The Water Framework Directive: advances in fish classification tools in Ireland

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ABSTRACT

The Water Framework Directive (WFD) has established the concept of ecological quality as a method to improve European Union (EU) surface and ground waters. Ecological quality status is based on the composition and abundance of different biological quality elements, including fish fauna, with the supporting elements of hydromorphology and chemical and physico-chemical parameters. Monitoring for fish for the WFD began in Ireland in 2007. In parallel, classification tools were developed or refined for each surface water type (lakes, rivers and transitional waters) and then intercalibrated in a cross-Europe exercise to ensure consistency across all EU states. The development and basic concepts of three WFD-compliant ecological classification tools for fish and the cross-Europe intercalibration exercise are described for rivers, lakes and transitional waters.

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INTRODUCTION

The European Union (EU) Water Framework Directive (WFD) 2000/60/EC (European Parliament and Council, 2000), adopted in 2000, is one of the most significant pieces of legislation covering conservation of aquatic ecosystems enacted in Europe (Walsh, 2005). The WFD takes a holistic approach to water management and applies to all surface waters (rivers, lakes, estuaries, coastal waters) as well as to groundwater. It aims to prevent further deterioration of aquatic ecosystems and to protect and enhance their status. The specific objectives of the WFD include the achievement of 'Good ecological status' (defined as slight departure from High-status 'reference conditions' principally in terms of plants, invertebrates and fish), no deterioration in the present class and meeting the objectives of protected areas (European Parliament and Council, 2000).

The WFD is not perfect and has its critics (Knepper, 2006; Keskitalo, 2010). Some feel that its approach to ecology is out of date; others that it makes too many assumptions about which groups of organisms are the most sensitive in terms of particular environmental pressures (Knepper, 2006; Keskitalo, 2010). But if nothing else, the WFD has achieved three important changes in direction. Firstly it has moved the EU formally beyond merely assessing chemical water quality, to include issues related to hydrology and morphology, expressed in a wide range of human pressures (e.g. nutrient enrichment, acidification, organic pollution, chemical pollutants, abstraction, flow regulation, river engineering). Secondly, it has shifted the focus of aquatic monitoring from one largely centred on macroinvertebrates to include composition and abundance of aquatic flora (i.e. phytoplankton, macrophytes, phytobenthos) and composition, abundance and age structure of fish fauna. It has also broadened physico-chemical monitoring from its traditional focus on pollution, to incorporate hydromorphological elements (e.g. hydrological regime, river continuity, substrate conditions, structure and condition of riparian zone) and physico-chemical elements (nutrient concentrations, pH, oxygen levels and specific pollutants).

One of the principal tasks in implementing the WFD is to evaluate the ecological status of water bodies. Under the WFD, a water body may be a stretch of river, a lake or part of a lake or a stretch of estuary or coastal water. There are five classes (High, Good, Moderate, Poor and Bad). Since the adoption of the WFD in 2000 substantial progress has been made in the ecological assessment of European waters. Many European countries now have a set of assessment tools for evaluating the state of water and for monitoring improvements in relation to investments in river basin management plans or deterioration in response to future environmental changes (Birk *et al.*, 2012; Brucet *et al.*, 2013; Poikane *et al.*, 2015). The outputs of the ecological assessment tools are expressed numerically as ecological quality ratios (EQRs) in the range between 1 and 0, with High ecological status represented by values close to 1 and Bad ecological status by values close to 0 (Environmental Protection Agency (EPA), 2007). The EQR scale is divided into five classes by assigning a numerical

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value to each of the boundaries between classes. The values for the boundary between the classes of High and Good status and between Good and Moderate status have been established through an EU-supported intercalibration exercise. This exercise attempted to ensure comparability of the results of biological monitoring across Member States (MSs) for High, Good and Moderate status (EPA, 2007).

Champ *et al.* (2009) described the research undertaken in Ireland between 2000 and 2008 to deliver standardised sampling methods for three water-body types (rivers, lakes and transitional waters) where fish are monitored for the WFD. Since then significant progress has been made in understanding the relationship of fish with various pressures in rivers, lakes and estuaries and in developing WFD-compliant fish classification tools, intercalibrating them in a cross-Europe exercise, calculating and reporting status for respective water bodies and interpreting the data (sense checks etc.). This paper describes the research undertaken in Ireland to develop these WFD-compliant fish classification tools for rivers, lakes and transitional waters, basic concepts and, briefly, the intercalibration process.

FISH IN LAKES CLASSIFICATION TOOL (FIL2)

SAMPLING FISH IN LAKES

Fish sampling for WFD monitoring in lakes in Ireland is conducted using standard Nordic monofilament multimesh benthic and surface floating survey gill nets (30m × 1.5m, twelve panels, mesh size ranging from 5mm to 55mm) and double fyke nets (2 × 0.55m diameter front hoops × 8m leader; three nets combined to form one fishing unit), and the sampling effort is supplemented using single panel larger mesh (62.5mm knot to knot) multifilament survey gill nets (27.5m × 2.0m) in moderate- and high-alkalinity lakes (Kelly *et al.*, 2012). The gill netting procedure was in accordance with a modified version of the European standard multimesh gill-netting method (CEN, 2005), which was adapted by Inland Fisheries Ireland (IFI) for WFD fish monitoring in Irish lakes (Kelly *et al.*, 2008a).

FISH IN LAKES AND WATER QUALITY

Prior to developing the fish in lakes classification tools the fish metric data were analysed to investigate if they showed any significant patterns in relation to changes in water quality in Irish lakes. The general trend is for fish species richness in lakes in Ireland to increase with trophic status progression from oligotrophy to eutrophy and to decrease slightly in hypertrophic lakes (Kelly *et al.*, 2008a,b). There is also a negative relationship between percentage composition and abundance (as indicated by catch per unit effort (CPUE—mean number of fish per metre of net) and biomass per unit effort (BPUE—mean biomass of fish per metre of net)) of intolerant fish species (i.e. salmon, *Salmo salar* Linnaeus, 1758; brown trout/sea trout, *Salmo trutta* Linnaeus, 1758; and Arctic char, *Salvelinus alpinus* Linnaeus, 1758) and trophic status in Irish lakes (Kelly *et al.*, 2008b; Olin *et al.*, 2014). In contrast there is a positive relationship between percentage composition and abundance of tolerant fish species (i.e. roach, *Rutilus rutilus* Linnaeus, 1758; bream, *Abramis brama* Linnaeus, 1758; perch, *Perca fluviatilis* Linnaeus, 1758; rudd, *Scardinius erythropthalmus* Linnaeus, 1758; tench, *Tinca tinca* Linnaeus, 1758; roach × bream hybrids, roach × rudd hybrids and pike, *Esox lucius* Linnaeus, 1758) and trophic status (Kelly *et al.*, 2008a,b; Olin *et al.*, 2014). This pattern of increasing abundance of tolerant fish species along a phosphorus gradient has been documented by many authors (Jeppesen *et al.*, 1990; Olin *et al.*, 2002). This change in community structure from intolerant species to tolerant species has also been noted within individual lakes in the Irish midlands, i.e. Loughs Sheelin (O'Grady and Delanty, 2008), Ramor and Gowna, Co. Cavan; however, it will only occur in Irish waters to which tolerant species (e.g. roach) have been introduced and where there is a concurrent deterioration in the water quality status (Kelly *et al.*, 2008a).

