Effective monitoring of freshwater fish

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Abstract

Freshwater ecosystems constitute only a small fraction of the planet's water resources, yet support much of its diversity, with freshwater fish accounting for more species than birds, mammals, amphibians, or reptiles. Fresh waters are, however, particularly vulnerable to anthropogenic impacts, including habitat loss, climate and land use change, nutrient enrichment, and biological invasions. This environmental degradation, combined with unprecedented rates of biodiversity change, highlights the importance of robust and replicable programmes to monitor freshwater fish assemblages. Such monitoring programmes can have diverse aims, including confirming the presence of a single species (e.g. early detection of alien species), tracking changes in the abundance of threatened species, or documenting long-term temporal changes in entire communities. Irrespective of their motivation, monitoring programmes are only fit for purpose if they have clearly articulated aims and collect data that can meet those aims. This review, therefore, highlights the importance of identifying the key aims in monitoring programmes, and outlines the different methods of sampling freshwater fish that can be used to meet these aims. We emphasise that investigators must address issues around sampling design, statistical power, species' detectability, taxonomy, and ethics in their monitoring programmes. Additionally, programmes must ensure that high-quality monitoring data are properly curated and deposited in repositories that will endure. Through fostering improved practice in freshwater fish monitoring, this review aims to help programmes improve understanding of the processes that shape the Earth's freshwater ecosystems, and help protect these systems in face of rapid environmental change.

Keywords: Biodiversity Targets; Ecological Monitoring; Environmental Assessment; Environmental Management; Rivers; Sampling Design

Table of Contents

1. Introduction

Human-driven environmental changes continue to raise substantial concerns for biodiversity conservation and have led to the development and implementation of many ecological monitoring programmes around the world (Nichols & Williams, 2006). These programmes generally aim to understand and manage the interactions of environmental change with biodiversity (Fölster et al., 2014). Given the increasing seriousness of environmental degradation, the need for effective ecological and biodiversity monitoring programmes has never been higher (Lindenmayer & Likens, 2010). Freshwater ecosystems are particularly imperilled by anthropogenic activities worldwide. Although fresh waters cover less than 1% of the earth's surface, they support high levels of biodiversity (Dudgeon et al., 2006; Strayer & Dudgeon, 2010). Extinction rates of freshwater taxa are considerably higher than terrestrial species (Sala et al., 2000), due to issues including habitat loss, climate and land use change, pollution, and biological invasions (Ormerod et al., 2010; Stendera et al., 2012). At approximately 13,000 species, freshwater fish represent 40-45% of global fish diversity (Lévêque et al., 2008), with this highly diverse group including some of the most imperilled animals on the planet (Cooke et al., 2012).

Freshwater fishes also provide ecosystem services of major economic, nutritional, scientific, historical, and cultural importance (IUCN FFSG, 2015). For example, freshwater and marine fisheries jointly constitute the largest extractive use of wildlife in the world and contribute to overall economic wellbeing by means of export commodity trade, tourism, and recreation (Santhanam, 2015). Freshwater fish provide a major source of protein for humans and support the livelihoods of many people (Holmlund & Hammer, 1999), particularly in the Global South. However, there are serious threats to this valuable resource related to over-exploitation and other anthropogenic stressors (Allan et al., 2005; de Kerckhove et al., 2015).

The wide range of responses of freshwater fishes to anthropogenic stressors make fish valuable indicators for assessing the biological and ecological integrity of fresh waters and their catchments (Fausch et al., 1984; Magurran et al., 2018; Schiemer, 2000). The breadth of fundamental information on ecology and taxonomy, combined with their higher societal importance compared to other freshwater taxa, makes freshwater fish a popular target taxon in assessments of ecological integrity (Simon & Evans, 2017). Correspondingly, freshwater fishes are commonly used for evaluating the functioning and status of freshwater ecosystems and habitat quality. These assessments, however, are only as good as the data that underpin them. For this reason, effective and meaningful monitoring of fish populations and communities in freshwater habitats is essential.

The need for effective monitoring in ecological research is well-recognized and there are many monitoring programmes that have provided important scientific advances and crucial information for environmental policy (Lovett et al., 2007). For example, freshwater fish monitoring has highlighted changes in species diversity and species status in rivers and lakes (e.g. Counihan et al., 2018; Holmgren et al., 2016; Wagner et al., 2014), played a central role in fish-based assessment systems (e.g. for the European Water Framework Directive, Pont et al., 2007), and resulted in guidelines on standardized fish sampling methods (e.g. Bonar et al., 2009).

There remains a series of issues and knowledge gaps with how these programmes are designed and implemented. In particular, freshwater fish monitoring that has been poorly planned and lacks focus results in ineffective programmes that rarely meet their aims (Lindenmayer & Likens, 2009, 2010; Marsh & Trenham, 2008; Nichols &

Williams, 2006). Moreover, there is considerable disparity across developed and developing regions in how monitoring schemes are implemented. This is an acute problem, as developing regions are often characterised by high levels of fish diversity but limited resources for research (e.g. Vörösmarty et al., 2010). Where monitoring programmes are in place, there are almost inevitably trade-offs in temporal and spatial scales of measurement (Pollock et al., 2002), but these trade-offs are often poorly quantified or justified, resulting in long-term data lacking statistical power. Finally, there are inherent issues over programmes being either question driven or mandated, with the latter often lacking rigour in design resulting in their provision of only coarse-level summaries of change (Lindenmayer & Likens, 2010).

In this review, we examine these issues and knowledge gaps, and make recommendations about how they can be addressed within monitoring programmes. Our aim is to foster improved practices by: a) summarizing key questions that monitoring can address when aims are clear, and the approach is rigorous (Section 3 and 4); b) synthesising issues related to sampling design and statistical models, and indicating how they might be overcome (Section 5); c) reviewing different monitoring and sampling approaches (Section 6); d) considering challenges related to species’ detectability, taxonomy, economical costs, and ethics (Section 7); and, e) discussing the importance of the appropriate management of monitoring data (Section 8).

