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Spatial Variability in Juvenile Sea Trout Data Collection and Assessment Methods Across Europe: Limitations and Opportunities for Standardising Analyses

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ABSTRACT

Brown trout (*Salmo trutta* L.) is a widely distributed fish species native to Europe, with high phenotypic plasticity, including resident and anadromous (sea trout) forms. Many populations are in decline, especially the anadromous ecotype towards the southern edge of their range. Evidence-based management is dependent on reliable assessment methods to characterise under-performing populations and identify mitigation actions. Assessment methods based on juvenile data are useful for trout given the species' plasticity. These typically involve comparisons between observed and expected juvenile densities, which vary geographically and with habitat quality. These estimates are mostly obtained through wading electrofishing and field-based habitat surveys. Although some national and regional trout assessment methods have been developed, efforts to develop range-wide evaluations have been constrained by a lack of common protocols. This paper summarises the national methods used to characterise juvenile trout and associated habitat based on information compiled by national representatives of the ICES WGTRUTTA. It then considers opportunities for harmonising data with the aim of developing a range-wide assessment. Survey methods varied substantially within and between countries but showed no clear spatial patterns of coherence. Development of a single set of survey and data collection methods appears highly unlikely. A more pragmatic approach could involve harmonising existing data to make them broadly comparable. This could involve selective use of fish survey data, integration of range-wide macro-scale spatial and habitat data obtained from spatial datasets and use of appropriate modelling frameworks. We also emphasise the critical importance of population data from index rivers to validate and scale juvenile assessments and to compare sampling methods.

1 | Introduction

Sea trout, the anadromous form of brown trout (*Salmo trutta* L.), is present throughout a significant portion of Europe, from Norway, in the north, to Portugal, in the south. Throughout its range, sea trout is considered a valuable resource, from ecological, economic and cultural perspectives (Blicharska and Rönnbäck 2018; Liu, Bailey, and Davidsen 2019). However, in recent decades, it has become increasingly apparent that populations in many countries are in decline (e.g., Jutila et al. 2006; Mota, Rochard, and Antunes 2016; Dębowski 2018; Adams et al. 2022) and that anthropogenic pressures are on the rise (Thorstad et al. 2015; Birnie-Gauvin et al. 2017; Dębowski 2018). Faced with these challenges, local, regional and national fisheries managers are placing an increasing emphasis on developing robust quantitative evidence to assess the status of trout populations, determine population trends and characterise the pressures (negative impacts) acting upon them.

Trout population assessments are conducted at a wide range of spatial scales including national, regional, catchment or site levels, depending on the associated legislative and management drivers. However, effective attempts to harmonise assessment methods across the whole species range have failed, despite the considerable efforts to standardise and intercalibrate assessments under the Water Framework Directive (WFD) (Allan et al. 2006; Champ, Kelly, and King 2009; Hering et al. 2010). The reasons underlying these failures are numerous but strongly reflect the presence of existing sampling networks and protocols that provided the historical data used to develop new WFD assessment tools (e.g., Allan et al. 2006; Champ, Kelly, and King 2009; Hering et al. 2010). Consequently, WFD approaches tended to focus on standardising the outputs of assessment models rather than the underlying data collection methods, in particular the ecological quality ratios (EQR) associated with WFD class boundaries for the commonly assessed biological quality elements (e.g., fish fauna, macroinvertebrates, macrophytes). In addition, WFD requirements for fish assessment were relatively broad, requiring consideration of species composition, abundance and age structure in a wide range of rivers, including those that cannot be sampled by fully quantitative methods. As a result, these needs were often met using relatively simple semi-quantitative sampling methods that provided data on relative abundance at a coarse level required for a broad classification of status. However, these types of data are often not sufficiently precise or free from bias to provide robust assessments of the abundance, status and trends of single species such as trout. The failure to agree common standards has maintained differences in data collection and analytical methods that constrain interregional data compilation and comparison, knowledge exchange and the development of effective assessments that would provide a wider understanding of the ecology and status of trout across its native range. Although the focus of this paper is on a single species, namely, brown trout, the heterogeneity of sampling methods, and need for calibration and standardisation, is a pressing and broader challenge in the scientific literature, affecting regional and international meta-analyses that aim to compile and compare fish data from different countries, collected with different methods (e.g., Comte et al. 2021).

The distribution and abundance of salmonids in freshwater are strongly influenced by habitat quality (Heggenes, Baliniére, and Cunjak 1999; Armstrong et al. 2003), with requirements varying by life stage (e.g., migration, spawning, nursery, rearing and overwintering). The habitat factors influencing abundance can include both abiotic (water quality, cover availability, substrate, hydraulic conditions) and biotic factors (competition, predation pressure, food availability) (Frissell et al. 1986; Heggenes and Wollebæk 2013) and have been hypothesised to be particularly influential when the standing stock is close to carrying capacity or habitat saturation (Armstrong et al. 2003). The physical and chemical processes that control the spatial variability of salmonid habitat quality and abundance during their freshwater phase operate over a range of nested spatial scales (Figure 1) (see also Armstrong et al. 2003). At the largest spatial scales, climate, geology and soil type will influence runoff, flow regime, stream power, river temperature and hydrochemistry (Frissell et al. 1986). These factors influence sedimentary characteristics, channel morphology and productivity, including food availability. At finer scales, hydrology interacts with channel morphology and substrate to generate patch-scale variability in depths and velocities, invertebrate drift rates and densities. At medium-to-large spatial scales, landscape variables obtained from spatial data can explain a substantial proportion of the spatial variance in fish abundance (e.g., Wyatt 2005; Pedersen et al. 2017; Malcolm, Millidine, Glover, et al. 2019). At finer scales, extensive literature reviews show that important correlates include channel morphology, habitat structure, weighted usable area, water velocity, water depth, substrate size, shade and water temperature (Frissell et al. 1986; Armstrong et al. 2003; Fausch, Hawkes, and Parsons 1988; Heggenes and Wollebæk 2013). It is thus clear that habitat characterisation at different spatial scales can be valuable for understanding and supporting assessment of salmonid populations, including sea trout (ICES 2011).

