At what spatial scale should risk screenings of translocated freshwater fishes be undertaken – river basin district or climo-geographic designation?

ABSTRACT

To inform aquatic conservation policy and management decisions, translocated freshwater fish species, i.e. those native to part but not all of Great Britain (GB), were assessed with the Aquatic Species Invasiveness Screening Kit (AS-ISK) at two spatial levels (River Basin District [RBD] and GB overall), the outcome scores calibrated and analysed to determine the relevance of geographical scale (GB, RBD and freshwater ecoregion) on AS-ISK outcome score rankings. The 16 species assessed received scores that showed limited among-RBD variation, with all but only one species (silver bream *Blicca bjoerkna*) receiving the same risk ranking across all RBDs for which they were assessed. A trend of increasing AS-ISK score with decreasing RBD latitudinal location was observed, with two species (bleak *Alburnus alburnus* and tench *Tinca tinca*) found to have significantly higher AS-ISK scores in west-coast RBDs than in RBDs to the north and east, and one species (bleak *Alburnus alburnus*) to have significantly higher AS-ISK scores in southern RBDs than in northern RBDs. The Water Framework Directive classification of Scotland was found to be inconsistent with the latitudinal gradients in that country's environmental conditions, which are better reflected in the distinction of northern and southern freshwater ecoregions. The ramifications of these legislative classifications for aquatic conservation are discussed.

Keywords: AS-ISK; Aquatic Species Invasiveness Screening Kit; Water Framework Directive; freshwater ecoregion; non-native species, invasive alien species

Running title: Translocated freshwater fish risk screening for Great Britain

1. Introduction

As governments around the globe strengthen their nature conservation policy and legislation to regulate and control non-native species (NNS), especially those that are or likely to become invasive, attention is eventually being directed towards translocated species, which are taxa native to part but not all of a nation state that have been introduced to non-native parts of that entity (Copp et al., 2005). This is of particular importance in the United Kingdom (UK), where de-centralisation of government regulatory processes has taken place. This transfer of administrative and legislative authority to devolved administrations in Scotland, Wales and Northern Ireland requires a transitional process during which the responsible government bodies develop their priorities for the implementation of local legislative regulations and controls. However, regardless of this autonomy and potential need for local regulation, as a Member State (of the European Union) and/or signatory to international agreements, the UK is subject to both international and national (i.e., UK) controls.

To inform these conservation policy and management decisions regarding translocated species, NNS risk analysis provides a means of identifying species that are likely to become invasive where introduced to other parts of a nation state that are outside the species' native distributions. This approach is identical to the evaluation of species that are entirely non-native to the risk assessment (RA) area (Baker et al., 2008), such as has already been done for freshwater fishes with regard to England & Wales (Copp et al. (2009). For the purposes of the present study, the focus was restricted to Great Britain (GB), i.e. England, Scotland, Wales, given that NNS on the island of Ireland are addressed collectively by Invasive Species Ireland (http://invasivespeciesireland.com/).

The identification of future potentially-invasive species is particularly important in cases where species can be easily translocated and introduced into an adjoining RA area (e.g., nation state, drainage basin). Such is the case in GB, where Scotland and Wales are species-poor countries in terms of native freshwater fish fauna relative to southern parts of England

(Wheeler, 1972; Treasurer, 1993; Maitland, 2004), which is the well-known donor region for several introductions of fish species into Scotland (Adams & Maitland, 2002; Maitland, 2007; Adams et al. 2014), to northern England (Winfield et al., 2010), and through water transfer schemes in the East of England (Copp & Wade, 2006). What remains unclear in risk analysis terms is the spatial scale at which such translocations should be assessed within a nation state. A biogeographical and climatic (climo-geographic) perspective is normally recommended (e.g., Copp et al., 2005), and there are several examples of risk screening of NNS for RA areas defined biogeographically (e.g., Ferincz et al., 2016; Glamuzina et al., 2017; Tarkan et al., 2017) or climo-geographically (e.g., Onikura et al., 2012; Puntila et al., 2013).

