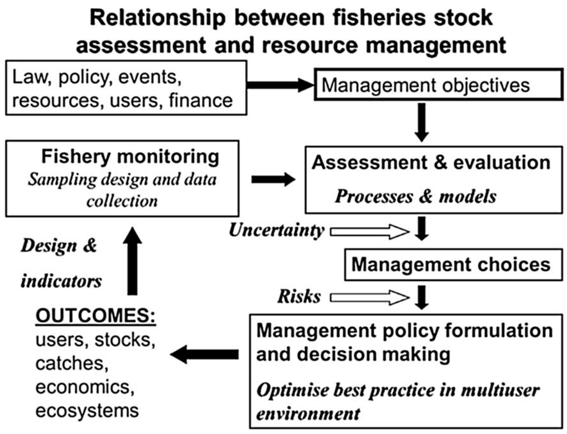



**Fig. 4** Influences and considerations impacting the resource management cycle of inland fisheries

## Stock assessment in the management of inland fisheries

### The management framework

Influences and considerations impacting the resource management cycle of inland fisheries are outlined in Fig. 4. Resource management objectives are influenced by externalities such as legal agreements, political and policy perspectives, diverse stakeholder and user groups, and financial limitations. Management decisions will attempt to maximize positive outcomes for multiple users based on evidence supplied by stock assesssments. Outcomes will then be measured using available indicators and monitoring processes.

Although the generic management framework in Fig. 4 applies to both marine and inland fisheries, the specific realizations of this framework are often distinctly different. Inland fisheries are characterized by greater diversity and complexity of objectives and often, management processses that do not 'cycle' frequently (e.g. management measures may not be re-assessed for years or even decades) and may apply to multiple separate but similar fisheries. Defined control rules for management responses in the light of assessment results are rarely used in inland fisheries, with the possible exception of stocking or habitat management rules (which are often more dynamic than fishing regulations).

### Managing for diverse objectives

Inland fisheries are often managed for diverse purposes including for example:

- Harvesting of fish for food, ornament, capture-based aquaculture and aquaculture brood stock
- Maintenance of employment and livelihoods;
- Maintenance of opportunities for recreational fishing;
- Control of invasive fishes
- Control of unwanted organisms (vegetation, disease vectors, plant pests);
- Conservation of species and ecosystems
- Aesthetic and cultural values.

Maximizing the harvest of fish for food or other purposes may call for fishing at close to $E_{MSY}$ (Fig. 1). Employment in and access to the fishery tends to be maximized in region C while still providing a high level of yield; while economic rent from the fishery (the difference between the revenue generated from fish catches and the costs of fishing) is maximized when fishing below $E_{MSY}$ in region B (Fig. 1). Inland fisheries support different types of livelihoods including commercially oriented fishers, subsistence-oriented fishers, and those fishing as a means of last resort for survival (Smith et al. 2005). While commercially oriented fishers are best served by a fishery operating below EMSY, fishing for subsistence and as a last resort may call for fishing in any region of the yield-effort curve depending on the level of subsistence needs relative to fisheries productivity. In particular, where fishing is an activity of last resort among people who have no other means of survival, even fishing at very high levels of effort to obtain small catches may provide social benefits (this extreme situation is, however, rare). Recreational anglers may be motivated by, and obtain utility from quite different attributes of the fishing experience including harvesting fish, catching fish (independent of whether the fish are harvested or released alive), catching trophy-sized fish, as well as various aspects that are not catch-related such as experiencing natural surroundings (Cooke et al. 2016a). While recreational harvest would be maximized by fishing at close to $E_{MSY}$, catch rates overall and catch rates of trophy-sized fish would be maximized at much lower effort. (Note that fishing effort here is assumed to be proportional to the rate of mortality exerted by fishing, which is correct for commercial, subsistence and harvest-oriented recreational fisheries. In catch-and-release oriented recreational fisheries, high levels of physical fishing effort can be associated with low levels of fishing mortality if fish survive well upon release). Control of invasive fishes by fishing requires very high levels of fishing effort to depress their population biomass and will yield low catches (Fig. 1). It is clear from this consideration that such activities will not in themselves be economically viable as fisheries (yielding small catches from very high effort and associated costs) but must be supported through payments for the service provided. Where fish provide important ecosystem services such vegetation or pest control, their biomass must be kept high by restricting fishing to a low level. Similarly, species and ecosystem conservation objectives generally call for maintenance of high biomass, i.e. fishing at low effort. This brief survey shows that (1) management of fisheries for different objectives implies different targets for stock management and (2) regardless of specific objectives and targets, stock assessments provide important information to guide management.

