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. Freshwaters 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 its 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 will help programmes improve understanding 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

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 freshwaters cover less than 1% of the earth's surface, they support high levels of biodiversity (Dudgeon et al., 2006; Strayer & Dudgeon, 2010). Yet 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), but this highly diverse group includes 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 freshwaters and their catchments (Fausch, Karr, & Yant, 1984; Schiemer, 2000, but also see Magurran et al., 2018). The breadth of fundamental information on ecology and taxonomy, combined their higher societal importance compared to other freshwater taxa (Simon & Evans, 2017), makes freshwater fish a popular target taxon in assessments of ecological integrity. 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 monitoring of fish populations and communities in freshwater habitats and understanding the rate and direction of biodiversity change over time is essential.

Although the need for effective monitoring in ecological research is well-recognized, there is a long history of monitoring programmes that have been poorly planned and lack focus, resulting in ineffective programmes that rarely meet their aims (Lindenmayer & Likens, 2009, 2010; Marsh & Trenham, 2008; Nichols & Williams, 2006). In fact, there remains a series of issues and knowledge gaps with how these programmes are designed and implemented. For example, there is considerable disparity in their implementation between developed and developing regions. 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 that must be explicit (Pollock et al., 2002), but these are often poorly quantified, or justified, resulting in long-term data lacking statistical power. 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 focus is primarily on riverine fishes, as the majority of long-term freshwater fish monitoring programmes are river-based. 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); b) synthesising issues with sampling design and statistical power, and indicating how they might be overcome (Section 4); c) reviewing different monitoring and sampling approaches (Section 5); d) considering challenges related to species' detectability, taxonomy, economical costs, and ethics (Section 6); and, e) discussing the importance of the appropriate management of monitoring data (Section 7). We start by providing some key definitions and background information (Section 2).

2. Definitions and background

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.**

2.1. History of fish monitoring

There is a long history of monitoring programmes that have provided important scientific advances and crucial information for environmental policy (Lovett et al., 2007), which has also been increasingly reflected in the scientific literature (Fig. S1.1). Very 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 majority of fish monitoring programmes were 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 fish occur in aquatic environments. Thus, in contrast to many terrestrial biota, that can be monitored by visual observations and where citizen scientists can be more 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 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).