Since the fundamental habitat requirements of trout are expected to be broadly similar across the species' range, developing approaches for obtaining juvenile density and habitat data, based on a common denominator descriptor set that can be compared, could present opportunities to develop new and improved assessments for this species. Wading electrofishing surveys are by far the most used sampling method for estimating juvenile salmonid densities in a wide range of environments and considered the standard for national juvenile trout assessments. Other methods, such as visual surveys, may be used for other purposes and specific research projects but are not commonly applied in support of national assessments.

Analyses of habitat–abundance relationships at larger spatial scales have the potential to draw on larger, more complex and robust datasets that also cover the full range of environments experienced by the species. This offers greater opportunities for understanding the key environmental controls on the species, while also enabling predictions of the likely outcomes of changing climate, land use and management

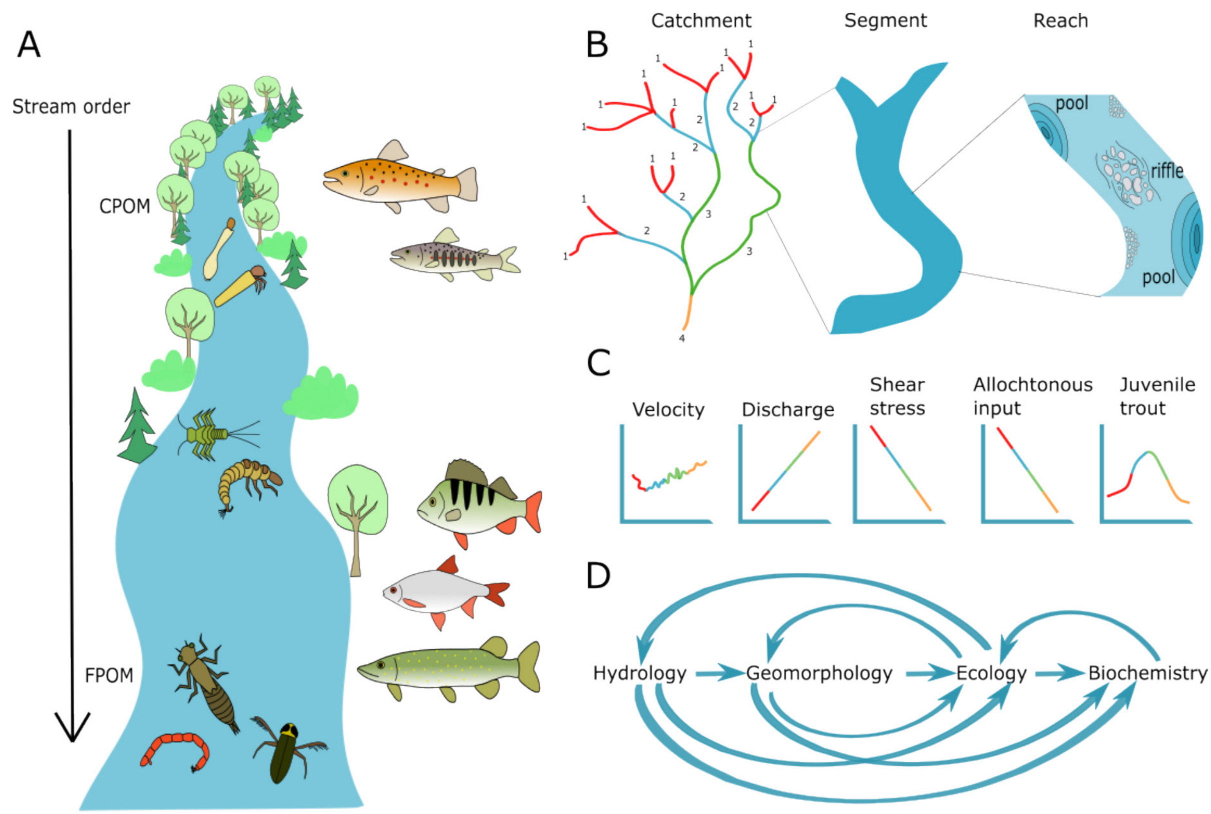

FIGURE 1 | Schematic representation of physical/hydrological and biochemical processes that interact with biological and ecological processes at different, but interconnected, spatial scales and stream orders (Panels B–D) in lotic ecosystems. Panel (A) illustrates the river continuum concept (RCC, adapted from Vannote et al. 1980), a model classifying flowing water based on the concept in which a stream forms a balance between physical parameters (width, depth, velocity and sediment load) while incorporating biological factors and predicting the occurrence of various organisms. In Panel (A): CPOM—coarse particulate organic matter; FPOM—fine particulate organic matter.

interventions. Furthermore, from a management perspective, working across the whole species range provides opportunities for improved assessments and larger scale perspectives on the status of fish populations through common frameworks. Consequently, there is an urgent need to better understand the spatial variability in field data collection methods and consider approaches that allow calibrated intercomparisons and collation between datasets.

In 2017, building on previous science and knowledge exchange workshops (e.g., ICES 2011, 2017), ICES established the WGTRUTTA working group to develop and test assessment methods for sea trout populations and to provide scientific, technical and management outputs that can be used across countries to provide an overall perspective on the status of sea trout. WGTRUTTA includes members from all European countries within the species' native range and is ideally placed to consider such understanding and calibrations. In the case of juvenile assessment methods and the data they require, a prerequisite to such work is to determine the availability and comparability of data relating to trout densities and habitat across contributing countries, which could support the development of intercalibrated habitat–abundance models and, thus, optimised and comparable assessment methods.