## Stakeholder involvement in data collection, assessment and management

There is a need for involvement and ownership by stakeholders (including the community) to ensure that the data collected are taken up in fisheries management. Management action is informed by science, but it also informs science; the involvement of communities in goal setting and decision-making helps to make this happen (Berkes 2003). More integration not only helps to avoid conflicts, but also allows more cost-effective assessment because the data needs overlap.

## Ecosystem considerations

Ecosystem management has been defined as “the application of ecological, economic, and social information, options and constraints to achieve desired social benefits within a defined geographic area and over a specified period” (Lackey 1999). It implies that the management of different resource uses should not be parallel, separate processes, but rather interconnected processes that have potentially conflicting objectives, overlapping data needs, and a common decision framework. It is “a management philosophy that focuses on desired states rather than system outputs” (Cortner et al. 1994). This focus on desired states is a foundation for comparing impacts, and thus net benefits, of different uses of aquatic resources. These three key drivers of fisheries (i.e., ecosystem, social, economic) can all provide important data to the fisheries management process (Andrew et al. 2007.)

Providing detailed and credible information on fisheries resources to evaluate trade-offs among human activities and development pressures (*e.g.,* agriculture, hydropower; Beard et al. 2011) is a key role for stock assessment in inland fisheries. In evaluating such tradeoffs, the same types of questions apply as for the assessment of exploitation (Fig. 5). For example, in the case of hydropower development, one would ask: How much power could be generated? How much power is being generated? How is power generation impacting the ecosystem? Much of the data needed to address questions about potential use and its impact are the same for both services, so there are obvious assessment benefits by taking an integrated approach. Lorenzen et al. (2007) provided a guidance manual for assessing the impacts of water resources development of irrigation on tropical small-scale

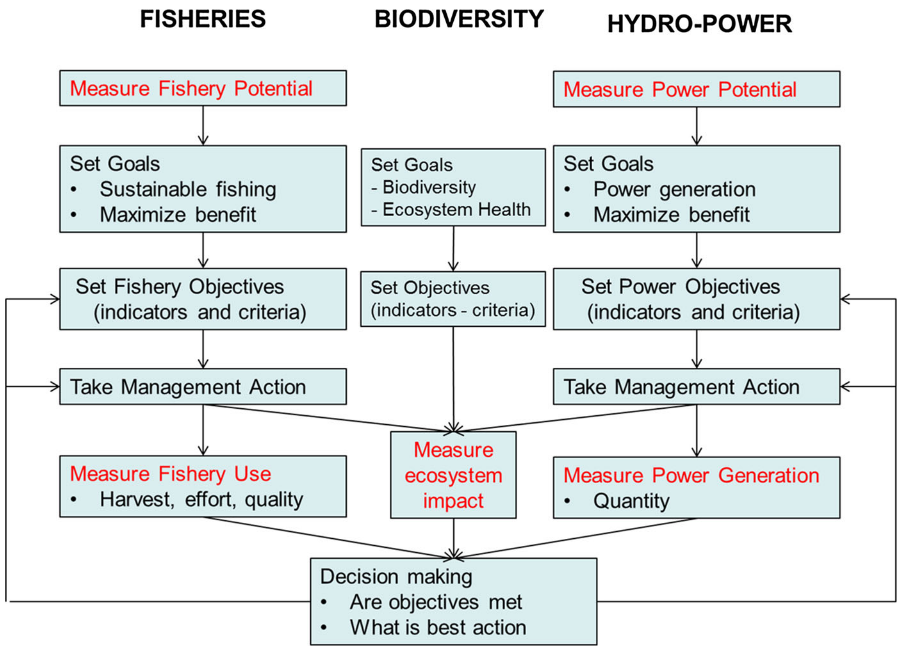



**Fig. 5** Management cycle for fisheries linked to management of other aquatic ecosystem services. The left column demonstrates traditional Fisheries Management (FM): management decisions are based purely on fisheries data (i.e., linkages to the Biodiversity and Hydro-Power services would not exist). The left and middle columns demonstrate Ecosystem-based Fisheries Management (EFM): fisheries management is influenced by other goals related to biodiversity and ecosystem health and assessment requires monitoring of the ecosystem (not just the fishery). The addition of the right column demonstrates how management of other ecosystem services can be integrated with EFM. The integrated decision-making process (*bottom box*) offers a means of evaluating trade-offs among different used of aquatic resources. Hydropower is used as an example of another service. A complete model would include all competing services (e.g., agriculture, forestry, and mining)

fisheries which could serve as a model for evaluating other stressors.