Combining the biogeographic and climo-geographic approaches is not straight-forward because the delineations of the world according to Köppen-Geiger climate types (Peel et al., 2007; Beck et al., 2018), to freshwater ecoregions (Abell et al., 2008) and to ecoregions of the European Union (EU) under the Water Framework Directive (WFD) (European Union, 2000), are not entirely consistent. For example, in Finland the RA area for a similar risk screening (Puntila et al., 2013) encompassed almost exclusively rivers along the country's southern coastline that discharge into the Baltic Sea. This is generally consistent with Köppen-Geiger climate type Dfb separation of the country's southern and northern catchments, but Finland falls entirely within a single freshwater ecoregion (Northern Baltic drainages) according to Abell et al. (2008). Elsewhere, the RA area in Japan for a risk screening of potentially invasive freshwater fishes (Onikura et al., 2012) was the northern, hydrogeographically separate part of Kyushu Island, which falls mainly into one of three Köppen-Geiger climate types (Cfa, Dfa, Dfb) but only one freshwater ecoregion (643 – Biwa Ko).

A similar conundrum exists for GB, which falls within a single Köppen-Geiger climate type (Cfb), and a single ecoregion under Europe's WFD (European Union, 2000), but comprises two freshwater ecoregions (Abell et al., 2008): '402' (Northern British Isles, which includes Scotland, Wales and island of Ireland [henceforth ‘Ireland’] to the west and north); and ‘404’ (Central and Western Europe of which England represents the most western extent). However, this single WFD ecoregion is sub-divided into twelve River Basin Districts (RBDs): Scotland, Solway & Tweed, Northumbria, North West England, Humber, Anglia, West Wales, Dee, Severn, Thames, South East England, and South West England (European Commission, 2016). A compounding factor is the long history of freshwater fish translocations within GB (e.g., Wheeler, 1972; Maitland, 1987; Winfield et al., 2011), with some of these translocations believed to have negatively impacted native fishes of conservation interest and their communities (e.g., Winfield et al., 2010). As such, GB is a good ‘test subject’ to assess the most appropriate spatial geographic and climatic scales of the RA area for the risk screening/assessment of translocated freshwater fishes.

The aim of the present study was to carry out the first risk screening of translocated freshwater fishes for GB (the RA area) to determine which species are likely to pose a risk of being (or becoming) invasive in those parts of GB where they are not native. The specific objectives were to: 1) compile an up-to-date list of species native to part but not all of GB, comprising both those known to have been translocated within GB and those that could potentially be translocated; 2) assess these species using the Aquatic Species Invasiveness Screening Kit (AS-ISK: Copp et al., 2016b) decision-support tool to obtain outcome invasiveness scores for RA areas at two spatial levels (RBD and GB overall); 3) analyse the outcome scores to calibrate and validate AS-ISK for GB with respect to freshwater fishes; 4) assess the relevance of geographical scale (freshwater ecoregion vs. river basin district) on the risk screening score; and 5) provide recommendations on the regulation of the assessed species in terms of their importation to, and their keeping and release within GB.

2. Material and methods

Three spatial scales within GB were considered in this study. Firstly, RBD as defined under the WFD (European Commission, 2016). Secondly, GB as an entity, whereby the RA area consisted of any part of GB outside the species presumed native distribution (see Table 1). And thirdly, freshwater ecoregion as per Abell et al. (2008), which for GB consists of: 'Northern' British Isles, encompassing the RBDs of Scotland, Solway & Tweed and those of Western Wales and the River Dee; and 'Southern' British Isles, comprising all other RBDs in GB attributed to the 'Central and Western Europe' ecoregion.

The species included in the list of translocated freshwater fishes encompassed: A) all native species that are known to have been introduced from their native distribution range in GB to other parts of GB where the species is not native; and B) any other native species likely to be translocated within GB. Note that in the case of crucian carp *Carassius carassius*, the RA area encompasses all parts of GB because a recent genetic study has demonstrated that this species was most likely introduced about the same time as common carp *Cyprinus carpio*, and therefore is most likely 'not native' to southeast England as was previously believed by some scientists (Jeffries et al., 2017). A similar approach, encompassing both extant and potential future species, has been used in all published applications of AS-ISK on freshwater fishes to date (i.e., Glamuzina et al., 2017; Li et al., 2017; Tarkan et al., 2017) and in most previous applications of FISK (see Copp, 2013), as this provides a means of assessing current species, which may or may not have expressed invasive patterns. It also represents a horizon-scanning function to aid in the identification of possible future invasive species (Copp et al., 2009; Copp, 2013). As such, this approach extends beyond that taken by Kolar & Lodge (2002), who considered only those species already present in the RA area and grouped them as having 'established' and 'not established' self-sustaining populations. Also, unlike that North American risk screening study, the listing of freshwater fishes for the present study is confounded by uncertainty as regards their original native distributions – this uncertainty is despite previous, seminal efforts to define the original species distributions through the compilation of historical records (e.g., Maitland 1972, 1977, 1987, 2004a, 2004b; Wheeler 1972, 1974; Treasurer 1993; Wheeler et al., 2004; Winfield et al., 2010).