Given the predominance of wading electrofishing as the main nationally applied method for estimating juvenile trout densities, and in the absence of an internationally applied standard operating procedure, the aim of this paper is to characterise and compare variability in methods used to measure juvenile brown trout densities and the habitats that support them across the species' native range. Subsequently, strategies and directions are proposed that would allow collation of calibrated and comparable data with a view to developing international assessments. Specifically, we used the WGTRUTTA forum to ask participants that are representatives of the respective national management agencies from all sea trout countries to provide summary details of existing national programmes used to measure (i) juvenile trout densities and (ii) habitat characteristics. We then (iii) discuss the comparability of these datasets and (iv) propose potential solutions to calibrate and harmonise data for compilation to allow the development of an integrated and standardised range-wide assessment of the species, while also making recommendations to harmonise future data collection with a view to improving future assessment methods.

2 | Methods

A template was distributed to WGTRUTTA members to capture information on the national methods used to characterise trout abundance using wading electrofishing. These surveys included those used for the general assessment of fish populations in addition to those used specifically to inform ICES assessments on the species, which are currently done only for the Baltic Sea region. Importantly, the survey deliberately excluded methods used only for local management or specific research objectives where data are often more variable, as well as data from boat-based electrofishing, which is less common, particularly any surveys providing quantitative estimates of abundance, and not readily comparable with wading methods. The data collection template (Data S1) was sent to national representatives from all 19 sea trout countries (Data S1) involved in WGTRUTTA. We collated an inventory of methods from 21 responses. In most cases, a single response was provided by each country. However, Norway and Ireland provided two responses each, reflecting distinctly different national approaches that were commonly applied for the same purpose. The pair of returns for Ireland and Norway was obtained from single representatives and reflected genuinely different sampling methods rather than alternative and differing interpretations of the same method.

The inventory of methods was processed in two matrices; one focused on methods used to measure juvenile trout densities, and the other on methods to characterise their associated habitats. The first was a 21 row × 6 column matrix, containing variables describing methods to sample juvenile trout density: (i) Q1.Sampling method; (ii) Q2.Sampling unit; (iii) Q3.Number of electrofishing passes; (iv) Q4.Use of stop nets; (v) Q5.Targeted species; and (vi) Q6.Sampling season (standardly defined as Autumn: October–December; Winter: January–March; Spring: April–June; Summer: July–September). The second was a 21 × 6 matrix, containing variables describing the methods used to characterise the following habitat variables: (i) Q1.Current velocity; (ii) Q2.Depth; (iii) Q3.Substrate; (iv) Q4.Presence of vegetation/debris; (v) Q5.Shade; and (vi) Q6.Slope, specifically, whether countries monitored slope in the field as a component of monitoring and assessments programmes. These variables were represented as categorical data on an ordinal scale (0–4) for each variable, reflecting an increasing complexity of applied methods, from 0 (e.g., nonrecorded variable) to 4 (e.g., use of standardised/technical methodologies or multiple methods) (see Data S1 for the association between categories and respective ordinal scale values used in following analyses). Principal component analysis (PCA) was used to represent and help visualise multivariate similarities and differences in the national methods to measure juvenile densities and characterise their habitats. The results of PCA analyses were visualised by plotting the position of countries in relation to the first four axes to represent the full range of existent variability. Countries with similar sampling methods plot closely together, while those with differing methods plot further apart. We also plotted vectors for each variable to identify variables that were most influential in defining differences and similarities within the ordination space (variables with Spearman's correlation with PCA axis higher than 0.50). The PCA ordination was computed using the statistical package PRIMER+v6.0 (Anderson, Gorley, and Clarke 2008).

Spatial variability in the methods differentiating between countries was subsequently mapped across Europe. All maps were produced in R (R Core Team 2023) using the package Eurostat (Lahti et al. 2017, 2023) for base mapping data.

3 | Results

There were substantial differences in the methodologies used to measure juvenile trout densities using electrofishing and to characterise habitats between the 19 countries that provided data (see Data S1 for a detailed and graphical description of the frequency distribution of the answers provided for each variable). PC1 and PC2 jointly explained 66% of the variation, while PC3 and PC4 explained 22%. Taken together, the first four components explained ca. 89% of the variance (Figure 2). Q1.Sampling method, Q2.Sampling unit and Q4.Stop nets were strongly associated with PC1 and PC2 while Q3.N° of electrofishing passes and Q6.Sampling season were strongly associated with PC3 and PC4. When considering PC1-PC2 ordination for electrofishing sampling variables (Figure 2a), the first axis primarily reflected differences in Q4.Stop nets (PCA1 = −0.862; PCA2 = 0.322). This separated countries that use stop nets from those that do not (i.e., ordinated on the positive upper side of the first axis). Two of these variables, Q1.Sampling method (PCA1 = −0.237; PCA2 = −0.575) and Q2.Sampling unit (PCA1 = −0.254; PCA2 = −0.687), strongly contributed to the separation of countries on the second axis, which was interpreted as a measure of sampling protocol, separating countries that sample the whole river width and are non-selective of sampling unit (on the negative side of the second axis)