2. History of fish monitoring

The long history of monitoring programmes is reflected in the scientific literature (Fig. S1.1). Early, though presumably less systematic, efforts in freshwater fish monitoring recorded temporal changes in fisheries, such as reports of Atlantic salmon *Salmo salar* declines in a central European river that date back to the 18th century (reviewed by Wolter, 2015). The 20th century marked a shift towards systematic sampling with the majority of fish monitoring programmes being established before 1979 (Mihoub et al., 2017). Despite this and in contrast to other taxonomic groups such as birds, mammals, and many plants, freshwater fish are generally under-represented in contemporary biodiversity studies and monitoring programmes (Mihoub et al., 2017; Troudet et al., 2017). This underrepresentation of fish, despite their high diversity, might be explained partly by the fact that they occur in aquatic environments. Thus, in contrast to many terrestrial biota, which can be monitored by visual observations and where community scientists (also known as citizen scientists) can be easily recruited (Thomas, 1996), fish require more specialized sampling methods. However, one feature shared with other taxa is that the spatial extent of fish monitoring is highly biased, being concentrated in the Global North (Fig. 1). Freshwater ecosystems (e.g. lacustrine and fluvial habitats) are also generally neglected in fish monitoring programmes, compared to the marine environments (Fig. 1). A further issue is that even when freshwater fish are monitored, the resulting data are often not published or electronically archived, and thus are often inaccessible to the broader scientific community (Lindenmayer & Likens, 2009; Revenga et al., 2005).

[Fig. 1]

3. Aims of effective monitoring

As it is now widely recognised, ecological communities experience continuous temporal turnover, i.e. change in species composition and abundances (e.g. Darwin, 1859; MacArthur & Wilson, 1967). Some degree of temporal turnover is necessary to maintain ecosystem functions and properties. However, the rate of temporal turnover in contemporary assemblages exceeds the baseline predicted by ecological theory (Dornelas et al., 2014). Consequently, the overall goal in effective monitoring of freshwater fish should not be limited to documenting change *per se*, but should also address the drivers of the observed change (thereby identifying potential remedies).

There are a number of definitions of monitoring in conservation, ecological, and aquatic contexts (Supporting Information Table S1.1). Here, we define **freshwater fish monitoring as repeated, field-based measurements of fish that are collected in a systematic manner, allowing the potential detection of important shifts at population or community levels.** Therefore, effective monitoring requires a clear set of specific objectives linked to the overall goal of detecting systemic shifts in fish populations or communities over time and space, and so should utilise methodologies and sampling effort that provide the data and statistical power sufficient to meet these objectives.

4. Different questions lead to different monitoring approaches

Monitoring programmes need a rigorous design and protocol for collection of data over a sufficiently long period to ensure sufficient statistical power to detect trends or changes and to enable the answering of the motivating questions (Lindenmayer & Likens, 2010; Nichols & Williams, 2006). Irrespective of the motivating question, freshwater fish monitoring should generally help to advance ecosystem understanding and provide information needed to identify potential remedies, requiring the detection of significant changes at the community level (e.g. quantifying trends in species richness, temporal α-and β-diversity, functional diversity, food web structure), and/or at the population level (e.g. quantifying trends in population size and dynamics, abundance of keystone, threatened or non-native species, genetic diversity, species ranges, fisheries stocks, size and age structure, behaviour, phenology, growth, shape, and/or condition). An exception to this might be in mandated-monitoring programmes where highly specific data (e.g. on species presence, abundance, and/or age structure) are compared against predetermined standards (Alexander, 2008; Hellawell, 1991; Hurford, 2010), such as in the Water Framework Directive of the European Union (Birk et al., 2012). In a restoration context, monitoring often aims at assessing the success of implemented measures (Kershner, 1997). Thereby, monitoring is not a stand-alone activity; it contributes to conservation oriented-science and is used to inform a structured decision-making processes in conservation management (Nichols & Williams, 2006).

It is the question(s) that determine the design of a monitoring programme. Some questions can be addressed with species-specific presence-only data, while others might require sampling of an entire community (Table 1). The latter case may utilise a range of capture methods (Zale et al., 2012) that can, in turn, help assess the spatial behaviour, trophic ecology, and genetic characteristics of individuals (Lucas & Baras, 2000;

Lundqvist et al., 2010). Alternative sampling methods include more recent approaches such as community science and the use of social media/crowd-sourced science (Section 6). The data needs associated with a suite of key monitoring questions are summarised in Table 1. We stress the importance of programmes clearly articulating their questions as this ensures that the sampling design can generate the data required to answer them. As a minimum, there should be identification of what needs to be measured (e.g. fish abundance, fish attributes), the spatial and temporal scope of the programme (e.g. duration, scale; *cf*. Dixon & Chiswell, 1996); the criteria for reliability (e.g. precision, power); and the practical constraints (e.g. human resources, costs, social conflicts).

[Table 1]

5. Sampling and network design, and statistical models

Sampling design relates to the temporal frequency of sampling within a designed network that comprises a series of spatially segregated sites. As such, decisions need to be made regarding how to allocate monitoring effort within and among years, and across sites (Larsen et al., 2001). Two major principles, the avoidance of bias in the selection procedure and achievement of high precision, should underlie the design (Crawford, 1997). A sampling design can be based on probabilistic or non-probabilistic methods. Probabilistic designs include simple random sampling, systematic sampling, and stratified random sampling, with the latter two being more appropriate for heterogeneous, hierarchically-structured aquatic environments, such as river drainages (Lowe et al., 2006; Thorp et al., 2006). However, in fish monitoring, sample sites are frequently selected non-probabilistically, often based on judgment or convenience (Pope et al., 2010; Wilde & Fisher, 1996). Irrespective of this, decisions on the design of the programme should be based on *a priori* defined statistical models that can reliably answer the questions motivating the monitoring programme, such as those related to quantifying community structure, species abundance or other population parameters (e.g. age structure). These questions require consideration during design phases as well as additional resources and time, separate from the monitoring programme itself, for completion.