For each species in each RBD, a systematic search was undertaken using two main sources of information: 1) the Web of Science, (https://login.webofknowledge.com/), to access peer reviewed publications and scientific abstracts from conferences; and 2) www.google.co.uk and its academic derivative, Google Scholar (https://scholar.google.co.uk/), to access peer reviewed, grey literature and web-based information. Boolean search terms were used to unify the search effort for each question/species combination (see example), and represented the minimum effort required to identify appropriate sources of information. Following the identification of appropriate publications, using the Boolean searches, an assessment of the information contained therein was used to highlight additional sources of information. Two online sources, FishBase (www.fishbase.org; Froese & Pauly, 2018) and the Invasive Species Compendium by CABI (Centre for Agriculture and Biosciences International: www.cabi.org/isc/) were used to access general information regarding known invasiveness risk. General climate information was based on the Köppen–Geiger climate classification system (Peel et al., 2007) and on the freshwater ecoregions defined by Abell et al. (2008). This process provided a means to differentiate between the northern RBDs (Scotland, Solway & Tweed, Western Wales and Dee; www.feow.org/ecoregions/details/northern_british_isles), and southern RBDs (Northwest England, Northumbria, Humber, Anglia, Thames, Southwest England and Southeast England; www.feow.org/ecoregions/details/central_western_europe).

To assess the potential each species poses as a vector for endemic and/or novel pests or infection agents, contemporary parasite information from GB (Brewster, 2016) was compared with the global known parasite fauna for each species available from the Natural History Museum (2018). As parasite information was only available at the GB level, resolution at the

RBD level was not possible. Information from the National Biodiversity Network was used to assess the likelihood of a species entering a protected area. Using the spatial analysis tool (https://spatial.nbnatlas.org/), point records of occurrence for each species were plotted separately and the map overlaid by maps of protected areas: Wetlands of International Importance (RAMSAR), Sites of Special Scientific Interest (SSSI), and Special Area of Conservation (SAC). The extent of each RBD was then visually assessed to look for the association between the point records and the extent of the protected areas. Direct overlaps between point records were taken as very high confidence that the species was in a protected area, this was then adjusted depending on the distance of the point record from a protected area. When occurrence records did not overlap, potential routes (i.e., presence of connected water courses) through which the species could enter a protected area were assessed and the likelihood of a species entering a protected area was assessed.

These information sources were used to screen the translocated fish species using AS-ISK, which is a combination of the architectural framework of FISK v2 (Lawson et al., 2013) and the generic screening module in the European Non-native Species in Aquaculture Risk Analysis Scheme, ENSARS (Copp et al., 2016a). The AS-ISK, which is a third-generation derivative of the Weed Risk Assessment (WRA) of Pheloung et al. (1999), may be applied to any non-native aquatic species, regardless of their aquatic environment (brackish, freshwater, marine) and climatic region.

The AS-ISK is fully compliant with the ‘minimum standards’ (Roy et al., 2018) for assessing species under the new EU Regulation on invasive alien species of EU concern (European Union, 2014). AS-ISK has already been used successfully to screen non-native fishes in at least three risk assessment (RA) areas, including translocated species in: China (Li et al., 2017), Turkey (Tarkan et al., 2017) and a large river catchment in the Balkans

(Glamuzina et al., 2017). A global trial of AS-ISK applications is in progress (L. Vilizzi, G.H. Copp et al., in prep.).