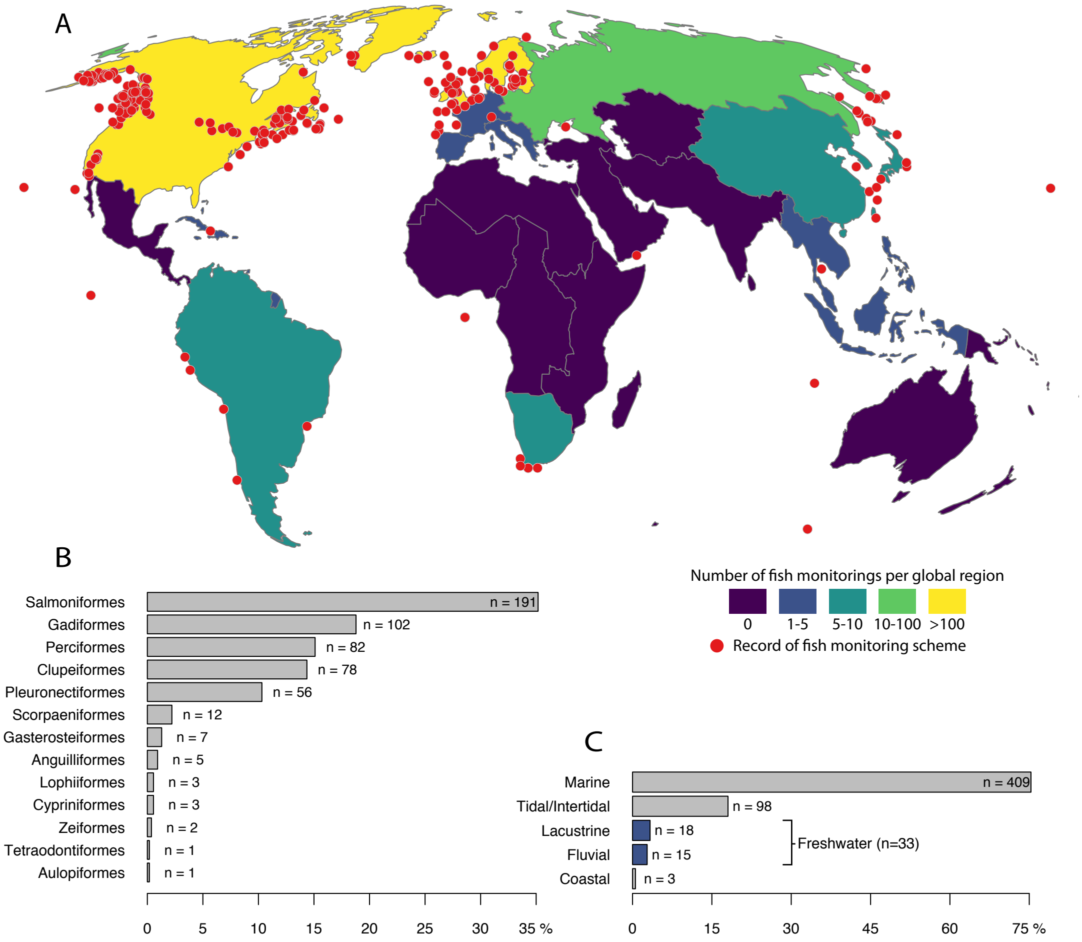

Fig. 1. Overview of fish monitoring programmes across global regions (A), taxonomic orders (B), and biotope types (C) based on records of the taxonomic order Osteichthyes (n = 543) in the Global Population Dynamics Database (GPDD, version 2.0, released 2010, www.imperial.ac.uk/cpb/gpdd2, NERC Centre for Population Biology, Imperial College, 2010).

3. Different questions lead to different monitoring approaches

3.1. Key questions

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). The overall goal in monitoring freshwater fish is thus not to document change *per se*, but rather to understand how much of the observed change is due to anthropogenic impacts. In particular, effective monitoring should facilitate the identification of drivers of systemic change (Dornelas et al., 2012). Linked to the overall goal of detecting systemic changes, the key questions of freshwater fish monitoring relate to detecting significant changes at the (i) community level (multi-species), such as quantifying trends in species richness, temporal α- and β-diversity, functional diversity, food web structure, and/or at the (ii) population level (single species), such as 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.

The diverse questions that can be addressed via monitoring necessitate different sampling designs. For example, some questions can be addressed with presence-only data, while other questions require sampling of an entire community (Table 1). Depending on the entity being measured, this might involve various fish capture techniques (see Zale et al., 2012), methods to assess fish spatial behaviour (see Lucas & Baras, 2000), genetic methods (Lundqvist et al., 2010), or more recent approaches such as citizen science and the use of social media (Section 5). For example, by monitoring fish communities (presence/absence of multiple species) in two rivers in the south-eastern of the U.S.A. over 20 years, Freeman et al. (2017) revealed important temporal declines in species' occupancies and overall species richness. By comparison, Hansen et al. (1986) monitored reported catches (i.e., rough abundance estimates) of Atlantic salmon in a Norwegian river to track changes in stock sizes over 100 years. In Table 1, we summarize the data needs associated with a suite of key monitoring questions. We also stress the importance of clearly articulating the question that needs to be answered, and of ensuring that the data provided by the monitoring are suitable for answering it. These points are developed in the next section.

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 hydro-acoustic assessment, 8 angler catch statistics, 9 data-mining, 10 citizen 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

3.2. Aims of monitoring

Effective monitoring (also termed 'target' or 'focused' monitoring) requires a clear set of specific objectives linked to the overall goal of detecting important shifts in fish populations or communities over time and space. Monitoring may additionally be guided by *a priori* hypotheses (Nichols & Williams, 2006). In particular, question-driven monitoring programmes need a rigorous study design and 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). In mandated monitoring programmes, the data might be 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). Whilst each fish monitoring programme is unique to a given system and its overall aim, there are several commonalities across systems and studies (see Supporting Information S2). Here, we focus on question-driven monitoring of freshwater fishes, and we use the term 'monitoring' in this more narrow sense in the following discussion.