DESCRIPTION OF THE FISH IN LAKES CLASSIFICATION TOOL(S) (FIL1 AND FIL2)

The development of a WFD-compliant classification tool for fish in lakes in Ireland formed two iterations; the first fish in lakes tool (FIL1) was developed during an Interreg-funded project, NS SHARE, that was set up to deliver the objectives of the WFD within the NS SHARE river basin districts (2004 to 2008) (Champ *et al.*, 2009). An important part of this project was the development of WFD-compliant classification tools for various biota, including fish (Champ *et al.*, 2009; Kelly *et al.*, 2008a). FIL1 followed a predictive multimetric approach; 145 potential fish metrics relating to abundance, species composition and age structure were calculated and redundant metrics were then excluded from the models where there was no correlation with total phosphorus, lakes were pre-assigned to a fish community type (salmonid, perch or roach) and each lake was assigned to a qualitative ecological quality status class (High, Good, Moderate, Poor or Bad) using discriminant analysis (Kelly *et al.*, 2008a; 2012). No WFD-compliant ecological quality ratios were generated (Kelly *et al.*, 2008a).

A second ecological classification tool project was initiated by IFI in 2010 to develop a WFD-compliant classification tool for fish in lakes, i.e. to generate EQRs between 1 and 0. Agencies from Ecoregion 17 (Republic of Ireland and Northern Ireland) contributed data from 137 lakes (151 sampling occasions) for model development (Kelly *et al.*, 2012) and IFI commissioned the Agri-Food Biosciences Institute (AFBI) Northern Ireland to undertake the model development. A range of lake types and trophic levels (oligotrophic to hypertrophic) were included in the dataset (Kelly *et al.*, 2012). A lake typology relevant to fish populations in lakes from Ecoregion 17 was produced as part of the ecological classification tool development. Four lake types were determined based on fish metrics and abiotic variables from 'reference' lakes using cluster analysis and stepwise discriminant analysis. The specific lake fish typology categorised lakes into low (≤ 67 $CaCO_3$ mg l^{-1}) or High (> 67 $CaCO_3$ mg l^{-1}) alkalinity, and shallow (≤ 17m) or deep (> 17m maximum depth) (Kelly *et al.*, 2012). The subsequent ecological classification tool follows a novel multimetric predictive approach assigning ecological status to a lake using two independent models (discriminant classification rules and a generalised linear model). It is recommended that both methods are used to validate output and cross-check and highlight potential misclassification. The tool defines fish ecological status of a lake using thirteen fish metrics that were chosen for their ecological relevance, ease of measurement and ability to meet the requirements of the WFD (Kelly *et al.*, 2012). Total phosphorus (mean) and chlorophyll a (max) were used as the pressure variables and indicators of water quality. WFD-compliant quantitative EQRs between 0 and 1 are calculated with associated confidence intervals (Kelly *et al.*, 2012). To determine class boundaries the results of the qualitative classification rule and quantitative EQR model were cross-tabulated at various cut-points of the models.

The outputs generated by the tool are:

1. a qualitative ecological status class based on discriminant classification rules;
2. a quantitative EQR value with associated confidence values (between 1 and 0);
3. a descriptive ecological status class (High to Bad).

RELATIONSHIP OF EQR TO PRESSURE

The mean EQR of lakes classified as 'reference' (0.71 ± 0.042 (CI)) during the tool development was significantly higher than those classified as 'impacted' (0.43 ± 0.055 (CI)) (Kelly *et al.*, 2012; Olin *et al.*, 2014). There was a significant negative relationship between the fish EQR and the pressure index (mean total phosphorus and maximum chlorophyll a) and also between each pressure index class (High, Good, Moderate, Poor and Bad) (Kelly *et al.*, 2012) (Fig. 1). The relationship between the EQR and the pressure index for fish was slightly weaker than for some other biological elements (e.g. phytobenthos, phytoplankton and macrophytes (Free *et al.*, 2006; 2016)) derived in Ireland and suggests that factors other than eutrophication are also influencing fish communities in Irish lakes, e.g. availability of food, suitable spawning habitat, water abstraction, inter-specific resource competition (Kelly *et al.*, 2012).

DESCRIPTION OF FISH COMMUNITIES IN EACH STATUS CLASS AND ASSESSMENT OF DRIVER METRICS

Intolerant or disturbance-sensitive fish species (such as brown trout and Arctic char) are normally the dominant fish species in High and Good status lakes. Nutrient-enriched lakes (Moderate and Poor/Bad) are characterised by a higher biomass of tolerant fish species (e.g. roach, perch) than intolerant fish species. Analysis also showed that in general, intolerant fish species decreased and tolerant fish species increased in relation to decreasing ecological status (Olin *et al.*, 2014). The project identified that mean total BPUE is one of the primary fish metrics influencing fish status in Irish lakes. There was a continuous increase in total BPUE in relation to decreasing ecological status, and statistical analysis revealed that total BPUE was significantly different between the High–Good boundary and the Good–Moderate boundary (Fig. 2).

FISH IN RIVERS CLASSIFICATION TOOL: FCS2-IRELAND

SAMPLING FISH IN RIVERS

To date, quantitative depletion electrofishing has been the method of choice for WFD fish monitoring in rivers in Ireland. Sampling is in accordance with the European standard for fish stock assessment in wadeable rivers (CEN, 2003). Surveys are conducted between July and September when rivers flows are moderate to low. Sampling areas are isolated using stop nets where possible. In small wadeable channels (> 0.5–0.7m depth), bank-based electrofishing equipment is used to sample in an upstream direction, whereas in deeper channels fishing is carried out from a flat-bottomed boat in a downstream direction. A representative sample of all habitats (i.e. riffle, glide and pool) is surveyed (Kelly *et al.*, 2014).