Similar to the FISK, the AS-ISK comprises 49 questions (Qs) to assess the biogeographical and historical traits of the taxon and its biological and ecological interactions. The basic 49 questions are complemented by an additional six questions that ask the assessor to assess how predicted future climate conditions are likely to affect their responses to Qs related to the risks of introduction, establishment, dispersal and impact. For each question, the assessor must provide a response, justification and level of confidence. Once the assessment has been completed (i.e., all 55 Qs answered), the basic risk screening (BRA) score is added to the score from the climate change questions to achieve a composite BRA + Climate Change Assessment (CCA) score (hence, BRA+CCA). The possible values for the BRA score range from −20 to 68, and for the BRA+CCA score from −32 to 80. Finally, the ranked levels of confidence (1 = low, 2 out of 10 chances; 2 = medium, 5 out of 10; 3 = high, 8 out of 10; 4 = very high, 9 out of 10) associated with each question-related response in AS-ISK mirror the confidence rankings recommended by the Intergovernmental Panel on Climate Change (IPCC, 2005; Copp et al., 2016b).

For each species, AS-ISK assessments were first undertaken at the RBD-level and were then compiled to provide a single risk assessment for each translocated species for GB-level assessments. The data compilation process was achieved by identifying which questions had different responses and using the most common response amongst RBD-level assessments as the response for the GB-level assessment for that species. The most common response was used for all questions except for question 36 ("*Will any of these pathways bring the taxon in close proximity to one or more protected areas (e.g. MCA, MPA, SSSI)?*" as it was felt the consequences of the introduction of a non-native to a single protected area within GB would have significant implications at a national level (e.g. legal obligations of maintaining protected areas). The assessments were carried out by the first author, who is familiar with the species being assessed, and then peer-reviewed by the other co-authors CB and GHC, both being freshwater fish biologists familiar with fishes of the RA area.

In the score data analysis, the number of translocated freshwater fish species for GB (n = 16) was insufficient for successful calibration of the dataset. Therefore, the calibrated FISK threshold score (i.e., 19), which was established by Copp et al. (2009) to distinguish between high risk from low-to-medium risk NN fishes for the UK, was used as the 'starting point' for categorisation of the translocated species. Given the changes in the 49 BRA Qs in AS-ISK relative to FISK (Copp et al., 2016b), it was not possible to 'transfer' directly the above threshold value to AS-ISK, so an 'estimated' threshold was computed. This was based on the two available AS-ISK applications that have assessed the same group of fish species for a certain RA area also under FISK, namely those by Tarkan et al. (2017) and by Glamuzina et al. (2017). In the former study, the AS-ISK (BRA) threshold of 27.75 was 4.75 units higher relative to the corresponding FISK threshold of 23; whereas, in the latter study (with a caveat for some additional species assessed in that application of AS-ISK), the AS-ISK (BRA) threshold of 10 was 0.25 units lower than to the corresponding FISK threshold of 23. The UK FISK threshold of 19 was therefore incremented by the mean value of 2.25 based on the two score differences above, leading to a (rounded) AS-ISK BRA threshold of 21 that will be used in the present study to distinguish between medium and high-risk species. To estimate the BRA+CCA threshold (hence, distinguish between medium- and high-risk translocated species for the BRA+CCA assessment), the only AS-ISK application on freshwater fishes providing both thresholds (namely, Glamuzina et al., 2017) identified a BRA+CCA threshold of 12.62, hence 2.62 units higher than the BRA threshold of 10. The AS-ISK BRA threshold was, therefore, incremented by this difference leading to a (rounded) BRA+CCA threshold of 24.

Notably, although based on limited information (i.e., only two studies), this approach is in line with Bayesian adaptive management practice (Hilborn & Mangel, 1997; Prato, 2005).

Based on the confidence level (CL) allocated to each response for a given species (see *Risk screening*), an overall confidence factor (CF_{Total}) was computed as:

$$\sum(CL_{Qi})/(4 \times 55) \ (i = 1, \ldots, 55)$$

where CL_{Qi} is the confidence level (CL) for Question *i* (Q*i*), 4 is the maximum achievable value for certainty (i.e., 'very certain') and 55 is the total number of questions comprising the AS-ISK. The CF_{Total} ranges from a minimum of 0.25 (i.e., all 55 questions with certainty score equal to 1) to a maximum of 1 (i.e., all 55 questions with confidence level equal to 4). Two additional confidence factors were also computed separately for the BRA and CCA questions, namely the CF_{BRA} (based on the 49 BRA Qs) and the CF_{CCA} (based on the six CCA Qs).