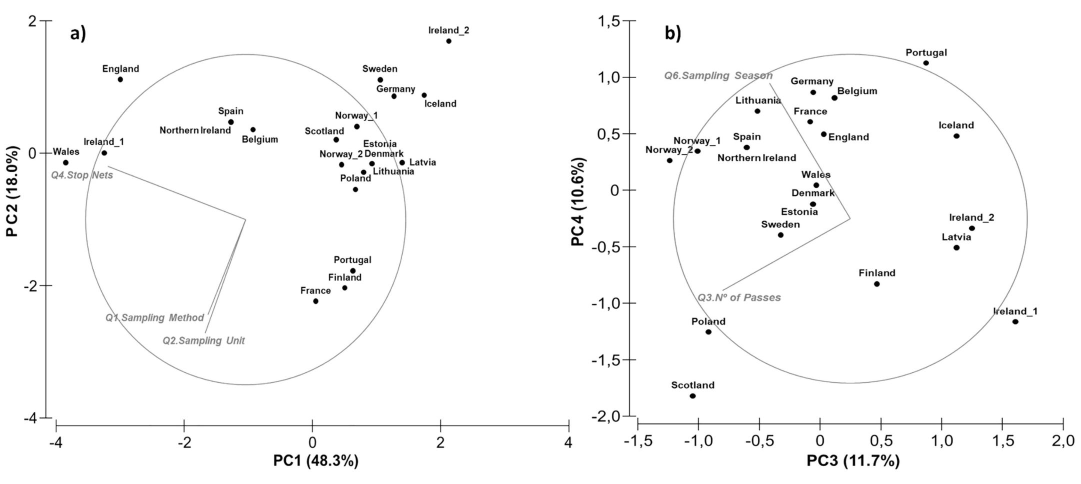

FIGURE 2 | PCA multivariate plot representing the distribution and ordination of the 19 sea trout countries included in the assessment (a total of 21 answers) according to their answers regarding implemented standard methods for collecting data on juvenile trout densities. Figure includes both PC1-PC2 (a) and PC3-PC4 (b) spatial ordinations to show full range of variability within our sample.

from countries that sample only part of the channel and select specific habitats (upper positive side of this second axis). When considering the PC3-PCA ordination of juvenile sampling methods (Figure 2b), two variables were identified as highly related with resultant spatial ordination of considered countries, namely, Q3.N° of electrofishing passes (PCA3 = −0.724; PCA4 = −0.438) and Q6.Sampling season (PCA3 = −0.457; PC4 = 0.823). Q3 was mostly responsible for the country's ordination along the third PCA axis, separating countries that usually apply less quantitative methods with a lower number of electrofishing passes (on the right, positive side of PCA3) from countries with a more complex and quantitative approach, with a higher number of passes (left, negative side of PCA3). Q4 was highly related with a country's ordination on PCA4, explaining the spatial segregation between countries that sample more in autumn, summer or in these two seasons (upper positive side of PCA4) from countries sampling trout with electrofishing in spring or winter (negative, lower side of PCA4).

For habitat characterisation methods (Figure 3), PC1 and PC2 jointly explained 69.8% of the variation, while PC3 and PC4 explained 25%. Taken together, the first four components explained ca. 94.8% of the variance. Five of the six variables contributed significantly to the observed regional differences in trout habitat characterisation. These were Q1.Current velocity, and Q4.Presence of vegetation & debris, in PC1-PC2, and Q2.Depth, Q5.Shade and Q6.Slope, in PC3-PC4 ordination.

When considering PC1-PC2 ordination for habitat characterisation variables (Figure 3a), both Q1.Current velocity (PCA1 = 0.654; PCA2 = 0.666) and Q4.Presence of vegetation & debris (PCA1 = −0.656; PCA2 = 0.710) were strongly related with the two axes, separating countries that use standardised/technical methods to collect data on velocity and vegetation/debris (e.g., graduated poles and aerial photography, respectively, with or without direct observation) on the right and upper positive sides of the PCA, from those located in the left side of the PCA, who usually rely solely on more qualitative visual assessments (i.e., direct observation) to estimate these two variables. When considering PC3-PC4 ordination (Figure 3b), three variables were identified as highly related with the ordination of countries, namely, Q2.Depth (PCA3 = −0.016; PCA4 = −0.755), Q5.Shade (PCA3 = −0.840; PC4 = 0.256) and Q6.Slope (PCA3 = −0.444; PCA4 = −0.559). Q2 and Q6 were responsible for the country's ordination along the fourth PCA axis, separating countries that use standardised/technical methods to collect data on depth (e.g., graduated poles, with or without direct observation), and that have data on slope, located in the lower negative side of PCA4, from those plotting towards the positive side of the axis, which usually rely solely on direct qualitative visual assessments (i.e., direct observation) and do not have slope data for their sampling sites. Q5.Shade was the habitat variable mostly contributing for a country's ordination along PCA3, separating countries that favour the use of more standardised/technical methods to collect data on shade (e.g., densiometers, with or without direct observation), mostly located on the left negative side of PCA3, from countries relying only on direct observation to assess this variable, ordinated on the right positive side of this axis.

Detailed PCA scores for the different variables and countries are presented in Data S1. Spatial variability in juvenile sampling and habitat characterisation methods is mapped in Figures 4 and 5, respectively. These maps suggest that there is substantial variability in the consistency with which methods are applied between countries, for example, only a single country used granulometry to characterise substrate size (Q3. Substrate), while there was more spatial variability in the application of methods used to characterise in-stream velocities (Q1. Current Velocity). Nevertheless, there did not appear to be clear spatial patterns to suggest regional coherence in the application of methods, for example, neighbouring countries each applying more directly comparable methods.