A given fishery is dependent on the ecosystem within which it resides and there are many anthropogenic impacts besides fishing that must be considered in management (Andrew et al. 2007). For example, there are a range of factors external to the fishery that can increase fish mortalities or reduce fish production or alter ecosystem pathways (*e.g.,* species at lower trophic levels; Matsuishi et al. 2006). Hence, both fisheries management and assessment are moving more to an ecosystem-based approach (Allan et al. 2005). Relatedly, one must also consider assessment of other performance indicators such as social values and perceptions and economic values. That is, biological assessment is only one form of assessment information needed by fisheries managers to make effective decisions that benefit the resource and stakeholders. Although the details of such human dimensions surveys are beyond the scope of this paper (*e.g.,* Parkkila et al. 2010), they are certainly important.

## Selecting assessment approaches and management strategies

Selecting assessment approaches and management strategies requires consideration of fisheries characteristics, management objectives, and the data that are available or can be obtained. Assessment informs management while management needs inform what should be assessed. All too often management occurs in the absence of assessment or assessment occurs but is not directly linked to the fisheries management cycle or integrated into adaptive management or an ecosystem approach framework.

### Considering fisheries characteristics

As outlined above, inland fisheries provide diverse context(s) for stock assessment that often differ markedly from those of large-scale marine fisheries in terms of the resources, fishing techniques, enhancement measures, habitat and environment, stakeholders, markets and governance arrangements. The specific characteristics of the inland fisheries to be assessed have important implications for the drivers of fisheries outcomes to be considered in assessment models and data collection (e.g. hydrological alterations or fisheries enhancement measures in addition to fishing), appropriate indicators of fisheries performance (e.g. total catch in commercial fisheries vs. availability of trophy-size fish to the creel or some recreational fisheries), the methods of data collection (e.g. effective use of tagging in small confined systems vs. reliance on long-term catch data series in large open systems), and the type, temporal and spatial scale of management advice to be provided (e.g. statewide size or bag limits that may apply to multiple recreational fisheries over multiple years or decades vs. lake and year specific catch limits). It is therefore crucial to consider broad fisheries characteristics when designing appropriate assessment and management systems.

An important consideration for the assessment of inland fisheries is the spatial scale of the individual fishery and any set of fisheries that may be assessed jointly in order to transfer information or use comparative modeling approaches.

In most cases, there will be multiple viable options for assessment and indeed, well developed fisheries management systems often rely on the use of multiple assessment approaches that serve different needs, allow cross-checking and combine empirical with process-based models and experimental management in order to identify causality as well as quantitatively precise management guidance (see e.g. Havens 1999; Havens and Aumen 2000; Hilborn 2016). When identifying assessment approaches, it is therefore best to screen multiple options for assessment methods and data collection in terms of their ability to answer management questions, data and skills requirements, and costs of implementation. This is likely to lead to a consolidated shortlist of options. A structured process and decision support tool for selecting assessment options for data-poor fisheries has been proposed by Dowling et al. (in press). The process and tool are applicable to inland as well as marine fisheries, but do not account for the specific issues and opportunities outlined in this paper that are germane to inland fisheries.

It is important to recognize here that different approaches are suitable for different situations and that, particularly where data are limited, complex approaches that account for more biological detail do not necessarily lead to better management advice (see *e.g.,* Ludwig and Walters 1985).

## Identify immediate assessment needs and approaches

In many cases, there may be pressing assessment needs that must be addressed more or less immediately. For example, a rapid increase in fishing pressure may prompt a need to assess and mange exploitation to avoid fisheries collapse or a proposed hydroelectric dam may necessitate assessment of its likely impacts on fisheries yields. In such cases, it is often necessary to develop assessment protocols that answer specific questions rapidly and without reliance on long-term data which may not be available. In our examples, this could be a size-structure based index of exploitation pressure or a comparative observational study of fisheries in rivers or river segments impacted by dams and non-impacted controls. Such approaches have limitations as outlined above, but they may provide crucial factual information for management where otherwise guesswork or unfounded assertions might rule the day.