To examine the effect of the geographical scale (freshwater ecoregion vs. RBD) on the risk screenings, the mean AS-ISK score for each species was subtracted from the mean AS-ISK score for each RBD. This standardised score provides a measure of the deviation of the species score from the mean and thus a measure that is comparable across all fish species.

The standardised AS-ISK score was regressed against freshwater ecoregion ('Northern' and 'Southern', as defined here above) and river basin district location (Fig. 1) in two separate linear mixed-effects models, including fish species as a random effect to account for pseudo-replication. Model significance is reported as the significance of the deviance explained compared with the null model. Additionally, for species that demonstrated the greatest variation among RBDs, these were examined to identify any geographical patterns (e.g., north vs. south), grouped accordingly and compared using the Students' unpaired *t*-test.

3. Results

In total, 16 translocated fish species were risk screened using AS-ISK across the twelve RBDs (Fig. 1), with *Carassius carassius* the only species assessed for all of them, and spined loach *Cobitis taenia* and roach *Rutilus rutilus* both assessed for one RBD only (Table 1; the AS-ISK report for each RBD assessment is available in the downloadable Supplementary Information data file). Outcomes for all species were consistent across RBDs except for one species (Table 2), namely silver bream *Blicca bjoerkna*, which was attributed scores of both medium and high risk for both BRA and BRA+CCA. All other species categorised as medium or high risk in all RBDs for which they were assessed and for both the BRA and the BRA+CCA. The only species for which the AS-ISK risk ranking differed between BRA and BRA+CCA was Arctic charr *Salvelinus alpinus*, which dropped from high (BRA) to medium (BRA+CCA) risk consistently across all RBDs for which it was assessed (Table 3). Species-specific mean AS-ISK scores showed relatively limited among RBD variation (SE bars in Fig. 2), the greatest being observed with bleak *Alburnus alburnus* and tench *Tinca tinca*. In the case of *T. tinca*, and with a caveat for small sample size, a trend of increasing AS-ISK score with decreasing RBD latitudinal location was observed, whereby AS-ISK scores were significantly higher (Students' $t = 5.422$, df = 3, $P < 0.02$) in west-coast RBDs (mean for Dee, Severn and West Wales = 31.3, SE = 0.833) than in RBDs to the north and east (mean for Scotland and Solway & Tweed = 25.5, SE = 0). For *A. alburnus,* there appears to be a significantly higher risk ($t = 2.729$, df = 6, $P < 0.04$) posed in southern RBDs (mean for Southeast, Southwest and Severn = 29.0, SE = 0) than those in the north (mean for Solway & Tweed, Dee, Northwest, Northumbria, and West Wales = 26.1, SE = 1.782).

Overall, responses to the 55 Qs across RBDs were very similar, with only Q4 (*How similar are the climatic conditions of the RA area and the taxon's native range?*) and Q36 (*Will any of these pathways bring the taxon in close proximity to one or more protected areas (e.g., MCZ, MPA, SSSI)?*) carrying a 'Medium' or 'High' and a "Yes" or "No" response, respectively.

At the GB level, based on the RBD-level assessments, seven (43.8%) were categorised as medium risk and nine (56.2%) as high risk, and this applied to both the BRA and the BRA+CCA scores (Table 3). Ruffe *Gymnocephalus cernuus* and *T. tinca*, common bream *Abramis brama* and *Alburnus alburnus* achieved the highest scores (≥ 29 for the BRA; ≥ 31 for the BRA+CCA) and were followed by chub *Squalius cephalus*, *Rutilus rutilus*, rudd *Scardinius erythrophthalmus* and *Blicca bjoerkna*; on the other hand, *Salvelinus alpinus* was categorised as high risk for the BRA but medium risk for the BRA+CCA. This was due to the −2 score for the CCA component of the risk screening, which was at variance with all other scores of either 2 or 4 that incremented the corresponding BRA score (Table 2). Amongst the species categorised as medium risk, grayling *Thymallus thymallus* and *Cobitis taenia* achieved the lowest scores, even though none of the species assessed was categorised as low risk (i.e., score <1).