Where the aims are to detect changes related to (local) management actions such as habitat restoration, or to impact assessment, before-after control-impact (BACI) designs are frequently used (Osenberg et al., 2006; Stewart-Oaten & Bence, 2001; Thiault et al., 2017). Here, *a priori* power analyses (Legg & Nagy, 2006; Marsh & Trenham, 2008; Maxwell & Jennings, 2005; Peterman, 1990) can guide the estimation of the minimum number of samples needed to detect a certain effect size (or minimum detectable difference) according to a desired level of significance (Peterman, 1990; Steidl et al., 1997).

However, as fish monitoring programmes are typically undertaken to detect temporal changes in populations over potentially larger scales (Cowx et al., 2009), statistical control and replication designs are often unfeasible (Carpenter et al., 1989; Hargrove & Pickering, 1992; Schindler, 1998; Turner et al., 2001). Advanced Bayesian (hierarchical) models (Hobbs & Hooten, 2015) offer useful alternatives, especially when working with imperfect datasets and/or uncertainty associated with sampling and observation, as it is often the case in fish monitoring. For example, Wenger et al. (2017) applied a Bayesian approach to predict the viability of multiple (potentially isolated) populations of Lahontan cutthrout trout (*Oncorhynchus clarkii henshawi*); this approach enabled predictions to be made in minimally-sampled or even un-sampled populations.

Other applications of Bayesian models to analyse monitoring data include estimations of occupancy and richness of fish while accounting for imperfect detection (Bayley & Peterson, 2001; Coggins et al., 2014), and for relating environmental drivers to stream fish population dynamics (Letcher et al., 2015; Wheeler et al., 2018).

The spatial structure of dendritic networks, and their associated connectivity and directionality, make river systems particularly challenging for monitoring. The effect of spatial variability can be reduced by stratified random sampling, i.e. the proportional sampling of strata that represent different habitat units (Downes et al., 2002) and is widely used in aquatic ecosystems (Dukerschein et al., 2011; Haxton, 2011; Wilde & Fisher, 1996). More recently, Spatial Stream Network models (SSN) have been developed to better capture the continuous nature of rivers (Fausch et al., 2002) and to account for the spatially autocorrelated relationships between locations within a stream network (Isaak et al., 2014). For example, Isaak et al. (2017) analysed a large fish density dataset using SSN models to obtain population estimates for trout species from 108 sites in a 735 km river network. The SSN methodology is accessible via the statistical tools 'STARS' (Peterson & Ver Hoef, 2014) and 'SSN' (Ver Hoef et al., 2014).

In a systematic sampling design, the first sample site is chosen randomly and all subsequent samples are regularly placed in space or time (Conroy & Carroll, 2009; Quinn & Keough, 2002). A systematic design is useful when investigating effects of environmental gradients. A recent development in this context is the Generalized Random Tessellation Stratified design (GRTS) (Stevens & Olsen, 2003, 2004), available from the statistical package 'spsurvey' (Kincaid & Olsen, 2016). GRTS allows design-based inferences to entire areas based on spatially-balanced samples, i.e. a spatial distribution of sample locations that balances the advantages of simple or stratified random samples or systematic samples (Larsen et al., 2008). GRTS has been evaluated as reliable and cost-effective, for example, for monitoring North American salmonids (Gallagher et al., 2010).

The adaptive approach (Box 1) argues that the sampling design should be re-evaluated and re-designed as necessary as data are gathered and their variability analysed. An analysis of the components of variance and their influence on trend detection capability can help in preparing design-efficient trend monitoring networks (Larsen et al., 2001). This ensures that changes in the chemical, physical, or biological conditions are accounted for in the sampling design (Buckland et al., 2012; Strobl & Robillard, 2008).

Box 1. Adaptive monitoring

There is often high uncertainty and complexity in the drivers of fish community change that can range from global environmental change (e.g. climate change; Graham & Harrod, 2009; Radinger et al., 2016) to more local issues (e.g. altered flow regimes; Harby et al., 2007). Monitoring programmes must be capable of providing data suitable for the continued management of the resources (Polasky et al., 2011). The informed decision-making process of adaptive monitoring (sensu Lindenmayer & Likens, 2009) enables monitoring programmes to evolve in response to new questions, information, situations, or conditions or the development of new protocols (Lindenmayer et al., 2011). Adaptive monitoring is considered a long-term activity closely related to scientific research and management. The ultimate aim of any adaptive monitoring programme is to demonstrate that new insights gained through its application will improve management practices (Lindenmayer et al., 2011), potentially leading to increases in the effectiveness of monitoring for conservation.

An example of adaptive monitoring is outlined by Fölster et al. (2014) for Swedish fresh waters. At the outset the early naturalists measured specific and localized natural phenomena such as the relationship between macrophytes and lake water chemistry (Lohammar, 1938). However, the scope of the freshwater monitoring programme in Sweden and the number of monitored sites increased along with the emergence of new challenges related to, for example, eutrophication in the 1960s, acid rain in the 1970s, and the EU Water Framework Directive in 2000. Today, the program consists of regular long-term monitoring of water chemistry and biodiversity (including freshwater fish) in 114 streams and 110 lakes (Fölster et al., 2014). This example not only illustrates the value of adaptive monitoring by providing long-term data to understand and overcome many of the emerging environmental problems, but also emphasizes its potential to investigate future challenges, e.g. related to climate change, testing resilience theory, or predicting regime shifts and tipping points.