## Identify longer-term monitoring strategies

At the same time as meeting immediate assessment needs, it is important to consider implementing approaches that will enhance data availability and assessment capacity in the longer term. This is important not only because of the limitations of most rapid and single-issue -focused assessment approaches but because implementation of multiple such approaches for different purposes will eventually be less effective and efficient than a more comprehensive approach to data collection and assessment.

## Evaluate management and monitoring strategies

The ultimate measure of performance of the management system (including objective setting, data collection, assessment, and management decision making) is its capacity to achieve management objectives or at least, acceptable outcomes. Management systems and procedures for large-scale marine fisheries are increasingly being evaluated using management strategy evaluation (MSE): simulations to evaluate the relative effectiveness for achieving management objectives of different combinations of data collection schemes, methods of analysis and subsequent processes leading to management actions (Punt et al. 2016). MSEs can be used to evaluate how well a particular strategy performs or to identify the best option from multiple competing strategies. While the use of MSEs is well established in marine fisheries, there has been no attempt yet to evaluate strategies for replicated systems as are commonly found in freshwaters.

## The way forward: research and development needs

### Promote wider adoption of appropriate assessment methods

We have outlined a suite of diverse assessment methods and approaches. Greater efforts should be made to adopt quantitative assessment methods in inland fisheries. Here we highlighted a number of examples where, often on a lake or regional (state/provincial) scale, assessment has helped to inform fisheries management. Sharing such examples can occur in the literature or electronic and social media, but may be more efficiently done through regional multi-jurisdictional working groups with external experts (as needed). For example, there may be opportunities for inland fisheries assessment practitioners to learn from those active in the marine realm (Cooke et al. 2014), while recognizing that different contexts imply different issues and opportunities. When water bodies cross jurisdictions (*e.g.,* Mekong River, Danube River), there have been some successes with the application of watershed-scale coordinated assessment and management through multi-national organizations (*e.g.,* Mekong River Commission; Danube River Commission). The UN FAO will undoubtedly play a role in making this happen yet professional societies (e.g., American Fisheries Society, Fisheries Society of the British Isles) should also take a more active role in international collaboration and capacity building. Adequate resourcing of quantitative assessments and their use in management should be a priority for management agencies at all levels and for funding bodies.

### Continue to develop and assess new tools

A variety of novel tools are under development that may provide assessment biologists with new ways to collect cost effective data using remote sensing, mobile technologies, or the expanded use of household surveys. Understanding the spatial ecology of fish using modern tools (*e.g.,* biotelemetry, hydro-acoustics) can help to optimise biological assessment protocols and ensure that field-based fisheries surveys are not biased (Cooke et al. 2016b). Although there are a wide range of tools available for contemporary fisheries assessment, many challenges still exist. Often, financial and human resources needed for sampling are limited. As a result of technological innovation, creativity and need, there is a number of promising tools that have the potential to expand the traditional fisheries assessment toolbox.

### Increased use of comparative studies and meta-analyses

As more fisheries data and assessment results become available for an ever wider range of fisheries, the potential for learning from comparative studies and meta-analyses (as part of systematic reviews) will increase. Greater efforts should be made to collate such information in standardized form and make it widely accessible.

### Develop and report meaningful indicators of fisheries and environmental status

Providing detailed and credible information on fisheries resources to evaluate trade-offs among human activities and development pressures (*e.g.,* agriculture, hydropower; Beard et al. 2011) will become increasingly important. Therefore, understanding the contribution of inland fish and fisheries to food security, societal wellbeing and livelihoods is critical. It is acknowledged that there are inherent weaknesses in reporting such statistics (such as an inability to incorporate measures of illegal, unreported and unregulated fishing, Evans 2001), thus more advanced and novel indicators may improve fisheries reporting. An example are the multi-dimensional Fishery Performance Indicators (Anderson et al. 2015).

Evaluating assessment and management strategies

The performance of alternative inland fisheries assessment and management systems should be evaluated empirically and through simulation-based management strategy evaluation. It is only through such evaluations that more specific guidance on the appropriateness of different approaches can be developed. Great progress has been made in this regard in the marine fisheries realm, yet the topic has received little attention in inland fisheries.

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