RELATIONSHIP OF FISH IN RIVERS TO PRESSURES

An improved understanding of how fish behave in relation to changes in water quality was achieved

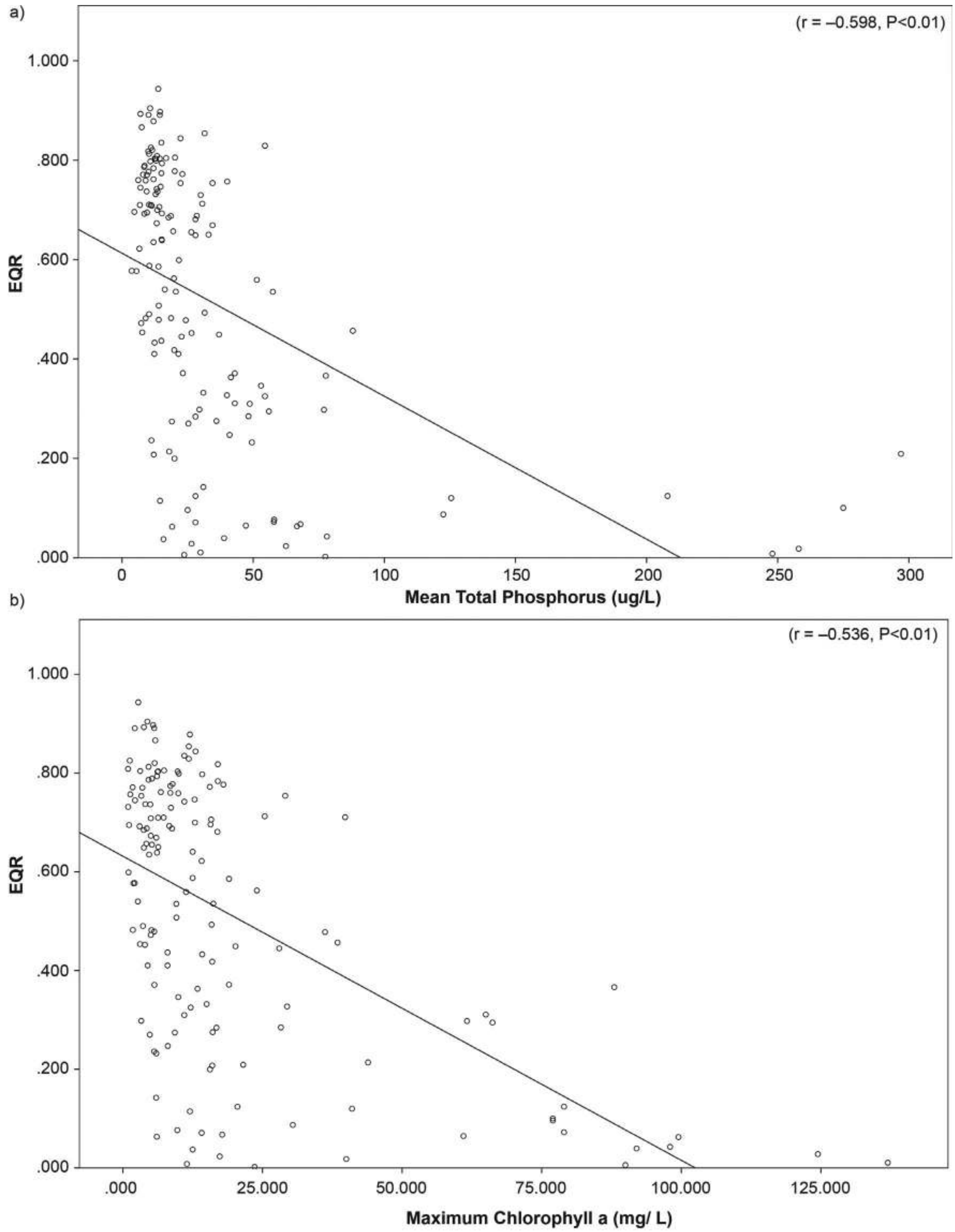

Fig. 1—FIL2 ecological quality ratio (EQR) scores vs (a) total phosphorus (mean) and (b) chlorophyll a (maximum) in Irish lakes (after Olin *et al.*, 2014).

during an EPA-funded project (Kelly *et al.*, 2007). This study found that in general, fish in Irish rivers follow similar trends to those observed in lakes; for example, species richness in rivers increases from zero fish species at Q1 (Bad status) to a maximum diversity at Q3–4 (Moderate status) and a slight decrease at Q5 (High status). Species richness also varies with altitude (Kelly *et al.*, 2007). These researchers found that water quality had a significant impact on the percentage composition and

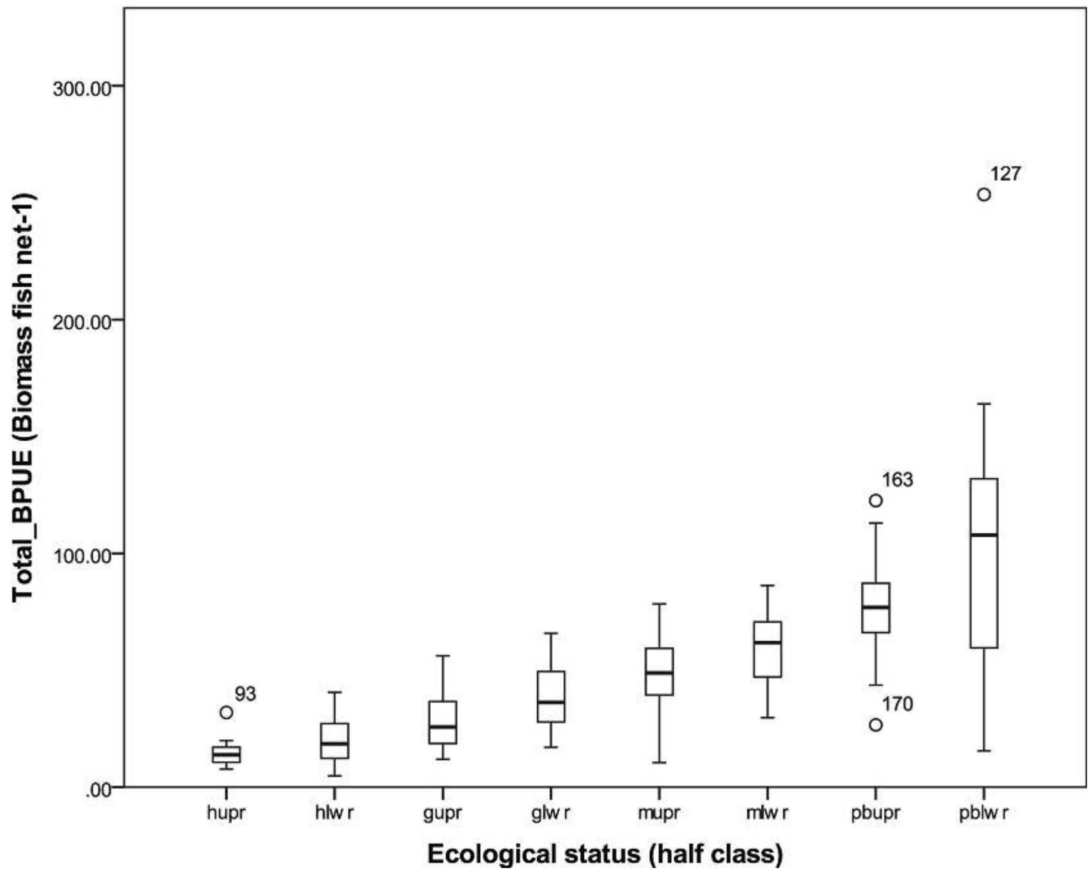

Fig. 2—Box and whisker plot of fish biomass (TOTAL_BPUE: all fish species) vs ecological status (as indicated by half class status boundaries) in Irish lakes (*N* = 176).