6. Approaches to fish monitoring

6.1. Monitoring questions versus sampling methods

The numerous sampling methods that can be utilised for fish monitoring, including capture and non-capture techniques, have been extensively reviewed (e.g. Bonar et al., 2009; Joy et al., 2013; Zale et al., 2012). Capture methods involve the physical removal of fish from the water to enable species identification, and the collection of biometric data (e.g. length, weight) and hard structures (e.g. scales) for ageing the fish to determine population demographics and dynamics. The most common methods available for capturing freshwater fish include electrofishing, netting, and trapping (Bonar et al., 2009). Non-capture methods (e.g. hydroacoustic surveys) can provide data complementary to capture techniques. They can also be used where capture methods lack sufficient power to provide robust estimates of population abundances (Hughes, 1998; Lyons, 1998). However, a feature of some non-capture methods is their taxonomic ambiguity due to either their lack of fish capture (Boswell et al., 2007) (Section 6.4) or through erroneous identification of specimens (Section 7.2).

The application of a sampling method in monitoring might differ markedly according to the programme's aims. For example, electrofishing can be applied within point abundance sampling designs that can be effective for monitoring the diel activity of (small) fishes (reviewed by Copp, 2010) or the status of rare species (e.g. the critically endangered European eel, *Anguilla anguilla*; Laffaille et al., 2005). However, capturing fish in longer river reaches using electrofishing might be more suitable where the monitoring aim is to assess biological/ecological integrity, as biotic indices require data at multiple organization levels, from size structure to assemblage richness (e.g. Noble et al., 2007; Pont et al., 2007; Schmutz et al., 2000), often in conjunction with data on habitat quality (e.g. Van Liefferinge et al., 2010; Milner et al., 1998).

6.2. Capture techniques and application within monitoring programmes

The challenge of ensuring that capture methods are fit for purpose, such as evaluating the composition of an assemblage (details in Box 2) (e.g. Zale et al., 2012), has resulted in a series of standardised protocols being made available for sampling inland fish populations in many areas of the world, including Europe, North America, and New Zealand (Bonar et al., 2009; CEN, 2003, 2006; Joy et al., 2013; Table S4.1). Standardization not only refers to the equipment used or how it is used, but also to the timing of sampling, the habitats that are sampled, and effort applied (Bonar et al., 2011).

Standardizing the collection and reporting of fish monitoring data offers many advantages including an improved ability to compare data across regions or time, improved communication across political boundaries, and the control of bias associated with different sampling techniques (Cooke et al., 2016). Standardization in fish sampling has been considered an important step forward in managing long-term data and assessing efficacy of large spatial scale management strategies (Bonar et al., 2017). This is of particular relevance in monitoring programmes where many researchers combine datasets to jointly address questions over time and space. For a comprehensive overview on standardisation of fish sampling across sampling gears and aquatic environments, see Bonar et al. (2009).

Two fundamental concepts have emerged in relation to the application of capture techniques and protocols to fish monitoring: the importance of sampling design (discussed earlier in Section 5) and response design (Stevens & Urquhart, 2000).

Response design incorporates decisions about how to measure the fish community and population metrics with accuracy and precision (Pollock et al., 2002). For example, where assessments of age structure, growth rates, and recruitment are required, then decisions are needed on the ageing method, such as whether to rely on length-frequency analyses or collect hard structures, such as scales, from captured fishes (e.g. Hamidan & Britton, 2015). If scales are collected, then decisions are needed regarding how many individual fish need to be sampled and over what size range (Busst & Britton, 2014). In addition, where hard structures are being used for ageing, the frequency of annulus formation might need validating to maximise accuracy (Beamish & McFarlane, 1983), requiring regular sampling throughout the year or mark-recapture methods (Britton et al., 2010; Chisnall & Kalish, 1993). Scale samples for fish ageing, and tissue samples for genetic and stable isotope analyses, can be collected from fish captured by anglers to complement on-going monitoring (Gutmann Roberts et al., 2017).

Box 2: Sampling effort and biodiversity estimation

Decisions about the spatial extent and duration of sampling have important implications. If the goal is to quantify an attribute of a population of interest, then, all other things being equal, estimates of abundance will scale predictably with effort. There are a range of statistical techniques, such as removal sampling (Southwood & Henderson, 2000), that can be used to estimate population size and/or to ensure that effort is adequate for the intended purpose. It is relatively straightforward, therefore, to compute trends for single populations.

If, on the other hand, the aim is to quantify compositional turnover (temporal β diversity), or to calculate a metric of α diversity, such as assemblage richness, it is essential that any temporal or spatial comparisons take account of the inherent unevenness of ecological assemblages. Although the number of individuals (across all species) will typically increase linearly if an assemblage is sampled over a longer time period, or the area sampled is increased, the species accumulation curve will gradually flatten (Fig. 2). As a result, any metrics that either explicitly or implicitly depend on richness cannot be scaled by simple multiplication or division. Species richness is the metric most obviously influenced by this, but most biodiversity indices, including, for example, the Berger-Parker dominance metric (Magurran, 2004, 2011; Magurran & McGill, 2011) and Jaccard similarity (Baselga, 2010), are also affected.

Fortunately, there are statistical solutions to this problem. Rarefaction is the traditional way of making fair comparisons across assemblages or of community diversity over space or time (Gotelli & Colwell, 2001, 2011). In essence, the samples (or assemblages) are rarefied to the smallest common sampling effort. Rarefaction can be computed in relation to the minimum number of individuals sampled, or to the smallest number of sampling units. While most rarefaction analyses focus on species richness, in principle many different biodiversity metrics can be rarefied. In the case of temporal or spatial β diversity comparisons, the investigator should use sample-based rarefaction as this automatically retains the identity of the species involved. A recent innovation is to extrapolate to the largest sample size rather than rarefy to the smallest one (Chao et al., 2014; Hsieh et al., 2016). Rarefaction can also be used to make informed comparisons about community structure and composition using null model approaches (Cayuela et al., 2015; Cayuela & Gotelli, 2014). In summary then, any computation of trends in community α diversity or β diversity should either be based on sampling that has been rigorously standardized or data that have been statistically standardized (by rarefaction or similar) – see Fig. 2 for an example.