4 | Discussion and Future Directions

4.1 | Variability in Assessment Methods Across Europe

Given that the focus of this study was on a single species of fish (brown trout) and the application of wading electrofishing methods for national monitoring, we may have expected to see a relatively uniform suite of methods to collect data on juvenile

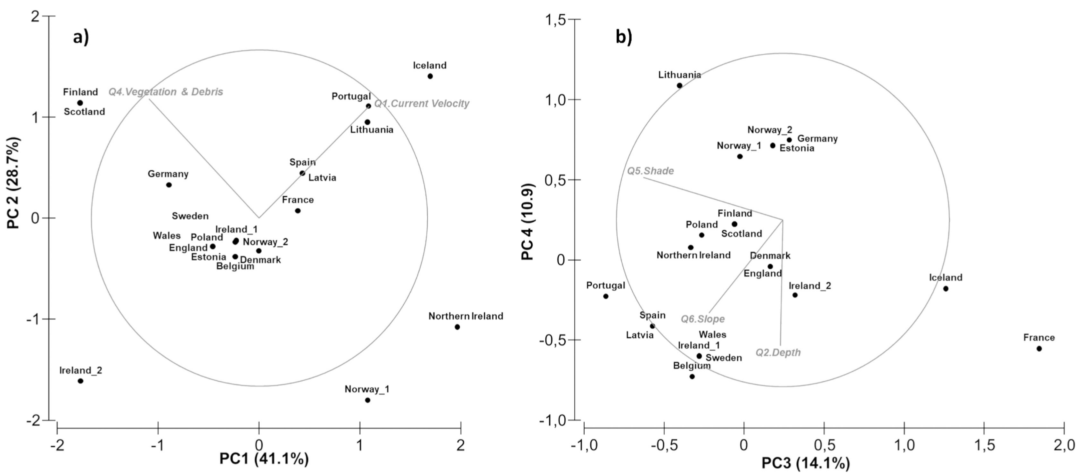

FIGURE 3 | PCA multivariate plot representing the distribution and ordination of the 19 sea trout countries included in the assessment (a total of 21 answers) according to their answers regarding implemented standard methods for characterising juvenile trout habitat. Figure includes both PC1-PC2 (a) and PC3-PC4 (b) spatial ordinations to show full range of variability within our sample.

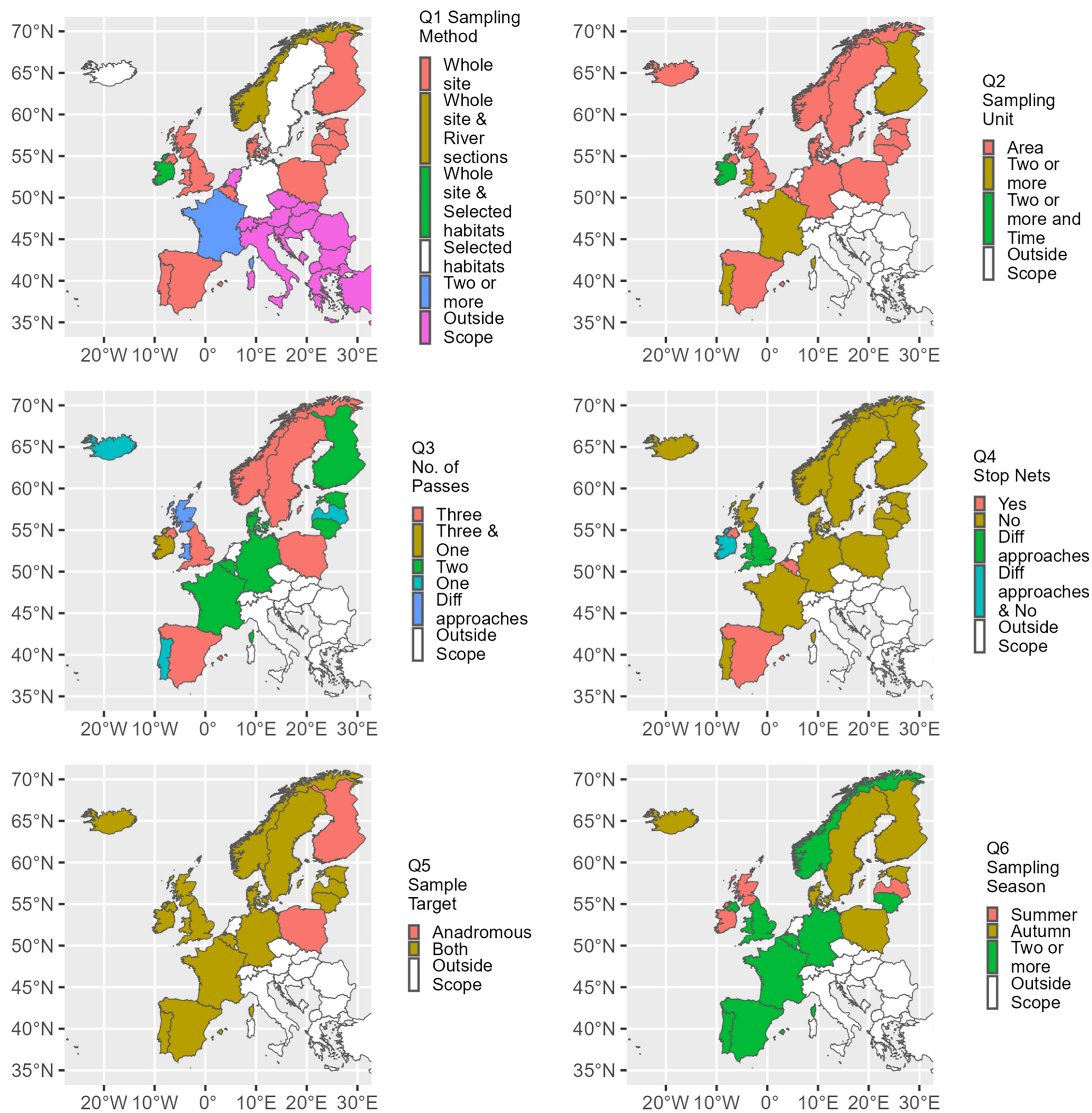

FIGURE 4 | Maps showing spatial variability in the electrofishing sampling methods among countries. Countries marked as outside scope were not relevant to the current analysis. Where more than one method was applied, this is reflected in the categorisation, including countries with two different questionnaires (i.e., Ireland and Norway), as well as specific response values: (i) 'Two or more'—countries apply two or more of the described methods in national trout sampling programmes for the same objective; (ii) 'Different approaches'—countries indicated the use of multiple approaches to characterise this variable, depending on the assessment specific objective.