abundance of the fish community; three-spined stickleback, *Gasterosteus aculeatus* Linnaeus, 1758 (a tolerant fish species) was one of the dominant species at the more polluted sites (e.g. Q1 to Q2–3), whereas salmonids (intolerant) were the dominant fish species at sites ranging from Q3 to Q5 (Kelly *et al.*, 2007). Two salmonid metrics, i.e. percentage composition of total salmonid and percentage composition of salmonids (1+ years and older) were identified as the best indicators of water quality.

Many studies have reported similar changes in river and lake fish community assemblages from sensitive species to more tolerant species in relation to a decrease in water quality (Eklöv *et al.*, 1999; Mehner *et al.*, 2005). Changes in fish community in rivers due to pressures is thought to be caused by a combination of effects such as a decrease in dissolved oxygen, siltation of spawning gravels and changes in morphology (e.g. Lee *et al.*, 1991). However, where tolerant species, such as cyprinids, have not been introduced or naturally colonised, as is the case in many Irish river channels, a clear distinction is less likely to be evident (Kelly *et al.*, 2007).

DESCRIPTION OF THE RIVERS TOOL (FCS2-IRELAND)

The Environment Agency's (England and Wales) ecological classification tool for fish in rivers, Fisheries Classification Scheme 2 (FCS2) (SNIFFER, 2008; Wyatt, 2007a,b), was adapted for the needs of Ireland and Scotland's WFD fish in rivers programmes following a Proof of Concept project funded by the EPA, Northern Ireland Environment Agency (NIEA) and Scottish Environmental Protection Agency (SEPA) (SNIFFER, 2011).

FCS2-Ireland is a Bayesian statistical model which classifies the ecological status of fish in rivers based on observed catch data (SNIFFER, 2011). A database was compiled by the relevant fisheries agencies north and south: IFI in the Republic of Ireland (RoI) and AFBI in Northern Ireland (NI). This included fish and associated biologically relevant abiotic data from 934 rivers sites (or 981 individual fish surveys—873 from RoI and 108 from NI) across the island of Ireland. Water quality data and sampling method information were also included. The tool works by comparing fish community metric values within a site (observed) to those predicted for a site (expected) under reference conditions using a geostatistical model based on Bayesian probabilities. Multiple and single pass data can be used in the model. The abundance and prevalence for seventeen individual fish models (including two age classes for salmon and trout, i.e. 0+ and 1+ and older) are calculated within the tool based on relationships between a selection of environmental, geographical and pressure variables (sixteen covariates: barriers downstream indicator, hydrometric area number (spatial term), fishing method (boat indicator), geology class (siliceous indicator), salmon stocking, trout stocking, catchment landuse, altitude, wetted width, distance to sea, distance from source, slope, mean depth, alkalinity, conductivity, soluble reactive phosphorus/molybdate reactive phosphorus) (SNIFFER, 2011). The resulting models can then be classified as Bad, Poor, Moderate, Good or High status as per the requirements of the WFD. To determine class boundaries, an artificial dataset was created with the expected fish counts from each survey under that class using the following rules (SNIFFER, 2011) based on the WFD normative definitions for fish (Annex V) (European Parliament and Council, 2000).

1. For *High* status sites, assume all expected species are present in addition to all of the type-specific disturbance-sensitive species (salmon and trout for NI/RoI). Also assume that all age classes are present (e.g. 0+ and 1+ and older salmon and trout). For each of these species, the total catch is set to the expected total catch (i.e. all expected species present at a site).
2. For *Good* status sites, assume 80% of the expected species are present in addition to at least one of the type-specific disturbance-sensitive species. For each selected species, the catch equals 80% of the expected total catch.
3. For *Moderate*, assume 55% of expected species are present (the most likely) and for these the catch equals 55% of the expected total catch.
4. For *Poor*, assume 30% of expected species are present (the most likely) and for these the catch equals 30% of the expected total catch.
5. For *Bad*, assume 10% of expected species are present (the most likely) and for these the catch equals 10% of the expected total catch.

For each of the artificial datasets, the joint EQR variables were calculated for each survey. This gave a spread of joint EQR values that may be expected under High status and similarly under Good, Moderate, Poor and Bad status. The plots of the artificial dataset were then used to compare the distribution of the mean EQR values for each class and this allowed developers to select class boundaries.

The outputs generated by the tool are:

- a single species EQR (seventeen in total; for one species for one survey at one site, two age classes for both salmon and trout are treated as separate fish species models) with associated confidence intervals (represented as probabilities);
- a single joint EQR for each survey (for all species combined for one survey at one site) with associated confidence intervals (represented as probabilities);
- a combined waterbody EQR (combines multiple surveys within a water body) with associated confidence intervals (represented as probabilities).

A comparison of model results with expert opinion identified that the statistical model provided a close match to the class that a fisheries expert would allocate to a water body (almost 85% were within one class of the expert opinion) (SNIFFER, 2011). There were, however, some differences between the model classification and the expert classification for selected sites (e.g. very low conductivity sites, sites upstream of natural barriers and sites with high numbers of tolerant species (such as stone loach, *Barbatula barbatula* Linnaeus, 1758 and three-spine stickleback) and small survey sites with low numbers of fish).

It is important therefore to recognise that a number of limitations of the model must be considered when using it and interpreting the outputs. The core of the FCS2 approach is the statistical model used to predict the fish fauna under reference conditions. Key to success of this model are adequate data when fitting to identify relationships between fish counts and other known covariates (SNIFFER, 2011). Although the database includes 981 fish surveys across the island of Ireland, there are still some areas with insufficient coverage of sites (e.g. sites with very low conductivity and high-altitude sites). Another flaw is that the barrier database for Ireland is not yet complete and the tool can downgrade a site upstream of a natural impassable barrier if it is expecting salmon or eels when they are not present at a site. Another issue which occurs from time to time is that EQR values are built under the assumption that higher numbers of observed fish indicate Good or High water quality, represented by a larger EQR: this is not always the case, as some species may be undesirable in high numbers (e.g. high numbers of three-spined stickleback or stone loach can be indicative of a pressure) (SNIFFER, 2011; Kelly *et al.*, 2007). Therefore, all outputs are regularly sense checked by fisheries experts prior to final reporting. It is also recommended that the models be refitted when more data (fish, abiotic and specifically a national database of barriers) are available in the future.