[Fig. 2]

6.3. Capture and release methods

It is often desirable to release captured fish, unharmed, to the site of capture, without further intervention. However, attaching tracking devices or marking fish, prior to release, can substantially increase the amount of information obtained. For example, biotelemetry using acoustic, radio, or passive integrated transponder tags (Cooke et al., 2011; Thiem et al., 2011) can reveal individual variability in movements and behaviours within and between populations (Lucas & Batley, 1996), elucidate population mixing and gene flow (Huey et al., 2011), assess the effects of connectivity and habitat fragmentation on river fishes (Capra et al., 2017; Lin et al., 2018), and help evaluate management units for fisheries or conservation (Funk et al., 2012).

Mark-recapture studies can also strongly complement fish monitoring by providing alternative estimates of population size and fish ages (Hamel et al., 2015; Sass et al., 2010). They can also reveal the extent of migrations of individual fish between habitats within specific populations (Sandlund et al., 2016).

6.4. Non-capture monitoring techniques

Non-capture monitoring methods to complement capture data include environmental DNA and hydroacoustic assessments. These methods are often applied within monitoring programmes to provide data on different components of the community or population, and are especially useful for larger water bodies where capture techniques are often difficult to apply or are inefficient.

Environmental DNA ('eDNA' hereafter) is based on the presence DNA of fishes in water samples originating from mucus and faeces, the sloughing off of cells from their gut lining, and the decomposition of dead individuals (Davison et al., 2016; Jerde et al., 2011; Turner et al., 2015). DNA is extracted from water samples, and polymerase chain reaction (PCR) used in conjunction with species-specific genetic markers to amplify DNA fragments to indicate the presence of target species (Turner et al., 2015). The method is increasingly being applied to the monitoring of freshwater species (Fig. S1.1), including those of conservation importance (Takahara et al., 2012; Thomsen et al., 2012).

There are two basic ways that eDNA can be applied in a fish monitoring programme. Water samples can be analysed to detect the presence of a specific species, or can be screened for whole communities of organisms using 'eDNA metabarcoding' (Hänfling et al., 2016; Lawson Handley, 2015). Recent refinements have improved the reliability of species' detection (Hänfling et al., 2016), but some questions remain, for example, on factors affecting the rate of DNA breakdown in the environment (Barnes et al., 2014). However, the non-detection of species-specific DNA fragments in a sample of river water does not necessarily imply the absence of the target species, nor does a positive signal necessarily imply that the species is present, as eDNA could have been transported from upstream areas (Roussel et al., 2015). Nevertheless, as refinements in the technique continue, it should increasingly provide a strong complement to capture methods, especially in regions where knowledge on the species likely to be present is available. Although issues over the reliability of eDNA to provide estimates of abundance are being addressed, they remain highly challenging (Lacoursière-Roussel et al., 2016). One important consideration will be the integration of data collected using traditional methods with inferences about fish communities obtained using eDNA (see 6.6 below).

Hydroacoustic assessments involve the application of an acoustic beam from a transducer through the water. Any fish within the beam returns a signal, with the target strength of the returning signal indicating the relative size of the fish. Whilst the method generates data on fish density, there is high taxonomic ambiguity in terms of species present, with no biometric data collected (other than conversion of target strengths to approximate fish lengths) (Boswell et al., 2007). Nevertheless, hydroacoustic assessments have been used extensively for fish monitoring, especially in lakes where sampling strategies have been developed (e.g. Guillard & Vergès, 2007), with target strengths related to species-specific attributes to increase knowledge on community composition (Frouzova et al., 2005). In lowland rivers, such as the River Thames and River Trent in England, mobile hydroacoustic techniques have been applied to monitor the spatial and temporal distributions of fish communities (Hughes, 1998; Lyons, 1998). The method has also been applied to assessing the status of endangered fishes (Zhang et al., 2009).

6.5. Anglers' data and data mining

Statistics on angler catch rates and species composition have been applied to the monitoring of fish community composition of large lowland rivers where other fish capture methods are either difficult to apply or inefficient (Jones et al., 1995). For example, in the River Trent, England, angler catch statistics monitored changes in the fish assemblage in relation to improvements in water quality (Cooper & Wheatley, 1981; Cowx & Broughton, 1986). More recently, catch statistics from individual anglers were used to assess the population status of mahseer fishes (*Tor* spp.) in the River Cauvery, India (Pinder et al., 2015a,b). An issue with angler-based data is that they tend to be biased for specific species and size ranges (Amat Trigo et al., 2017).

Data mining, where spatial and temporal data on species are gathered through information available from on-line sources, is a different non-capture technique for monitoring changes in the distribution of species. Databases including the Global Biodiversity Information Facility (GBIF; www.gbif.org/), the Global Population Dynamics Database (GPDD; www.imperial.ac.uk/cpb/gpdd2/secure/login.aspx), or VertNet.org enable users to access global distribution records of species via directed searches that provide records with location coordinates for use within GIS. The GPDD also provides data on population dynamics, rather than just distribution data. The FishBase database (Froese & Pauly, 2018) provides species-level information gathered from the literature, including occurrences and a wide range of ecological data.

An alternative method to using these online databases is monitoring the distribution of fishes via community science, particularly via social media platforms.

Indeed, the application of community science and crowd sourcing to the collection of biological data is increasingly frequent (e.g. www.inaturalist.org, Fig. S1.1), thanks to many smartphones now having GPS, high-resolution cameras, and continuous internet connection (Bik & Goldstein, 2013; Di Minin et al., 2015). For example, for monitoring distributions of non-native fish, a number of smartphone 'apps' are available, with these generally enabling the user to send a geo-referenced image of the species to a specific organisation for validation and recording. Current examples include '*That's Invasive'* (http://www.rinse-europe.eu/resources/smartphone-apps/) and '*AquaInvaders*' (http://naturelocator.org/aquainvaders.html). Both of these 'apps' also provide users with information and images on specific invaders to facilitate their identification of species. Venturelli et al. (2017) have recently reviewed the opportunities and challenges associated with angler 'apps'.