densities and the habitats that support them. However, analysis of data provided by national representatives from the 19 sea trout countries contributing to WGTRUTTA suggests that there are major interregional differences in sampling methods. To some extent this may reflect the use of electrofishing data from national programmes with multiple objectives, rather than purely trout assessment. Nevertheless, at this stage, it is not possible to directly compare data or status assessments between countries without further careful work. This poses a significant constraint on international comparisons, which are increasingly necessary to understand the ecology and management of trout in a rapidly changing world. While some countries applied carefully standardised, specific and constant methods and techniques, others applied much less standardisation, allowing data to be collected using a variety of methods that are ultimately used for the same purpose. Such differences were observed not only between but also within countries as revealed by the findings from Norway and Ireland, both of which use two contrasting data collection methods for measuring juvenile densities and characterising habitats in their national monitoring programmes (Figures 2 to 5).

The information gathered to support this manuscript was deliberately focussed on national sampling programmes. Our aim was not to characterise all the different potential survey approaches used to characterise trout abundance in different scenarios, as this would have required a more formal and widespread survey design, neither was our aim to evaluate only those national sampling programmes used specifically to inform ICES trout assessments (conducted in the Baltic

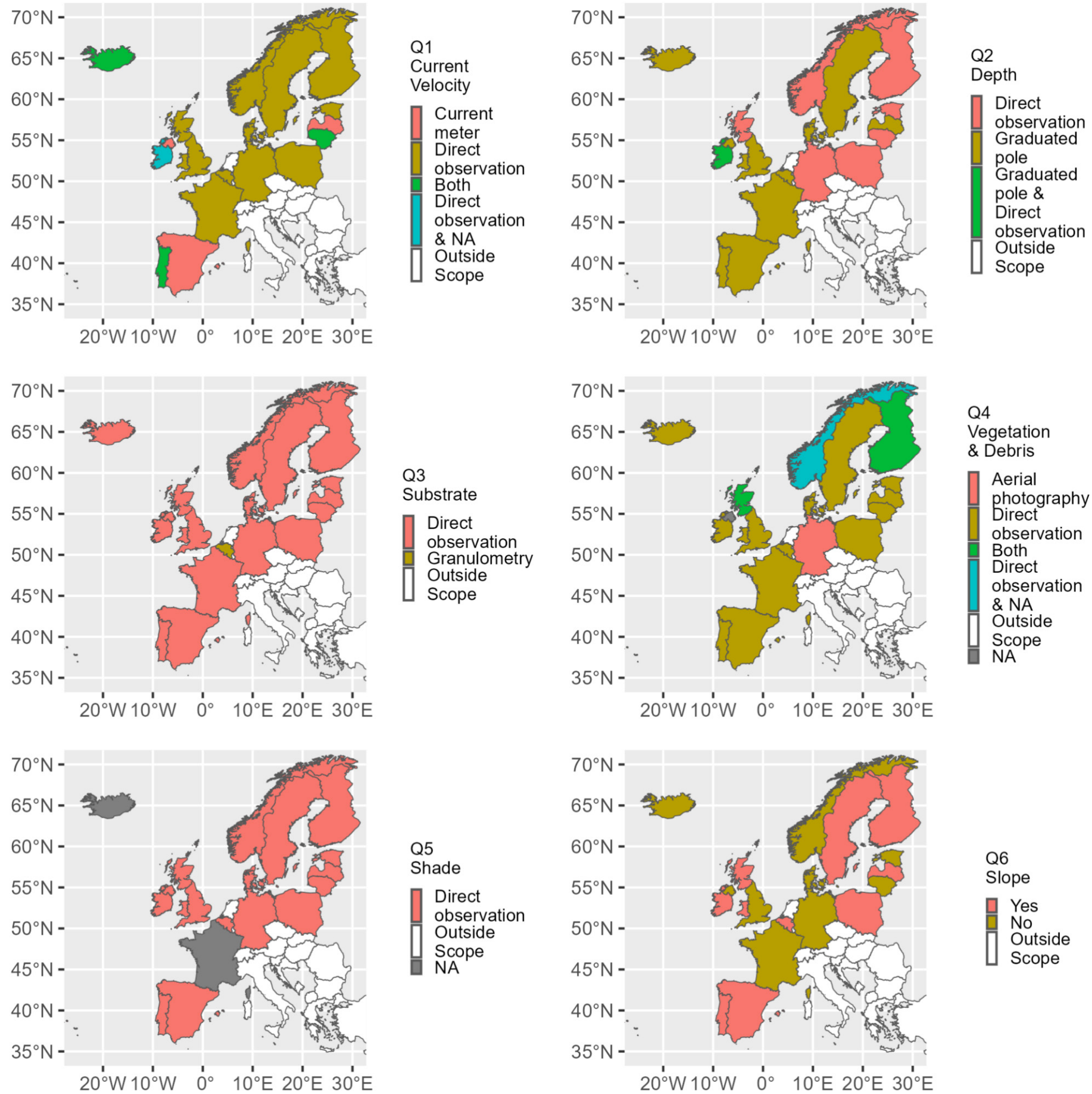

FIGURE 5 | Maps showing spatial variability in habitat characterisation methods among countries. Where more than one method was applied, this is reflected in the categorisation, including countries with two different questionnaires (i.e., Ireland and Norway), as well as specific response values: (i) 'Two or more'—countries apply two or more of the described methods in national trout sampling programmes for the same objective; (ii) 'Different approaches'—countries indicated the use of multiple approaches to characterise this variable, depending on the assessment specific objective. In the case of slope measurements, the answers refer only to field data collection, rather than other GIS derived approaches.