RELATIONSHIP OF FISH AND EQR TO PRESSURES

The relationship between the individual fish metrics (models) and abiotic variables (including pressures) was investigated during the development of FCS2 Ireland; for example, there was a positive correlation between roach and phosphate (as indicated by molybdate reactive phosphate) (SNIFFER, 2011) and there was a negative correlation between 0+ salmon and 1+ and older salmon in relation to connectivity (SNIFFER, 2011). There was also a significant relationship between fish metrics and various abiotic variables (covariates) such as mean wetted width of the river channel, alkalinity, mean depth, distance to sea, slope (SNIFFER, 2011).

The most important pressures considered by fisheries scientists in a cross-Europe WFD fish in rivers classification tool intercalibration exercise were water quality alteration, hydromorphological modifications and connectivity disruption. The response of FCS2 Ireland EQRs in relation to two water quality indices (common water quality index and water quality alteration pressure) was investigated in the intercalibration exercise and found to be significant. The relationship between a common pressure index developed for intercalibration from seventeen individual pressure variables (Table 1) and FCS2-Ireland EQRs was also examined. Although it was not possible to detect a statistically significant relationship between the common pressure index and FCS2-Ireland EQRs due to the absence of sites in the Moderate and Poor/Bad categories, analysis revealed that there was a negative response to the pressure index between High and Good status.

DESCRIPTION OF FISH COMMUNITIES IN EACH ECOLOGICAL STATUS CLASS

Intolerant fish species (brown trout and salmon) were found to be the dominant species at Moderate through to High ecological status sites (Fig. 3). Nutrient enriched/organically polluted Moderate quality sites, particularly the lower half of the Moderate status class, were characterised by a higher abundance of tolerant fish species such as three-spined stickleback and no or very occasionally low numbers of salmonids (Fig. 3). Fish were more or less absent from Poor or Bad quality sites (Fig. 3). The most significant fish community change occurred at the Good–Moderate boundary, where there was a change from intolerant fish species (salmon and trout) dominance to tolerant species dominance (mainly three-spined stickleback and cyprinids such as minnow roach) in the lower half of the Moderate status class (Fig. 3), with a loss of intolerant species at Poor and Bad sites (Fig. 3).

Table 1—Individual pressure variables used to calculate a common pressure index for river fish intercalibration.

Variable	*Explanation*
P_barrierup	Artificial barriers upstream from the site
P_barrierdown	Artificial barriers downstream from the site
P_impoundment	Impoundment
P_hydropeaking	Hydropeaking
P_waterabsrt	Water abstraction
P_reservoir	Colinear connected reservoir (fish farms, fish ponds, etc.)
P_dam	Upstream dams influence
P_watertemp	Water temperature modification (excuding dam effect)
P_chan	Channelisation/Cross-section alteration (segment scale)
P_vegrip	Riparian vegetation
P_habalt	Local habitat alteration (site scale)
P_dyke	Dykes (flood protection)
P_tox	Toxic risk. Priority substances list
P_waterac	Water acidification
P_waterqualindex	National water quality index (segment scale)
P_wateralt	Water quality alteration (local scale)
P_navigation	Navigation

FISH IN TRANSITIONAL WATERS: CLASSIFICATION TOOLS

SAMPLING FISH IN TRANSITIONAL WATERS

Fish monitoring of transitional waters in Ireland follows a standard multimethod approach designed for the implementation of the WFD (Harrison and Kelly, 2013). This includes the use of a seine net (30m × 2m—14mm mesh with a 5m long × 6.5mm central panel), double fyke nets (2 × 0.55m diameter front hoops × 8m leader; three nets combined to form one fishing unit) and a beam trawl (1.5m wide × 0.5m high; net body 3m × 10/14mm mesh with a 1m long × 5/6.5mm mesh cod end). Seine netting is conducted in shallow littoral areas at low tide, while the fyke nets are set for 24h in deeper waters. Trawling is conducted in mid-channel areas and towed at a speed of 1–2 knots for a set distance of 100m. Sampling effort varies according to estuary size but is undertaken to