Clear articulation of the monitoring aim(s) is essential (Bisbal, 2001; Lindenmayer & Likens, 2009). At a minimum, these aims should: define what should be monitored (e.g. fish abundance, fish attributes); define the spatial and temporal scope (e.g. duration, scale; *cf.* Dixon & Chiswell, 1996); establish criteria for reliability (e.g. precision, power); and identify practical constraints (e.g. human resources, costs, social conflicts).

4. Sampling design, network design and statistical power

The sampling and network design, and statistical power, of monitoring programmes are crucial to their success and effectiveness. In this context, the sampling design relates to the temporal frequency of sampling within a designed network comprising a series of spatially segregated sites. Consequently, to answer the monitoring question requires *a priori* decisions regarding how to allocate effort within and among years, and across sites (Larsen et al., 2001).

Although often difficult to implement in large-scale ecological studies, the basic principles of experimental design (e.g. Quinn & Keough, 2002) are generally also applicable to monitoring (Conroy & Carroll, 2009). These principles include replication (to ensure representativeness and assess variability), control (to identify and allow comparisons with baselines), and randomization (to enhance the independence of errors, García-Berthou *et al.*, 2009). However, as fish monitoring programmes are typically undertaken to detect temporal or spatial changes in populations (Cowx et al., 2009), statistical controls and replication are often unfeasible (Carpenter et al., 1989; Hargrove & Pickering, 1992; Schindler, 1998; Turner et al., 2001). Instead, other statistical techniques, such as regression analysis (García-Berthou et al., 2009; Hurlbert, 2004; Osenberg et al., 2006) or before-after control-impact designs (Osenberg et al., 2006; Stewart-Oaten & Bence, 2001; Thiault et al., 2017), are frequently used to estimate effects in cases without spatial replication. Moreover, descriptive statistics or exploratory multivariate techniques have less rigorous assumptions, and often might be appropriate for analysing monitoring data where formal hypothesis testing is not required (Økland, 2007).

The spatial distribution of the sampling sites should match the monitoring aim(s) (Dixon & Chiswell, 1996). Two major principles, the avoidance of bias in the selection procedure and achievement of high precision, should underlie all sampling designs (Crawford, 1997). A sampling design can be based on probabilistic or non-probabilistic methods (Fig. 2, for details see Supporting Information S3). 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, sampling sites are frequently selected non-probabilistically, often based on judgment or convenience (Pope et al., 2010; Wilde & Fisher, 1996). The adaptive approach (Larsen et al., 2001) argues that the sampling design should be re-evaluated and re-designed as necessary as data are gathered and their variability analysed (Box 1). 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).

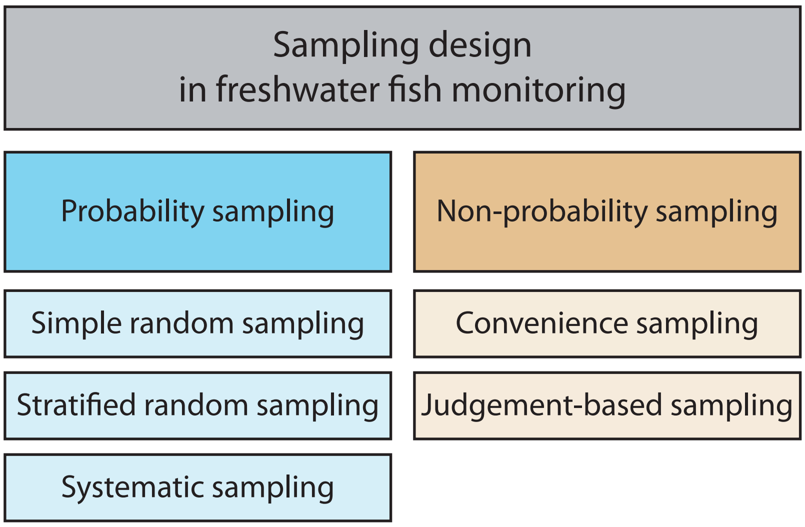

Fig. 2. Overview of possible sampling designs in freshwater fish monitoring (see also Supporting Information S3).