Sea region). Instead, we gathered information from the 19 national representatives of each of the countries contributing to WGTRUTTA and asked them to characterise the national methods that collected data on trout and could be used for assessment purposes. As national representatives, these scientists are responsible for implementing these national programmes in their countries, and we are thus confident that they provide a comprehensive description of relevant national juvenile trout sampling programmes.

One of the main drivers of methodological differences between countries is likely related to the original purposes of the data collection. For many countries, Atlantic salmon rather than trout is the primary target species during surveys. In other cases, surveys aim for a general inventory of fishes (e.g., relative abundance) at local/regional scales rather than accurate characterisation of trout abundance. Furthermore, despite previous attempts to intercalibrate assessment methods, fisheries managers have tended to favour well known and established methods and analyses within their countries. In some circumstances, these may also be driven by practical and financial considerations. In general, fisheries and environmental organisations are unwilling to start new monitoring programmes, even if they would better align data with other countries, when this would come at a cost to existing national time series. Thus, instead of proposing changes to historical assessment programmes, fisheries managers have often chosen national data coherence over the potential for international comparisons. This has likely prevented wholescale critical assessments of methods and analyses applied at the national level. Because sampling practices vary widely, data compilation, integration and comparisons are challenging. Recent research has highlighted the limitations of certain juvenile abundance metrics and methodologies to provide unbiased assessments of both spatial variability (Millar et al. 2016) and temporal trends (Dauphin et al. 2019; Glover et al. 2019). The following sections explore the potential for collating and analysing existing juvenile trout density and habitat data, including opportunities for integration using spatial data (e.g., digital elevation models providing data that could be derived over the trout's native range) as an integrator of apparently incomparable datasets. Furthermore, we consider opportunities and constraints to the development of juvenile trout assessment methods and monitoring approaches across the species' distribution range.

4.2 | Characterising Juvenile Trout Density

Accurate estimates of fish densities are required to monitor population trends, underpin habitat-association models and develop assessment methods. Reliable estimates of fish density depend on unbiased and precise estimates of capture probability, which vary within and between river networks and over time. Spatial controls on capture probability include fish size, habitat (e.g., water level, channel morphology, substrate, presence of wood and hydraulic characteristics), electrofishing team (or organisation) and equipment (e.g., Speas et al. 2004; Hense, Martin, and Petty 2010; Benejam et al. 2012). Temporal controls potentially include changing fish size, water quality and quantity, teams, equipment and the sampling time (e.g., standardising sampling period to a narrow time window could affect latitudinal variability in capture probability due to variation in river temperature and fish size). Estimates of capture probability require multipass removal electrofishing sampling. Unfortunately, this type of sampling is resource intensive, thereby constraining the numbers of sites that can be sampled. Consequently, resource managers have often sought to use alternative, more rapid sampling methods to increase sample sizes. These include single-pass electrofishing, which is broadly equivalent to the first pass of a multipass method, and timed electrofishing where abundance is expressed in terms of fish caught per unit time. Some resource managers have uncritically considered single-pass and timed electrofishing to be reliable measures of relative abundance that allow comparisons between sites, and over time. However, this assumes constant capture probability across space and time, and there is strong evidence that is not true and that such approaches produce biased abundance estimates (Niemelä, Julkunen, and Erkinaro 2000; Benejam et al. 2012; Millar et al. 2016). Timed electrofishing is particularly problematic in this regard as it is affected both by spatio-temporal variability in capture probability and by sampling rate (the area surveyed by a team in a given time). Despite these concerns, single-pass electrofishing data can provide useful abundance estimates when combined with multipass electrofishing data in a suitable analytical framework (Wyatt 2002; Wyatt 2003; Rivot et al. 2008; Millar et al. 2016; Glover et al. 2018; Malcolm et al. 2023). This presents a reasonable compromise between data accuracy, sample size and associated spatial coverage that improves assessments.

Given these considerations, single-pass data alone are not suitable for developing assessment methods, or for monitoring population status and trends, but timed electrofishing would present additional challenges not easily addressed in a single modelling framework. Nevertheless, several potential options are available for harmonising collection and collation of juvenile trout electrofishing data across the species' native range. The simplest option would be for countries to provide density estimates only from multipass electrofishing sites. In many cases, these data are readily available, but the methods used to estimate densities may vary between countries and estimates of abundance obtained on a visit-by-visit basis can be imprecise, especially where fish numbers are low. Consequently, although this approach is pragmatic, it could also be considered suboptimal. With additional effort, it may be possible to collate a dataset that includes fish counts on each pass (separated by life stage or age) and associated information on wetted areas. This would allow capture probability to be modelled across all sites and used to obtain more precise estimates of density. This approach may improve the likelihood of detecting relationships between fish density and habitat that would be necessary in the development of new assessment methods. However, such an approach would allow for the integration of single-pass and multipass area-based electrofishing data in a single coherent modelling framework, increasing the availability of both historical and future data (Malcolm et al. 2023).

4.3 | Characterising Trout Habitat

Habitat can be measured, characterised and related to fish abundance across a wide range of spatial scales. At meso- and microhabitat (patch) scales, researchers have related the presence or abundance of fish to in-stream habitat variables such as depth, velocity, substrate size and invertebrate drift density (Millidine et al. 2012; Millidine, Malcolm, and Fryer 2016). These variables are typically measured directly in the field or derived from detailed hydraulic models. At larger spatial scales typically associated with individual electrofishing sites (Milner, Wyatt, and Broad 1998; Pedersen et al. 2017), fish densities are often correlated with summary measures of in-stream habitat characteristics. For example, mean depth or velocity can be obtained from transect surveys, while the percentage of different habitat features (e.g., pool, riffle, run) or the dominance of a particular habitat characteristic (e.g., dominant and subdominant substrate size, shade) can be visually approximated by survey teams at the time of sampling.