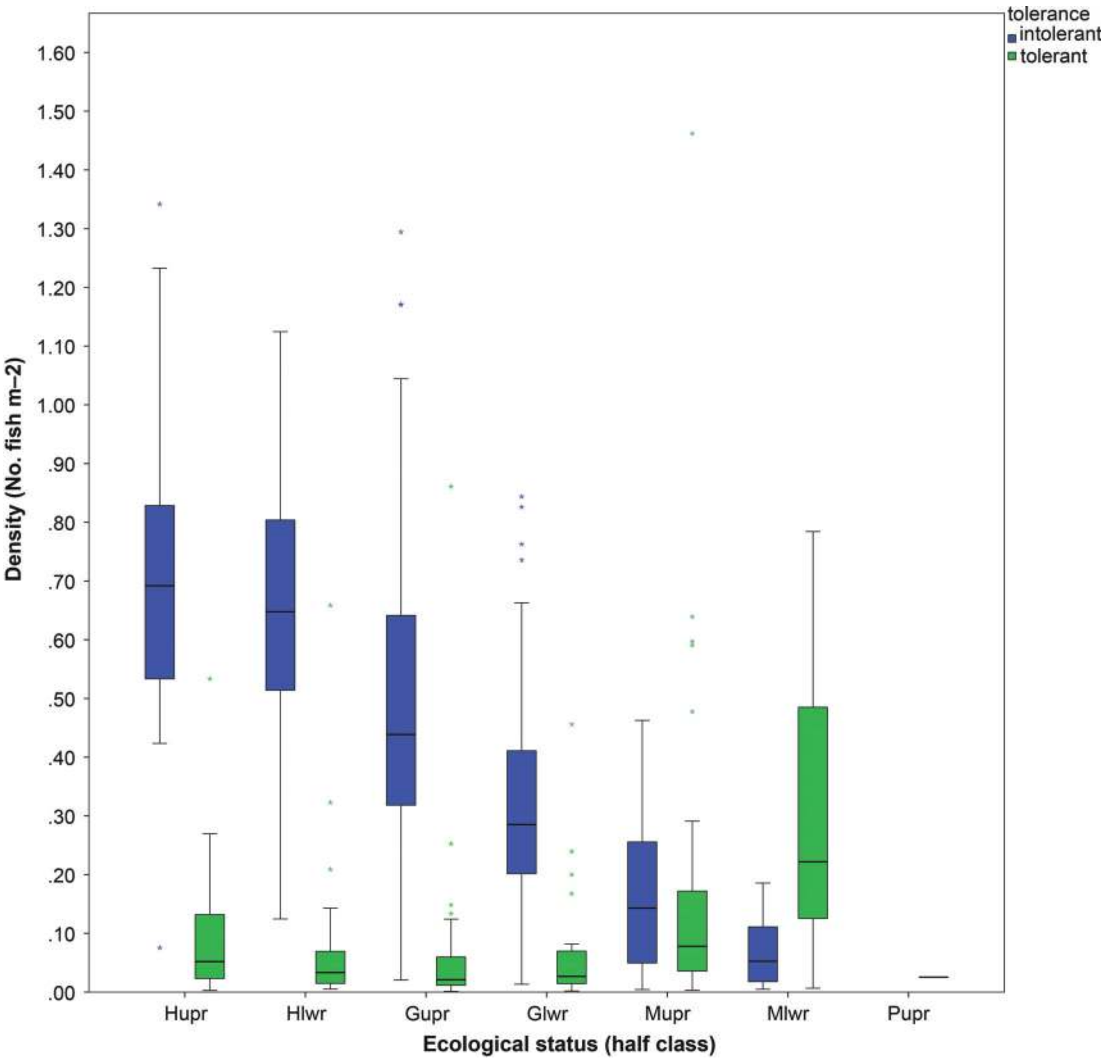

Fig. 3—Minimum density (no. fish m^{-2}) of intolerant (blue) vs. tolerant fish (green) species in Irish rivers in relation to ecological status (as indicated by half class status boundaries) ($N = 206$). (The box represents the interquartile range, the dark horizontal line represents the median and the whiskers represent the minimum and maximum, stars represent outliers and extreme values.)

ensure an adequate spatial coverage of each system (Harrison and Kelly, 2013).

DESCRIPTION OF THE FISH CLASSIFICATION TOOLS FOR TRANSITIONAL WATERS

Initially the monitoring agencies in Ireland (Department of Agriculture, Environment and Rural Affairs (NI) (DAERA) and IFI (RoI)) used a modified version of the Transitional Fish Classification Index (TFCI) (Coates *et al.*, 2007) to assign fish status in transitional water bodies. Subsequently data generated from an extensive monitoring programme (2005 to 2010) conducted across the island of Ireland by IFI and DAERA were used to develop a modified version of the TFCI for Ireland (TFCI-Irl). The tool uses a multimetric approach broadly based on that developed for estuarine habitats in South Africa and the United Kingdom, with a total of ten metrics (species composition, presence of indicator species, species relative abundance, number of taxa that make up 90% of the abundance, number of estuarine resident taxa, number of estuarine-dependent marine taxa, functional guild composition, number of benthic invertebrate feeding taxa, number of piscivorous taxa, feeding guild composition) used in the index calculation (Harrison and Whitfield, 2004; Coates *et al.*, 2007). The TFCI-Irl was successfully intercalibrated in a Europe-wide exercise (European Commission (EC), 2013). In 2013, a second project was initiated by DAERA in collaboration with IFI to develop an improved classification tool known as the Estuarine Multi-metric Fish Index (EMFI) (Harrison and Kelly, 2013). The assessment concept of the EMFI also followed a multimetric approach, which has been successfully applied to transitional waters both globally and in Europe in the context of the WFD (e.g. Miller *et al.*, 1988; Deegan *et al.*, 1997; USEPA, 2000; Hughes *et al.*, 2002; Borja *et al.*, 2004; Harrison and Whitfield, 2004; 2006; Breine *et al.*, 2007; Coates *et al.*, 2007; Delpech *et al.*, 2010; Jordan *et al.*, 2010; Cabral *et al.*, 2012). The EMFI consists of a balanced and complementary set of 14 metrics that include both qualitative and quantitative measures representing four broad fish community attributes: species diversity and composition, species abundance, estuarine utilisation and trophic composition (Harrison and Kelly, 2013). The metrics included in the EMFI not only meet the WFD requirement of measures of species composition, abundance and disturbance-sensitive taxa, but also include functional elements of the fish community (estuarine utilisation and trophic composition). The metrics were selected based on their ecological relevance, ease of measurement, and their ability to meet the requirements of the WFD; overall the EMFI provides a sensitive and integrated measure of the ecological status of fishes in transitional waters (Harrison and Kelly, 2013). The EMFI is applied at the transitional water (whole estuary) level where each metric is allocated a discrete score between 1 and 5 according to the degree of deviation from reference conditions. The EMFI is calculated by summing the scores of each metric and has the range 14–70. The final EMFI values are rescaled to an EQR between 0 and 1, where a value of close to 1 represents High ecological status and values close to 0 represent Bad ecological status (Harrison and Kelly, 2013).

RELATIONSHIP OF THE FISH EQR AND PRESSURE

The response of the EMFI to anthropogenic disturbance was examined using two independent measures of estuarine condition. Article 5 of the WFD requires that EU MSs provide a review of the impact of human activity on the status of surface waters within each river basin district. Various impacts were assessed, including point sources (e.g. waste water treatment plants), morphology (e.g. channelisation, impoundments), water balance (e.g. abstraction), hazardous substances and nutrients. The second measure of estuarine condition included a common pressure index (PI) based on that described by Aubry and Elliott (2006) that was used in the second phase of WFD intercalibration (Table 2). The common PI comprised eight indicators that were classified into three broad categories of disturbance: coastal morphological change, resource use change and environmental quality (Table 2). The results of the Article 5 assessment and the common PI were both converted into a ratio with a range of between 0 (high impact/pressure) and 1 (low impact/pressure). The overall EMFI-EQR exhibited a moderate but significant relationship with both the Article 5 index and the common PI (Fig. 4); EMFI-EQR values increased with a corresponding increase in Article 5 index and common PI values (Harrison and Kelly, 2013).

INTERCALIBRATION OF THE FISH CLASSIFICATION TOOLS

In order to harmonise ecological assessment systems and to ensure a consistent quantification of the level of protection and restoration of surface water bodies across the EU, an intercalibration exercise was launched involving several hundred fisheries experts from the majority of MSs across Europe (Nõges *et al.*, 2009; Poikane *et al.*, 2015). Fifty experts participated in the river fish intercalibration exercise, 31 in the lakes and 26 in the transitional waters. In total, 230 methods from 28 countries have been intercalibrated and published in the EC (2013) decision. The intercalibration exercise aimed to ensure that the High–Good (H/G) and Good–Moderate (G/M) boundaries in all MSs' assessment methods for all biological quality

Table 2—Metrics used in the common pressure index for transitional water intercalibration (from Aubry and Elliott, 2006)