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). Thus, monitoring programmes must be capable of providing data suitable for the continued management of the resources (Polasky et al., 2011). Given on-going environmental change, the decision-making approach of adaptive management, based on 'learning by doing', is generally a preferred option to integrate scientific knowledge into the policy-making process (Ludwig et al., 2001). Within the framework of adaptive management, adaptive monitoring tends to be presented as a new paradigm, which views long-term monitoring as a management activity closely related to scientific research. 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). Adaptive management requires the integration of long-term monitoring programmes and cause/effect-based experimentation, allied with modelling frameworks that prioritize strategies that shift the ecosystem towards ecological and socioeconomic stable states. Adaptive monitoring thus has the potential to significantly improve the poor record of high-quality, long-term ecological research and monitoring.

An example of adaptive monitoring is outlined by Fölster et al. (2014) for Swedish freshwaters. Starting with the work of early naturalists measuring rather specific and localized natural phenomena, 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 demands related 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, test resilience theory, or predict regime shifts and tipping points.

Adequate sampling frequency depends on the aim of the monitoring programme, the relative importance of a sampling location, and the expected data variability (Canter, 1985; Strobl & Robillard, 2008). The latter is particularly important, as fish monitoring programmes strive to detect 'real' trends and changes, as opposed to stochastic variation (e.g. resulting from inter-annual variation in recruitment) and baseline turnover (Dornelas et al., 2014). Here, analysis of statistical power should avoid Type II errors, i.e. the probability of not detecting a trend, when in fact there is one (Fairweather, 1991; Miller et al., 2009).

Power analyses should be considered *a priori* during the planning of the monitoring programme (Legg & Nagy, 2006; Marsh & Trenham, 2008; Maxwell & Jennings, 2005; Peterman, 1990). They can guide the development of an effective sampling and network design, as well as 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 over time and/or space (Peterman, 1990; Steidl et al., 1997). *A posteriori* power analyses are more controversial (Hayes & Steidl, 1997; Hoenig & Heisey, 2001; Thomas, 1997). These compute the statistical power of a study after it has been conducted and a non-significant result (i.e. failure to reject the null hypothesis) obtained (Peterman, 1990; Thomas, 1997). There are some examples that have applied *a priori* power analysis in freshwater fish monitoring (e.g. Liermann & Roni, 2008; Maxell, 1999). Several other studies have highlighted the low statistical power of many programmes (Maxwell & Jennings, 2005; Wagner et al., 2013) or the failure to consider statistical power (Marsh & Trenham, 2008). Critical design errors can be problematic and '*no amount of statistical "magic" will remedy a faulty study design*' (Conroy & Carroll, 2009).

Consequently, the final sampling design should establish the temporal frequency of sampling across a spatial network of sites, with these determined according to *a priori* statistical power analysis. The next step is then selecting the sampling methods required to collect the monitoring data needed to address the programme's aims (Section 5.1).

5. Approaches to fish monitoring

5.1. Monitoring aims versus sampling methods

Among the sampling methods that can be utilised for fish monitoring, distinctions can be made between capture and non-capture techniques. 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 (Casselman et al., 1990). Non-capture methods (e.g. hydro-acoustic 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 5.4) or through erroneous identification of specimens (Section 6.2).

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 larval fishes (Copp, 2010) and the status of the critically endangered European eel *Anguilla anguilla* (Laffaille et al., 2005). However, capturing fish in longer river reaches using electrofishing or trawling might be more suitable where the monitoring aim is to assess biological/ecological integrity, as the 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).

5.2. Capture techniques and application within monitoring programmes

The application of capture methods requires determination of the sampling effort required for accurately estimating the composition of the assemblage (details in Box 2). The applicability of the different capture techniques available to monitoring programmes (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 (Table S4.1), and so these are not discussed further here. However, two fundamental concepts have emerged in relation to the application of these techniques and protocols to river fish monitoring: the importance of sampling design (already discussed in Section 4) 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. 3). As a result, any metrics which 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 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 be based on data that have been statistically standardized (by rarefaction or similar) – see Fig. 3 for an example.