Mean confidence level for all Qs (CL_{Total}) was 2.74 ± 0.04 SE, for the BRA Qs (CL_{BRA}) 2.85 ± 0.05 SE, and for the CCA Qs (CL_{CCA}) 1.89 ± 0.03 SE, hence within the ‘high’ category overall and for the BRA but within the ‘medium’ category for the CCA. Similarly, the mean values for CF_{Total} = 0.69 ± 0.01 SE and CF_{BRA} = 0.71 ± 0.01 SE were higher than the mean value for the CF_{CCA} = 0.47 ± 0.01 SE. In all cases, the narrow standard errors indicated overall similarity in CLs and CFs across the species assessed.

With regard to geographical assessment scale, the standardised risk score for translocated species in the Southern ecoregion was significantly higher ($\chi^2_{(1)} = 32.24$, $P < 0.0001$) than for the Northern ecoregion (Fig. 3). The standardised risk score was also significantly related ($\chi^2_{(1)} = 10.21$, $P = 0.001$) to a general north-west to south-east geographical gradient (Fig. 4).

4. Discussion

The rationale for conducting risk screening at both RBD and GB scales in the present study is apparent for some species but not others. For example, risk screenings may be necessary at a relatively small geographic scale for a few species, e.g. *Blicca bjoerkna*, which was the only species to be attributed different risk rankings (either medium or high) across the RBDs for which it was assessed (Table 2). The variation in AS-ISK scores for several species (and risk rankings for *B. bjoerkna*) could be attributed to variations in the response to Q4, reflecting differences in climate between the taxon's native range and the RA area. Species with a more restricted native range are more likely to show such variation. And in the case of *B. bjoerkna*, the 2–3 point increase in score was enough to elevate this species over the threshold for different risk categorisation. With the species showing the greatest among-RBD variation in AS-ISK score (Fig. 2), i.e., *Tinca tinca* and *Alburnus alburnus*, there was a consistent pattern of higher score for *T. tinca* in southern RBDs (Western Wales, Dee, Severn) than in northern RBDs (Scotland, Solway & Tweed; Table 2). This contrasted *A. alburnus* for which there was no discernable latitudinal or longitudinal trend.

In GB, fresh waters to the north are significantly more species-poor than those to the south, thus risk screening at a national or RBD level has the potential to mask biogeographical differences, resulting in a measure of risk which may be appropriate for one part of the nation and not the other. In the case of the RBD 'Scotland', climate and aquatic habitat vary from north to south and west to east, which is recognised in the freshwater ecoregions of Abell et al. (2008) for the north–south gradient, but not for the east–west gradient, given that Scotland and Wales comprise the same freshwater ecoregion ('Northern' British Isles'). That said, and as mentioned above, there appears to be a greater risk posed by *T. tinca* in western RBDs of GB than in other RBDs for which the species was assessed (Table 2). As such, the fact that Scotland is classified as comprising a single RBD is very unhelpful from a regulatory perspective. Indeed, there could be variations in the risk rankings of some species among river catchments within the RBD Scotland (e.g., those more northerly vs. those in the south of Scotland), which were not revealed in the present, RBD-level study. Indeed, some of the most important conservation risks are likely to be site-specific. For example, the translocation of fish to water bodies of conservation interest (e.g., containing locally-important species or natural fish communities, or naturally lacking a fish fauna) could have a greater conservation impact than translocation into an adjacent water body of lesser conservation value. That said, the pattern of increasing deviation in standardised AS-ISK scores (Fig. 4) suggests that the risks of translocated fishes being invasive are higher in southern RBDs than in the northern RBDs, in part due to increased likelihood of establishment due to climate compatibility, which may change in the future (Britton et al., 2010).