Metric		*Description*
Coastal morphological change	1	Intertidal area lost; realignment schemes; land claim; gross change in bathymetry and topography
	2	Interference with the hydrographical regime
Resource use change	3	Anthropogenically affected coastline by human activity
Environmental quality	4	Water chemical quality
	5	Water quality biological effects
	6	Benthos
	7	Dissolved oxygen (% saturation) (temporal)
	8	Dissolved oxygen (spatial)

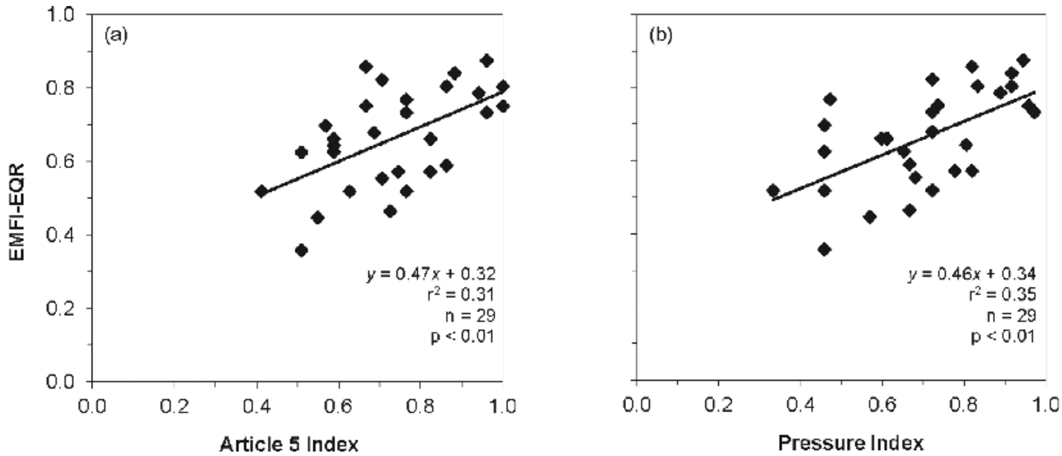

Fig. 4—Relationship between the EMFI and (a) Article 5 index and (b) common pressure index (after Harrison and Kelly, 2013).

elements correspond to comparable levels of ecosystem alteration (EC, 2011). This exercise led to the development of innovative new approaches to accomplishing this complex task (Birk *et al.*, 2013; Poikane *et al.*, 2015). Large Geographical Intercalibration Group (GIG) databases were created for the intercalibration exercises, for example fish data for approximately 4515 river sites from 24 MSs and data for 1300 lakes were pooled from national datasets. These databases also contained basic data such as altitude, surface area, mean depth, alkalinity, water quality data (chlorophyll a, nutrients, Secchi depth) and pressure data (land use, population, other parameters). Data quality was checked by revealing outliers and testing of well-established relationships (Poikane *et al.*, 2015). RoI and NI were members of the Nordic GIG for lakes and rivers (with Finland, Sweden, Scotland, England and Wales, and Norway) and of the North East Atlantic GIG for transitional waters (with Belgium, France, Germany, Portugal, Scotland, Spain and The Netherlands).

In total, eighteen fish in rivers classification tools from 26 countries were intercalibrated; nine fish in transitional water classification tools from eight countries were also successfully intercalibrated, while only five MSs have successfully intercalibrated the lake fish tools to date (a full description of results is provided in EC, 2013). Three options for intercalibration were provided depending on the nature of data acquisition and numerical evaluation (EC, 2011).

- Option 1—direct comparison of classification boundaries, where countries use the same data sampling methods and processing techniques and the same assessment methods.
- Option 2—indirect comparison, where sampling methodologies, data processing and assessment method differ among countries, intercalibration was achieved indirectly through the development of common biological tools into which national methods were converted before being compared.
- Option 3—in cases where data sampling techniques were similar among countries but the assessment methods differed, intercalibration was achieved by applying each assessment method to every national dataset within the GIG and comparing the class boundaries (H-G and G-M) directly (Poikane *et al.*, 2015).

The fish in rivers intercalibration was completed at the end of 2011. Standard sampling methodologies assisted in progressing the completion of the river fish intercalibration before the other two water-body types. Four fish classification tools fulfilled the compliance criteria for intercalibration within the Nordic GIG and gave a similar output, therefore there was no need for boundary adjustment as boundary biases were below 0.25 class equivalents. Intercalibration was undertaken using Options 2 and 3 and the final intercalibrated class boundaries for FCS2-Irl are shown in Table 3.

The intercalibration exercise for fish in lakes proved more difficult than for the rivers as the sampling methodology across MSs was quite different. Intercalibation was undertaken using Option 3 by the Nordic GIG, of which Ireland and Finland were the only countries that could successfully intercalibrate their tools. The Finnish and Irish tools gave on average a very similar output and were comparable without harmonisation when applied to the common intercalibrated lake types, and again there was no need for boundary adjustment as boundary biases were below the required level (Olin *et al.*, 2014). To date, only five countries (Austria, Finland, Germany, Ireland and Italy) have succeeded in intercalibrating their 'fish in lakes' classification tools. The final intercalibrated class boundaries for FIL2 are shown in Table 3.

Table 3—Final intercalibrated class boundaries for four fish classification tools: Fisheries Classification Scheme 2 Ireland (FCS2-Irl), Fish in Lakes 2 (FIL2), Transitional Fish Classification Index Ireland (TFCI-Irl) and Estuarine Multimetric Fish Index (EMFI)

Ecological status	*FCS2-Irl*	*FIL2*	*TFCI-Irl*	*EMFI*
High	0.845 < EQR ≤ 1.000	0.76 < EQR ≤ 1.00	0.86 < EQR ≤ 1.00	0.92 < EQR ≤ 1.00
Good	0.540 < EQR ≤ 0.845	0.53 < EQR ≤ 0.76	0.53 < EQR ≤ 0.86	0.65 < EQR ≤ 0.92
Moderate	0.120 < EQR ≤ 0.540	0.32 < EQR ≤ 0.53	0.40 < EQR ≤ 0.53	0.35 < EQR ≤ 0.62
Poor	0.007 < EQR ≤ 0.120	0.125 < EQR ≤ 0.32	0.20 < EQR ≤ 0.40	0.10 < EQR ≤ 0.35
Bad	0.000 < EQR ≤ 0.007	0.000 < EQR ≤ 0.007	0.00 < EQR ≤ 0.20	0.00 < EQR ≤ 0.10

Intercalibration of various fish methods in rivers and lakes was undertaken indirectly using a common biological metric or index (Option 2) (Harrison *et al*., 2015). However, the fish intercalibration in transitional waters differed from this procedure; rather than a common biological metric or index, a common (abiotic) pressure index was used to compare and intercalibrate the various methods and intercalibration was achieved through the use of pressure–impact relationships (Lepage *et al*., 2016). The final intercalibrated class boundaries for both fish in Irish transitional water classification tools (TFCI-Irl and EMFI) are shown in Table 3.