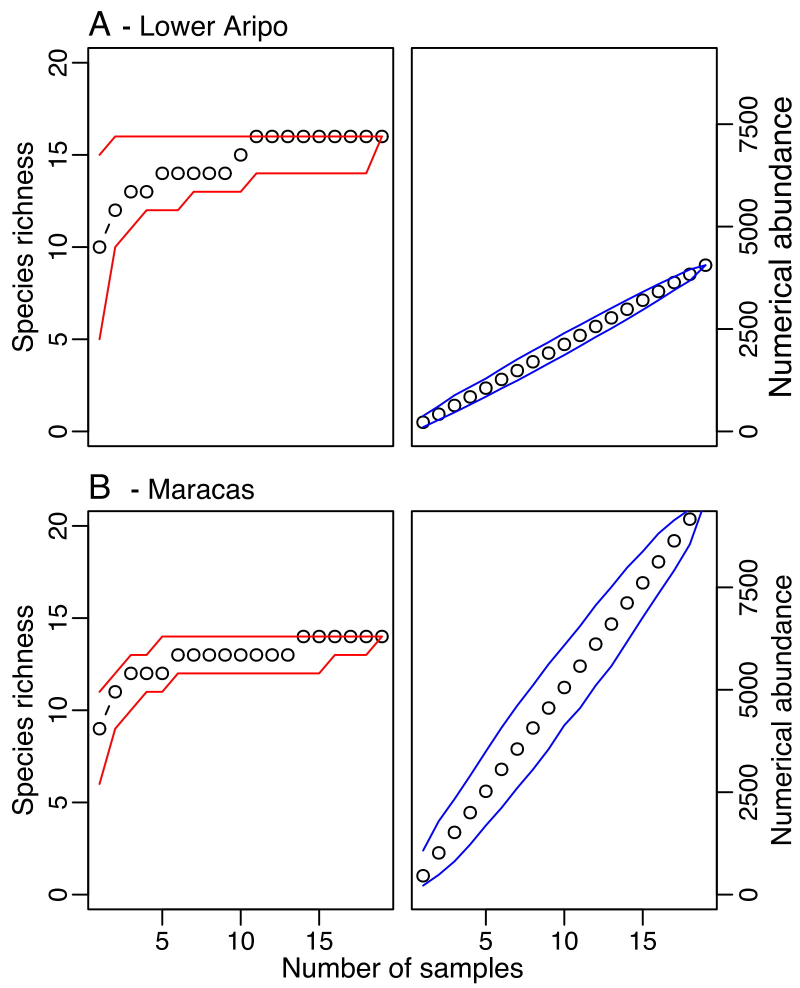

Fig. 3. Illustration of the variation of the number of species (species richness) and numerical abundance with sampling effort. The data are for two river sites in Trinidad (top – (A) Lower Aripo, bottom – (B) Maracas, sampled four times annually for five years. The data are described in Magurran *et 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 - red lines; numerical abundance - blue lines).

5.3. Applications of capture methods to monitoring fish movement and behaviour

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).

5.4. Non-capture monitoring techniques

Monitoring programmes can incorporate non-capture methods to complement capture data. These methods include environmental DNA, hydro-acoustic assessments, angler catch statistics, and data-mining exercises. 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/absence 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 reliability of species’ detection (Hänfling et al., 2016), but some questions remain on, for example, 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 the 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 remain over the reliability of eDNA to provide estimates of abundance, these are now starting to be overcome (Lacoursière-Roussel et al., 2016).

Hydro-acoustic 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, hydro-acoustic 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 hydro-acoustic 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, such as the Chinese paddlefish *Psephurus gladius* in the upper Yangtze River, China (Zhang et al., 2009).

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; https://www.gbif.org/) and the Global Population Dynamics Database (GPDD; https://www.imperial.ac.uk/cpb/gpdd2/secure/login.aspx) 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 large variety of ecological data.

An alternative method to using these online databases is monitoring the distribution of fishes via citizen science, particularly via social media platforms. Indeed, the application of citizen science and crowd sourcing to the collection of biological data is increasingly frequent (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. 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. 4) and thus temporal and spatial distribution patterns better understood (Coding Club, 2018).