Data can also be sourced from user-generated content on various social media platforms (Di Minin et al., 2015). By data-mining these non-biological sources, such as via searches of specific social media sources (e.g. https://www.youtube.com/), recreational fisheries forums and blogs, and news-media channels, fish distribution and dispersal data can be generated. For example, this approach has been applied successfully to assessments of non-native fish invasions, such as perch *Perca fluviatilis* and channel catfish *Ictalurus punctatus* in Portugal (Banha et al., 2015, 2017). Increasingly, these searches can be automated through use of computer code. For example, geo-referenced images and video of specific species within image and video hosting websites (e.g. flickr) can be searched, with GIS interfaces enabling distribution maps to be constructed (see Fig. 3) and thus temporal and spatial distribution patterns better understood (Coding Club, 2018).

[Fig. 3]

6.6. Complementarity of capture and non-capture methods

Data acquired from capture and non-capture methods within the same monitoring programme need to be integrated effectively. For example, fish monitoring in Windermere, England, a relatively large and deep glacial lake, has recently been complemented by application of eDNA that recorded the presence of 14 of 16 fish species known to be present, when concomitant gill net surveys only captured four fish species (Hänfling et al., 2016). Windermere has also been monitored regularly for over 60 years by other methods, including fish traps, gillnets, hydroacoustics, and piscivorous fish diet composition (Langangen et al., 2011; Winfield et al., 2008, 2012). The high complementarity of these datasets has improved understanding of environmental (e.g. nutrient enrichment, warming) and other changes (e.g. invasive fishes), and illustrated their potential for monitoring other systems (e.g. Vindenes et al., 2014; Winfield et al., 2010).

7. Major challenges in fish monitoring

7.1. Detectability

Many evaluations of biodiversity, including those of freshwater fishes (Magurran, 2004; Southwood & Henderson, 2000), assume that individuals have been sampled randomly from the assemblage (Buckland et al., 2011; Pielou, 1975). This is rarely achievable in nature (Pielou, 1975). In many cases, the problem arises because it is difficult (or impossible) to know if a species that is absent from a site or sample is truly absent, or is missing through the ineffectiveness of the sampling method. Thus, it is important to thoroughly consider observation error and capture probabilities and to address issues of detectability and detection bias also in fish monitoring. Potential solutions to issues of detectability have been extensively discussed elsewhere and include modelling occupancy (Bayley & Peterson, 2001; Iknayan et al., 2014; MacKenzie et al., 2002, 2006; Royle & Link, 2006; Wenger & Freeman, 2008), estimating the probability of detection of species (and/or individuals) through mark-recapture (Borchers et al., 2002, 2015; Buckland et al., 2011) or distance sampling (Buckland et al., 2001, 2004, 2011), and/or demonstrating that the data are sufficiently robust to address the question posed without further correction (Buckland et al., 2011; Magurran et al., 2018).

7.2. Taxonomy

Taxonomic issues can often emerge in biological monitoring programmes, with the most obvious one being taxonomic uncertainty and the risk of species misidentification in the field or the laboratory. For example, Daan (2001) reported extensive species misidentifications in a marine fish database and there are many other cases in the freshwater fish literature (e.g. Hänfling et al., 2005; Serrao et al., 2014; Vidal et al., 2010). Nevertheless, a well-appreciated advantage of fish is that their taxonomy is better known and easier than in most other freshwater groups, such as invertebrates or algae, and thus fish can often be identified in the field without sacrificing individuals. However, this is less likely to be the case in species-rich regions such as the tropics, where the taxonomy is less well known, compared to regions with well-characterised fish faunas.

The extent of species misidentification in more taxonomically challenging groups, such as stream invertebrates, receives greater attention than in freshwater fish. For example, Stribling et al. (2008) compared taxonomic identification of stream macro-invertebrates across eight U.S. laboratories and found means of 21% taxonomic disagreement. These kinds of errors might also occur in fish monitoring, especially in samples with high species richness or in samples from regions where taxonomy is poorly described. These studies reinforce the importance of adequate training and experience, documentation of standard procedures, and routine quality control (Stribling et al., 2003, 2008). Species misidentification is even more important when fishers are interviewed to obtain local knowledge data. Here, thorough validation procedures are essential (Poizat & Baran, 1997; Valbo-Jørgensen & Poulsen, 2000).

A similar problem is when taxonomy changes and it is recognised that a single species in fact comprises several cryptic species. This problem is increasingly frequent given the increasing power of molecular tools (e.g. April et al., 2011; Lara et al., 2010; Young et al., 2013). For example, Young et al. (2013) found that the majority of species-level taxonomic units of the genus *Cottus* as evaluated by DNA barcoding did not assign to previously recognized species in this region. New taxonomic alignments hinder comparison with old samples if no specimens were preserved. In addition, the same species names may have had different synonyms in the past, meaning that databases need to be carefully revised for inconsistencies and errors. Erroneous sequences and misidentifications are also frequent in GenBank and similar sequence databases (Harris, 2003). It has been estimated that up to 56% of German freshwater fish species may be incorrectly identified to species level in some databases (Knebelsberger et al., 2015). Consequently, errors in genetics databases might have major adverse impacts on eDNA as a robust technique. It is likely that the frequency of such taxonomic problems in data is more prevalent in monitoring of freshwater fish than in research (Stribling et al., 2003). It is thus important to fully reference the taxonomic resources used in studies, not just as a quality check on methodology, but also to recognize the importance of taxonomy and the work of taxonomists (Santos & Branco, 2012; Vink et al., 2012; Wägele et al., 2011).