All fish assessment methods for the three surface water types showed a significant relationship between fish-based status and the pressure index (Kelly *et al*., 2012; Olin *et al*., 2014; Harrison *et al*., 2015). The regressions also met the requirements set out in the intercalibration guidance, i.e. the relationship should be significant ($p \leq 0.05$) and sufficiently strong ($r \geq 0.5$) (EC, 2011).

Intercalibrated 'High–Good' and 'Good–Moderate' class boundary values have been established for the three fish classification tools, lakes-FIL2, rivers-FCS2-Irl and TRAC-TFCI-Irl; these results have been accepted in EC decision 2013/480/EU of 20 September 2013 for rivers and lakes. In addition the EMFI was recently intercalibrated in 2015 in the completed NEAGIG exercise (Harrison *et al*., 2015). The process followed the European Commission, Joint Research Centre procedure to fit new or updated classification methods to the results of a completed intercalibration (Willby *et al*., 2014).

OTHER RESEARCH

There are still gaps in our knowledge with respect to fish and their performance in relation to pressure; however, some of these gaps are currently being filled by ongoing research projects. One project, funded by the Irish Research Council, is focusing on indicator fish species, i.e. the three 'at risk' fish species (pollan, *Coregonus autumnalis* Pallas; Killarney shad, *Alosa fallax killarnensis* Pallas, 1776; and Arctic char) that occur in some Irish lakes. The project began in 2012 and is a collaboration involving IFI and University College Dublin. It aims to use hydroacoustics and various netting techniques to facilitate the monitoring of population trends of these at-risk species and subsequently to identify and quantify the pressures driving these trends for both the WFD and the Habitats Directive (92/43/EEC) as these Directives overlap in their intended outcomes, i.e. improvements in the ecological status of the water bodies in which these fish reside should result in the improved conservation status of each species.

A second research project, funded by the EPA STRIVE programme in 2013, is being undertaken by Trinity College Dublin to fill the knowledge gap for biological assessment methods, including fish, in the tidal freshwater transitional waters (TFTW). Wilson *et al*. (2016) have completed a preliminary assessment of IFI WFD fish monitoring data from selected TFTWs and have proposed a modified version of the EMFI for classification of these water bodies.

There is a major gap in knowledge of the ecological and hydromorphological effects of various types of barriers and potential effects of their removal on connectivity/continuity of fish habitat and other aquatic ecology (of fish, macrophytes and macroinvertebrates) in Ireland. A project has recently been funded by the EPA (Reconnect; initiated January 2016) to assist in filling some of this knowledge gap in relation to barriers in freshwater (J.J. King, pers. comm.). One of the deliverables of the project will be a spatial GPS layer of barrier locations across the country which will then be used to refine the fish in rivers classification tool (FCS2). In addition, IFI is participating in a large Horizon 2020 project led by Swansea University (AMBER—Adaptive Management of Barriers in European Rivers; initiated June 2016), also focusing on barriers. The Munster Blackwater has been chosen as one of the demonstration catchments for this project.

DISCUSSION AND CONCLUSIONS

The use of fish as indicators of environmental quality is well established (e.g. Henriksen *et al.*, 1989; Minns *et al.*, 1994; Belpaire *et al.*, 2000; Degerman *et al.*, 2001; Mehner *et al.*, 2005) but potentially challenging (Kelly *et al.*, 2012), with several problems: (1) a wide variety of sampling methods are used; (2) the activities of fishing, stocking and the introduction of non-native species can have an impact on the natural fish fauna; (3) water bodies can be subjected to multiple pressures and fish, typically secondary, tertiary or higher level consumers, indirectly integrate the effects of these on lower trophic levels; (iv) high natural variability in fish metrics which may be related to natural factors such as lake size, depth and water chemistry (e.g. alkalinity); (5) fish are mobile and can avoid sensitive areas of environmental stress, so they are less sensitive to some pressures than others, but more sensitive to some, for example barriers to connectivity (Poikane *et al.*, 2015).

Fish can be difficult and more expensive to monitor in certain water-body types than other biotic elements such as macroinvertebrates, but they are important indicators (Poikane *et al.*, 2015); they are at the top of the food chain, have significant economic and social importance (for example, recent estimates value angling at €876 million to the Irish economy; NSAD, 2015) and their assessment is an important part of an integrated approach to water management.

It is well established that fish are sensitive indicators of environmental degradation and offer the major advantage of integrating the direct and indirect effects of stress over large scales of space and time (Poikane, 2015). Fish exhibit reactions to eutrophication, habitat destruction, shoreline degradation, lake use intensity, hydromorphological degradation, connectivity, acidification and combined degradation. Nevertheless, prior to the WFD the fish community was often an overlooked and neglected aspect of river, lake and transitional environmental water assessments, with many monitoring programmes concentrating on chemical water quality and macroinvertebrates (Poikane, 2015).

There is quite a distinction between water quality and ecological status, and systems used to establish the former are only a small part of those needed for the latter (Moss *et al.*, 2003). To date in Ireland there has been a lot of emphasis on water quality and maybe not enough emphasis on ecological quality, particularly in relation to River Basin Management Plans (e.g. WRBD, 2009) and Programmes of Measures (ESBI, 2008). Results from the biological quality elements (BQEs) (e.g. fish, invertebrates, macrophytes) are sometimes confusing and contradictory; for example, the various BQEs do not always give the same status output for each water body: in fact in some cases they can be quite different (e.g. two to three class differences) and the reason for this is not always understood by non-experts. During the 2010 to 2012 reporting period fish were the driving BQE in 57/160 (36%) rivers and 20/81 (25%) lakes, and a greater understanding of these differences is now required for river basin planning. Trend analysis and further analysis of the performance of metrics is currently under way to provide some of this information; for example, assessment of multiple pressures such as water quality combined with introduced species (e.g. zebra mussels) which may be impacting on the fish population more than the other biota. Although the text of the WFD does not explicitly mention alien species, it is clear that they may at times constitute a pressure on other aquatic species, particularly fish, as well as detracting from the 'naturalness' of a water body—a fundamental concept that underpins the WFD—therefore a greater understanding of how native fish species interact with non-native or alien species is required.

Despite the challenges described above, Ireland has been successful in developing WFD-compliant ecological classification tools for fish in all three surface water types and in intercalibrating them in a cross-Europe exercise. Conclusions of the work discussed above are that fish can be used as indicators of ecological quality, they are more sensitive to some external pressures than other BQEs, and they are sensitive to multiple factors in many cases which can drive fish status more than one class away from other BQEs.

Notwithstanding the work that has been undertaken, the ecological classification tools should be reviewed from time to time as more data become available (Kelly *et al.*, 2012), as this may provide further insights into the pressure–response relationships with the fish data and improve the outputs of the various tools.

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