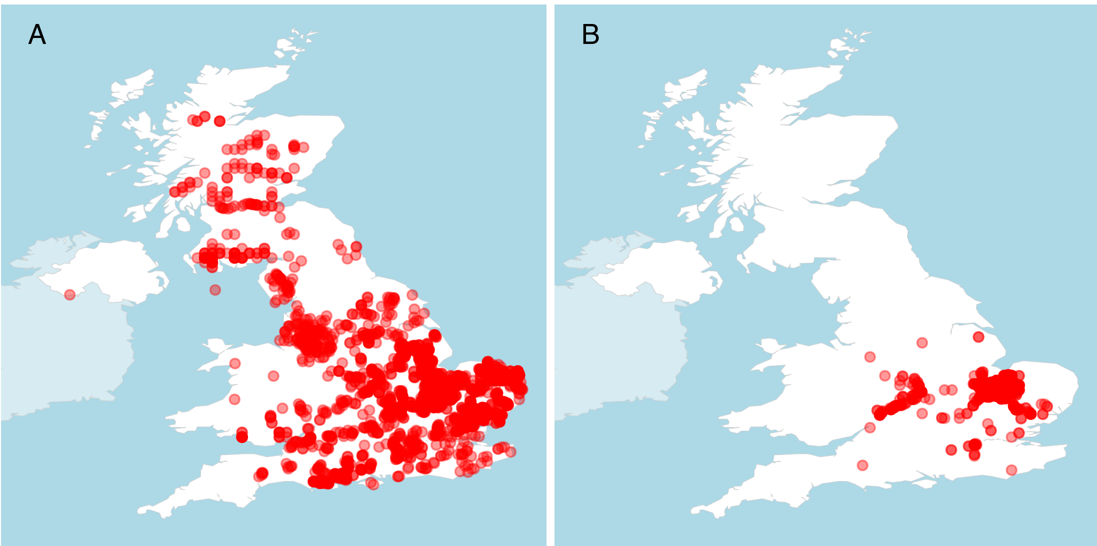

Fig. 4. The distribution of (A) Northern pike (*Esox lucius*) and (B) Zander (*Sander lucioperca*) in the UK, between 1986 and 2016, based on data from GBIF (www.gbif.org). The R code (R Core Team, 2017) used to construct the figure was adopted from the Coding Club (https://ourcodingclub.github.io/2017/03/20/seecc.html).

5.5. 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, hydro-acoustics, 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 other systems (e.g. Vindenes et al., 2014; Winfield et al., 2010).

6. Major challenges in fish monitoring

6.1. 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. Potential solutions to this problem include modelling occupancy, estimating the probability of detection of species (and/or individuals) through mark-recapture or distance sampling, 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).

Occupancy methods (MacKenzie et al., 2002, 2003, 2006) draw on presence/absence information and necessitate repeated (at least two, but ideally substantially more) samples at a site (assuming no underlying change in the community between samples). However, it can be challenging to disentangle occupancy from detection, with McGill (2014) arguing that 'ignoring detection ensures bias', can, under certain conditions, result in a more accurate occupancy estimate than one based on detection probabilities (McGill, 2014). In addition, occupancy based methods are generally unsuitable when evaluating changes in abundance metrics (but see Iknayan et al., 2014). In addition, for freshwater fish, if repeated sampling has an adverse impact on the organisms involved (as may happen, for example, if the same individual fish are repeatedly electro-fished over a short time period (Gatz & Linder, 2008), this sampling may itself lead to shifts in structure and diversity of the assemblage being studied. A new generation of occupancy models (Iknayan et al., 2014) may provide solutions to some of these concerns, but one of their main assumptions – that sites are closed to immigration and local extinction over replicated surveys – makes their application problematic in open habitats (such as rivers), and in studies where the quantification of temporal turnover is the aim.

Detectability can also be estimated using mark-recapture methods and distance sampling (Buckland et al., 2011). Mark-recapture (Borchers, Buckland, & Zucchini, 2002; Borchers *et al.*, 2015; Section 5.3) is widely used and informative although it makes a number of assumptions. Distance sampling (Buckland et al., 2001, 2004, 2011) typically involves the investigator noting the distance of each individual from a transect or point. Although distance sampling is an effective method of accounting for detectability where the investigator can locate and identify each individual by sight (e.g. birds or trees), it is not workable in most freshwater surveys that encompass multiple taxa, many of which cannot be identified in-situ. Furthermore, these methods fit a detection function to each species in the assemblage and use this information in the calculation of diversity statistics. However, detection functions cannot be fitted for rare species, which must either be excluded from the analysis or assumed to have the same detectability (Buckland et al., 2011).