7.3. Economic costs

For a monitoring programme to be effective, successful and sustainable over the longer-term, it must not only be ecologically relevant and statistically credible, but also cost efficient, i.e. the perceived benefits of ecological monitoring (e.g. information on trends or status changes) must justify its cost (Caughlan & Oakley, 2001; Charles et al., 2016; Hinds, 1984). As financial limitations always apply, sustained monitoring requires a proper selection of relevant variables that need to be measured (Braun & Reynolds, 2012). Often the true costs of monitoring are not recognized and likely underestimated (Caughlan & Oakley, 2001), and its benefits depend on the value that society gives to the long-term sustainability of freshwater ecosystems. Hence, costs of monitoring need to be contrasted with the costs of not monitoring. These include increased uncertainty in evaluating outcomes and future projections, and the possibility that managers may not detect important shifts until it is too late to effectively address them.

Caughlan & Oakley (2001) provided a breakdown of monitoring costs, comprising of budgetary expenses related to, for example, data collection, data management, quality assessment, data analysis, reporting and scientific oversight, opportunity costs (i.e. other benefits forgone by allocating resources to monitoring), and external costs (i.e. costs not directly covered by the monitoring programme budget). The costs for data collection – which are frequently the largest – may vary depending on the methods applied. While established methods in fish monitoring, such as field-based capture methods (e.g. electrofishing, netting, trapping), are commonly labour intensive in the field and thus costly, the financial costs of emerging methods, such as use of eDNA, the automatized collection of data (e.g. hydroacoustic assessments), and the use of community science and data mining, are often related to post-processing, managing and analysing big data (Section 6.4). A detailed review of the costs associated with ecological monitoring can be found elsewhere (e.g. Caughlan & Oakley, 2001).

7.4. Fish welfare and ethics in monitoring

The importance of ethical issues relating to biological fieldwork and the need to minimize harm to species and ecosystems has repeatedly been emphasized (e.g. Bennett et al., 2016; Costello et al., 2016; Farnsworth & Rosovsky, 1993); a detailed consideration of these matters is beyond the scope of this review. We note, however, that fish welfare issues have received much attention (e.g. Sloman et al., 2019), often centred around the question of whether fish are sentient and can experience pain and suffering (e.g. Arlinghaus et al., 2007; Braithwaite, 2010; Huntingford et al., 2006, 2007; Rose et al., 2014) – a challenging question that has a number of implications in a scientific, ethical, and legal context (Browman et al., 2019). Browman et al. (2019) argue for a pragmatic approach using objective indicators of stress, health status, and behaviour to inform about fish well-being.

Irrespective of the scientific debate on fish-welfare, institutional requirements and legal regulations need to be considered during freshwater fish monitoring. Fish sampling usually requires specific permits from responsible authorities, particularly when working with protected species or in protected areas. Depending on the aim and sampling method, fish monitoring might involve the capture and treatment of fish or might even require methods of destructive sampling, i.e. the killing of fish (e.g. Blessing et al., 2010), such as when individuals require taxonomic identification in the laboratory, including where voucher specimens are required (Bortolus, 2008; Rocha et al., 2014; Section 7.2). However, alternative methods of identification should be used to avoid collection of rare species (Costello et al., 2016; Minteer et al., 2014). Protocols for fieldwork (e.g. Barbour et al., 1999; Brenkman et al., 2008; CCME, 2011; Cowx et al., 2009; Cowx & Fraser, 2003; Joy et al., 2013) typically provide guidelines on appropriate and least invasive techniques (e.g. non-capture techniques such as hydroacoustics and eDNA where applicable, Section 6.4) and are designed to minimize stress or damage caused by catching, handling, and holding. Developmetal stage and species differences are also taken into account . The sampling method and design should consider trade-offs of the potential harm to fish versus the quality of the obtained data in relation to sampling efficiency. In particular, when capture techniques are applied, potential cumulative effects should be paid specific attention as fish monitoring involves repeated sampling of species that can be long-lived (> 20 years) and is often targeted for protected or endangered species (Benejam et al., 2012). For example, an efficient and common capture technique such as electrofishing might cause sub-lethal injuries that are often not externally obvious and possibly fatal (Snyder, 2003). Moreover, ethical issues related to fish monitoring extend beyond fish-welfare and must also consider impacts on non-target species and ecosystems or the potential transmission of pests and/or invasive species (Costello et al., 2016).

8. Management of monitoring data

For the sustainable success of a monitoring programme and to potentially infer future changes, policies and procedures that guarantee the quality of data capture, documentation, and preservation for long-term use is required (Michener, 2015; Michener & Jones, 2012; Rüegg et al., 2014; Sutter et al., 2015). For example, Vines et al. (2014) found that the availability of research data declines with article age, with the probability of finding the dataset decreasing by 17% per year.

Although the importance of integrating data management into long-term ecological (monitoring) projects has been emphasized repeatedly in previous papers (Costello & Wieczorek, 2014; Sutter et al., 2015), this is often a neglected area in freshwater fish studies (but see Moe et al., 2013; Peterson et al., 2013 for some examples). Thoroughly considering data management to preserve data for long-term use and accessibility (even beyond the lifetime of the work that generated them) will require more time and resources to fish monitoring programmes and should be considered at the earliest stages and accounted for in budgetary plans.

Data management is not limited to 'what' was collected (i.e. fish sampling data); many other data often associated with sampling, such as geospatial information, multimedia content, voucher specimens, associated environmental variables, and other biological data, also need to be considered (Costello & Wieczorek, 2014). Furthermore, to ensure the utility of a dataset, it must be accompanied by metadata, i.e., a detailed description of who created the data, when and where the data were collected and stored, how and why the data were generated, processed, and analysed (Michener, 2006).