Overall, the use of RBDs as the RA Area for risk screenings appears to work well enough when the RBD is effectively a geographically-defined area (e.g., drainage basin), e.g. rivers Thames and Dee. However, this may not be appropriate in areas where risk needs to be assessed at a finer geographical scale. Scotland is a good example of a composite RBD, encompassing several drainage basins across a latitudinal cline within a single RBD, where assessment at the RBD level may limit the powers of the main regulatory body (the Scottish Environment Protection Agency) to take appropriate restorative action. So, whilst species such as *R. rutilus*, northern pike *Esox lucius*, Eurasian perch *Perca fluviatilis*, European minnow *Phoxinus phoxinus* and stone loach *Barbatula barbatula* are considered to native to this RBD as a whole, they are native to only certain drainages within the RBD. The translocation of locally non-native, but still nationally native, species such as these to new water bodies can lead to the permanent loss or damage of native biota, particularly fish. The power of WFD legislation to restore fish communities to those that reflect ‘good’ reference conditions is greatly weakened when the RBD is so large that it fails to identify that species may be native to the RBD in general but not native, and damaging, to individual water bodies of the RBD. For example, the widespread distribution of *Phoxinus phoxinus* to water bodies throughout Scotland (e.g. Maitland, 2007) as food or bait for native brown trout *Salmo trutta* may have exerted adverse consequences for populations of that native species (e.g., Borgstrøm et al., 2010). As such, the WFD River Basin Plan may not identify the need for control or removal of *Phoxinus phoxinus* as a priority because they are 'native' to the RBD that covers all of Scotland. The same applies to introduced *Esox lucius*, *Perca fluviatilis*, *Rutilus rutilus* and *Barbatula barbatula*, which may either predate native species or compete with them for limited resources during part or all of those species' life cycles.

Assessing risk at the RBD scale may not allow risk to be properly assessed in parts of that RBD where these 'native' species are in fact non-native, and possibly invasive. In view of the potential variation in risk score (though not necessarily risk ranking) screening should take place at a scale that is appropriate to answer the conservation management question being asked. As this geographic scale gets smaller, from RBD to hydrometric area to individual catchment level, for example, so too does the quality and quantity of data required to support any assessment, including evidence of which species are native and which are not. Failure to identify risk at smaller geographical scales may also result in the loss of opportunities for control or removal. This, in turn, could lead to further spread of species identified as potentially posing a high risk of being invasive in previously un-invaded or connected water bodies. This may lead to a downgrading of waterbody status (*sensu* WFD), and the application of further pressure on regulators to initiate restorative action. This data-quality issue is particularly relevant in countries with a long history of non-native fish introductions, such as Germany, France, Italy and the United Kingdom (Copp et al., 2005).

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Figure captions

Fig. 1. Location of the 12 River Basin Districts (RBDs) of Great Britain (as per European Union, 2000), numerically ordered from north-west to south-east (1 = Scotland, 2 = Solway & Tweed, 3 = North West, 4 = Northumbria, 5 = Humber, 6 = Western Wales, 7 = Dee, 8 = Severn, 9 = Anglian, 10 = Thames, 11 = South West, 12 = South East). Northern freshwater ecoregions (after Abell et al. 2008) are shaded grey, southern are white. Three river basin districts straddle the freshwater ecoregion divide and have been ascribed to the ecoregion in which the largest area of the river basin falls: Solway & Tweed attributed to the 402th ecoregion (Northern British Isles), with Northumbria and Severn attributed to the 404th ecoregion (Central and Western Europe). The information used to generate this map follow conditions for data use specified under Open Government Licence with all rights reserved (©Environment Agency 2015; ©Natural Resources Wales.) for the RBDs, and at www.feow.org/copyright (©The Nature Conservancy and World Wildlife Fund 2008, Inc. All Rights Reserved) for the freshwater ecoregions.

Fig. 2. Mean and standard error of AS-ISK scores (basic risk assessment [BRA] and climate change assessment [CCA] calculated from Table 2) for freshwater fish species across all RBDs for which they were assessed using the Aquatic Species Invasiveness Screening Kit (AS-ISK). Species codes are: Ct = *Cobitis taenia*, Tm = *Thymallus thymallus*, Bb = *Barbus barbus*, Cg = *Cottus gobio*, Ll = *Leuciscus leuciscus*, Cr = *Carassius carassius*, Gg = *Gobio gobio*, Bj = *Blicca bjoerkna*, Se = *Scardinius erythrophthalmus*, Rr = *Rutilus rutilus*, Sa = *Salvelinus alpinus*, Sc = *Squalius cephalus*, Aa = *Alburnus alburnus*, Tt = *Tinca tinca*, Gc = *Gymnocephalus cernuus*, Ab = *Abramis brama*.

Fig. 3. Standardised AS-ISK scores (deviate of the mean AS-ISK score for each species from the mean AS-ISK score for each RBD) for RBDs in the north (grey bars) and south eco-region (open bars).

Fig. 4. Linear relationship between standardised risk score and the geographical location of the river basin district (see Fig. 1). Low numbers are RBDs located in the north-west and high numbers are RBDs located in the south-east.