As Buckland *et al.* (2011) note, *'Ignoring detectability might not be a major problem if bias is consistent over time or space'*. Adopting the same methodology throughout, and comparing sites (e.g. river sections with similar dimensions, water depth, and substratum) where biases in capture probabilities can be minimized, may be the pragmatic solution to detectability issues in many cases. But it is important that investigators are aware that their data sets will contain biases, and to be confident that data quality is sufficient to answer the question being posed. Repeat sampling in at least some localities, and comparison of different sampling methodologies (Deacon et al., 2017; de Paiva Affonso et al., 2016) is helpful in understanding detectability issues in the context of a given study system.

6.2. 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 the need of 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 frequency and consequences of species misidentification tends to be rarely investigated for freshwater fish when compared to taxonomically more challenging groups, such as stream invertebrates. Stribling et al. (2008) compared taxonomic identification of stream macro-invertebrates across eight U.S. laboratories and found averages of 21% taxonomic disagreement. Similarly, Haase et al. (2006, 2010) identified considerable errors in species sorting and identification of stream macro-invertebrates among European laboratories. 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, which requires thorough validation procedures (Poizat & Baran, 1997; Valbo-Jørgensen & Poulsen, 2000).

A similar problem is when taxonomy changes and it is realised that what was previously referred to as a single species comprises actually several cryptic species. This problem is increasingly frequent given the increasing power of molecular tools. New taxonomic alignments hinder comparison with old samples if no specimens were preserved. In addition, the same species 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 (Knebelsberger et al., 2015) may be incorrectly identified to species level in some databases. 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).

6.3. Economic costs

For a monitoring programme to be effective, successful and sustainable over long-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 clear aims of what to monitor (Lindenmayer & Likens, 2009, 2010; Section 3) and a proper selection of relevant variables that need to be measured (Braun & Reynolds, 2012; Section 5.1). 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. In this regard, 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 traditional methods in fish monitoring, such as field-based capture methods (e.g. electrofishing, netting, trapping), are commonly labour intensive and thus costly, the financial costs of emerging methods can be lower, such as use of eDNA, the automatized collection of data (e.g. hydro-acoustic assessments), and the use of citizen science (including use of angler catches), mining social media, and managing and analysing big data (Section 5.4). A detailed review of the costs associated with ecological monitoring can be found elsewhere (e.g. Caughlan & Oakley, 2001).

6.4. Ethics

Depending on the aim and sampling method, fish monitoring might involve the capture and treatment of fish directly impacting their welfare or might even require destructive sampling, such as when individuals require taxonomic identification in the laboratory, including where voucher specimens are required (Bortolus, 2008; Rocha et al., 2014; Section 6.2). Nevertheless, harming or sacrificing fish has strong ethical implications and potential conservation impacts, and should be carefully considered and minimized where possible (Bennett et al., 2016; Blessing et al., 2010; Costello et al., 2016). This is particularly important, as fish monitoring involves repeated sampling of species that can be long-lived (> 20 years) and is often targeted for protected or endangered species. Fish surveys and monitoring programmes involving capture methods commonly require specific permits from responsible authorities, especially when working with protected species or in protected areas.

The impact on fish welfare depends on the sampling method used (Joy et al., 2013), ranging from low impact (e.g. spotlighting, hand-seining) to moderate (e.g. electrofishing) and high (e.g. gillnets, rotenone) impact methods (CCME, 2011; Deacon et al., 2017; Joy et al., 2013). The sampling method and design should thus consider trade-offs of the potential harm to fish versus the quality of the obtained data in relation to sampling efficiency. Many studies and protocols suggest how fish should be handled to minimize stress or damage caused by catching, handling, and holding (Barbour et al., 1999; Brenkman et al., 2008; CCME, 2011; Cowx et al., 2009; Cowx & Fraser, 2003; Joy et al., 2013). It is recommended that fish are held in the least stressful conditions possible, i.e. in shaded buckets with ambient temperature stream water, with supplementary aeration, separating predators from their prey, and at low densities. Fish must not be handled with dry hands to minimise damage and the use of anaesthetics might be needed for certain procedures (e.g. for marking fish). After completing measurements, fish should be released near their point of capture in a calm area near the bank.