Data management is a key element in freshwater fish monitoring programmes. A detailed discussion of challenges and opportunities of data management, as well as practices of how it can or should be implemented in fish monitoring is provided elsewhere (Costello et al., 2013; Costello & Wieczorek, 2014; Michener & Brunt, 2000; Reichman et al., 2011; Sutter et al., 2015).

9. Conclusions

Given the rapid environmental degradation of the Earth's freshwater ecosystems and associated unprecedented rates of biodiversity change, the importance of robust, replicable, and effective programmes to monitor freshwater fish has never been higher. Future challenges related to habitat degradation, climate and land use change, and biological invasions necessitate monitoring programmes that systematically collect quality data allowing the potential detection of systemic shifts of populations or communities and thereby improve our understanding of ecosystem responses to environmental change. There is a pressing need for effective monitoring to comprehensibly quantify biodiversity change and to inform evidence-based environmental decision-making.

At a minimum, when establishing a monitoring programme, clear articulation of the monitoring aim(s) is essential and should address: (i) what should be monitored and how; (ii) how to allocate effort within time and across sites; (iii) establish criteria for data reliability; and (iv) identify practical constraints.

Monitoring must also take into account issues related to the detectability of species, taxonomy, and animal welfare. Additionally, monitoring programmes must integrate data management practices that ensure the quality of data capture, documentation, and preservation of information for long-term use and re-use.

In summary, careful reflection on aims(s) and the extent to which the data collected will meet these aims will greatly improve the quality and usefulness of monitoring data. Consistently high monitoring standards will improve data comparability within and amongst countries and systems. Finally, effective monitoring of freshwater fish will advance our overall understanding of freshwater ecosystems and contribute to the preservation and management of freshwater fish diversity while helping mitigate anthropogenic impacts.

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Tables

Table 1. Overview of key questions in fish monitoring programs, associated data needs and applicable sampling methods. Sampling method: 1 electrofishing, 2 netting, 3 trapping, 4 telemetry (e.g. acoustic, radio or passive integrated transponder tags), 5 mark-recapture, 6 environmental DNA, 7 hydroacoustic assessment, 8 angler catch statistics, 9 data-mining, 10 community science. -/orange = no, yellow = maybe, green = yes, na not applicable.

	Key questions in freshwater fish monitoring Detecting relevant changes/shifts/trends in …															
	Non-native species	Species distributional range	Phenology	Fish as ecological indicators	Food web structure	Fish behaviour	Species richness	Temporal Alpha-Diversity	Temporal Beta-Diversity	Population size and recruitment	Fishery performance	Productivity	Fish trait metrics	Genetic diversity	Diseases, Parasites	Size and/or age structure
Population / single-species																
Occupancy (presence only)	1-3,6,8-10	1-3,6,8-10	1-3,6,8-10	1-3,6,8	na	1-3	na	na	na	-	-	-	-	-	-	-
Presence / Absence	1-3,6	1-3,6	1-3,7	1-3,6	na	1-3	na	na	na	-	-	-	-	-	-	-
Counts, uncorrected for effort	1-3,7,8	1-3,8	1-3,7,8	1-3,7,8	na	1-3	na	na	na	1-3,5,7,8	1-3,5,7,8	1-3,7	-	-	-	-
Abundance estimate	1,2,5,7	1,2	1,2,5,7	1,2,5,7	na	1,2,5	na	na	na	1,2,5,7	1,2,5,7	1,2,7	-	-	-	-
Individual attributes	1-5	1-3	1-5	1-5	na	1-5	na	na	na	1-3,5	1-3,5	1-3	1-3	1-3	1-3	1-3,5
Community / multi-species																
Occupancy (presence only)	1-3,6	1-3,6	1-3	1-3,6	1-3,6	1-3	1,2,6	1,2,6	-	-	-	-	-	-	-	-
Presence / Absence	1-3,6	1-3,6	1-3	1-3,6	1-3,6	1-3	1,2,6	1,2,6	1,2,6	-	-	-	-	-	-	-
Counts, uncorrected for effort	1-3	1-3	1-3	1-3	1-3	1-3	1,2	1,2	1,2	1-3,5,7,8	1-3,5,7,8	1-3,7	-	-	-	-
Abundance estimate	1,2	1,2	1,2	1,2	1,2	1,2,5	1,2	1,2	1,2	1,2,5,7	1,2,5,7	1,2,7	-	-	-	-
Individual attributes	1-5	1-3	1-5	1-5	1-3	1-5	1,2	1,2	1,2	1-3,5	1-3,5	1-3	1-3	1-3	1-3	1-3,5

Figure legends

Fig. 1. Overview of fish monitoring programmes across global regions (A), taxonomic orders (B), and biotope types (C) based on records of the taxonomic order Osteichthyes (n = 543) in the Global Population Dynamics Database (GPDD, version 2.0, released 2010, www.imperial.ac.uk/cpb/gpdd2, NERC Centre for Population Biology, Imperial College, 2010). Note: The apparent lack of monitoring in, for example, Africa and Australia might reflect a limitation of the database rather than an actual lack of monitoring.

Fig. 2. Illustration of the variation of the number of species (species richness) and numerical abundance with sampling effort. The data are for two river sites in Trinidad (top – (A) Lower Aripo, bottom – (B) Maracas, sampled four times annually for five years. The data are described in Magurran *et al.* (2018). In each case the species (and numerical abundance) accumulation curves are constructed by randomly shuffling the temporal order of the samples a 1000 times. The open points represent the median value of the randomised accumulation curves; their 95% confidence limits (0.025 and 0.975 quantiles) are also shown (species richness – left column; numerical abundance – right column).

Fig. 3. The distribution of (A) Northern pike (*Esox lucius*) and (B) Zander (*Sander lucioperca*) in the UK, between 1986 and 2016, based on data from GBIF (www.gbif.org). The R code (R Core Team, 2017) used to construct the figure was adopted from the Coding Club (https://ourcodingclub.github.io/2017/03/20/seecc.html).