Fish sampling can also cause sub-lethal effects. For example, electrofishing with alternating current or high-frequency pulsed direct current might harm fish and cause internal injuries that are often not externally obvious and possibly fatal (Snyder, 2003). Potential cumulative sub-lethal effects should be paid specific attention in fish monitoring with repeated samplings over time (Benejam et al., 2012). Increasingly, many non-capture methods are becoming available such as hydro-acoustics and eDNA (Section 5.4). Where capture techniques are needed for obtaining tissues for genetic and stable isotope analyses, use of fin biopsies and scales provide non-lethal methods (Busst et al., 2015; Busst & Britton, 2018). Gastric evacuation and genetic analyses of faeces for diet studies can also replace the need for stomach contents analyses of sacrificed fish (Jo et al., 2014).

7. Management of monitoring data

To draw meaningful conclusions from monitoring results 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 and needs to be conceptualized in a data management plan (Michener, 2015; Michener & Jones, 2012; Rüegg et al., 2014; Sutter et al., 2015). Free online platforms can facilitate the elaboration of data management plans such as the DMPTool (https://dmptool.org) or DMPonline (https://dmponline.dcc.ac.uk).

Many practical considerations of data collection, such as the design of field forms, are important and should mimic a logical workflow with explicit reminders of units, format, measurement precision, and codes with a unique identifier (Borer et al., 2009; White et al., 2013). Many other data often associated with fish sampling, such as geospatial information, multimedia content, voucher specimens, associated environmental variables, and other biological data, also need to be managed accordingly (Costello & Wieczorek, 2014). In terms of quality assurance and quality control, verification, validation, and certification are important to minimize or prevent errors in data sets in the field, the laboratory, or at the computer. This might include visualizing data, identifying missing values, detecting illogical combinations and possible inconsistencies in the data (Sutter et al., 2015). For the correct use and interpretation 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). Information standards, such as the Ecological Metadata Language (EML) (Fegraus et al., 2005; Michener et al., 1997) facilitate the use and integration of ecological data by providing a detailed and machine-readable description of the structure of data tables (Jones et al., 2006). The Humboldt core (Guralnick et al., 2018) is a set of standards and terms that allows to document the sampling data and also other environmental characteristics such as dataset information, spatial and temporal resolution, habitat, taxonomic coverage, methodology, effort, and completeness.

For the sustainable success of a monitoring programme, it is also important to preserve data for a long-term use. For example, Vines et al. (2014) estimated that the availability of research data declines with article age, with the probability of finding the dataset decreasing by 17% per year. Therefore, data and metadata should be stored in non-proprietary formats (e.g. csv, xml, txt, tiff), preferably in a scientific repository (e.g. institutional repository, thematic repositories such as GBIF, or others such as DataOne, Dryad, Figshare, Mendeley Data, Re3data, or Zenodo) (Hart et al., 2016; Sutter et al., 2015). A unique and persistent identifier such as the Digital Object Identifier (DOI) is necessary for citation and reuse of data. Another emerging option is to publish the data in journals that publish data papers (Chavan & Penev, 2011), which has the additional advantage of peer-review before publication (Costello et al., 2013; Costello & Wieczorek, 2014; Kratz & Strasser, 2015). Examples of data papers on freshwater fish are increasingly available (e.g. Brosse et al., 2013; Rodeles et al., 2016; Tedesco et al., 2017).

In summary, there are many recent improvements in data management science that could benefit ecological monitoring in general and seem scarcely applied for freshwater fish studies (but see Moe et al., 2013; Peterson et al., 2013 for some examples). Thoroughly considering data management will demand more time and resources to fish monitoring programmes, but could enormously benefit their quality, outputs, and re-use to explore larger-scale patterns and trends.

8. 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, a clear articulation of the monitoring aim(s) is essential and should include defining: (i) what should be monitored and how; (ii) how to allocate effort within time and across sites; (iii) establishing criteria for data reliability; and (iv) identifying practical constraints. Therefore, effective monitoring necessitates making decisions about the capture and/or non-capture sampling methods and the sampling design – both of which are explicitly described and discussed in this review – ensuring that the data provided by the monitoring are suitable for answering the questions posed.

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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