In recent decades, with the advent of Geographical Information Systems (GIS), spatial data and improved remote sensing techniques, there has been increasing interest in using landscape (or network) characteristics to understand and predict spatial variability in fish densities (Milner, Wyatt, and Broad 1998; Wyatt 2003; Kim and Lapointe 2011; Flitcroft et al. 2012). At large spatial scales, fish densities are often correlated with geographical and network position and proxies for habitat that can include landscape controls on channel morphology and water quality. For example, land use, geology and soil type can strongly influence water quality (Cresser et al. 2000) and nutrient status. Similarly, drainage area (as a proxy for discharge) and channel slope can affect the distribution of morphological units (Addy, Soulsby, and Hartley 2014) and substrate characteristics (Montgomery and Buffington 1997) (Figure 1). Given that salmonids, including trout, exhibit preferences for specific hydraulic, morphological and sedimentary characteristics and that these characteristics vary across rivers in response to landscape controls (Gleason 2015), these variables are potentially powerful predictors of ecosystem function (Doretto, Piano, and Larson 2020), fish species distribution and abundance. Indeed, where the relative explanatory power of landscape (or network) and in-stream habitat variables for predicting fish density has been compared, landscape predictors have been shown to be considerably more important. For example, Milner, Wyatt, and Broad (1998) found that landscape variables, derived from spatial data, accounted for about two-thirds of the spatial variance in trout fry densities explained by habitat models. Similarly, Flitcroft et al. (2012) established that network variables explained a greater percentage of spatial variation in juvenile coho salmon density than in-stream variables, which explained little variation.

In the current study, a wide range of in-stream habitat characteristics were considered as important predictors of trout juvenile density and included in the template used to capture information on methods. However, there was little consistency in the choice of habitat variables that are recorded, or the methods used to characterise them in national assessments. In some cases, habitat characteristics were not recorded at sampling sites, although reliable density data were available. In the short term, these substantial differences in data collection procedures limit opportunities for collation and harmonisation of in-stream habitat data at the target species range scale. In contrast, landscape- or network-scale habitat characterisation, which is available in specific and often free databases for almost all countries, provides an opportunity to generate macrohabitat descriptors with a consistent and standard approach for both historical and contemporary abundance data. Indeed, such an approach was recently undertaken in Scotland (Malcolm, Millidine, Glover, et al. 2019) where a lack of common data collection standards resulted in similar data constraints at a national scale. In recent years, the availability of Europe-wide standardised digital river networks (e.g., EU-Hydro—River Network Database, Version 1.3) and other spatial datasets including digital elevation (e.g., EU-DEM v1.1), the Catchment Characterisation Model (CCM) (De Jager and Vogt 2007) and flow accumulation datasets allows for the development of standardised regionally wide habitat characterisation in a way that was not previously possible.

Although current in-stream habitat data are generally inconsistent between countries, detailed analyses of the relationships between habitat and trout are underway for rivers in the Baltic (supported by the ICES Working Group for Baltic Salmon [WGBAST], and the European Union's Data Collection Framework [DCF]) and Scotland (through the National Electrofishing Programme for Scotland [NEPS]). These regional initiatives have the potential to identify the most important habitat variables for predicting juvenile trout abundance. When considered alongside the effort and cost required for obtaining different habitat measures, this offers a promising approach for prioritising future data collection and making recommendations for the standardisation of methods.

4.4 | Potential for Regional Biases Due to Historical Site Selection

Given the plasticity of trout biology and life history, and the difficulties in differentiating between origins at the juvenile stage, anadromous and resident trout are typically assessed together at the juvenile life stages although some anadromous trout assessments focus specifically on river reaches that are accessible to trout migrating from sea. Because of the constraints of electrofishing equipment, sampling is usually undertaken only in wadeable rivers, although some limited exceptions exist in deeper and still waters where electrofishing sampling may instead be conducted by boat. However, quantifying densities is more challenging in these settings (Thompson, White, and Gowan 1998), and these data cannot be readily integrated with data obtained by wading methods. Despite many commonalities in the approaches taken to site assessment, substantial regional variability remains. In part, this relates to the original purpose of the surveys, their objectives and whether trout were the key target species for the survey. For example, in Scotland, historical data derive from sites that were selected at a local level, typically on an ad hoc basis, and based on a variety of objectives. In these circumstances, the relationship between sampling sites and the wider target population is unknown. More recently, the NEPS has implemented a generalised random tesselation stratified (GRTS) survey design for site selection that ensures that sampling is representative of wadeable habitats that are accessible to salmon and sea trout, providing an unbiased sample of densities. Elsewhere, trout monitoring sites can be chosen specifically because they are in good quality trout habitats, excluding poorer sites that may nevertheless be suitable for trout. Moreover, sampling sites can also be chosen because they are more accessible to survey teams than other sites. Unless supported by statistical survey designs, differences in site selection procedures between regions can generate spatial biases in the abundance data used to support the development of habitat–abundance models and, thus, related assessment methods. These biases need to be considered carefully when interpreting model outputs, and any consequences for inference and reliability of assessment should be carefully considered.

4.5 | Development of Assessment Methods From Habitat–Abundance Models

There is a long history of developing fish assessment methods from habitat–abundance models (Milner, Wyatt, and Broad 1998). In common with assessment methods for other species, juvenile trout assessments typically involve a comparison of observed and expected densities (hereafter referred to as a benchmark) for a given habitat or location (Malcolm, Millidine, Glover, et al. 2019; Helcom 2022; ICES 2023a; ICES 2023b). However, within this general context, there are a wide variety of approaches that vary in terms of their philosophical basis, underlying data requirements and modelling approaches.

One common approach for defining benchmark densities is to select 'reference sites' where there are no known anthropogenic pressures (or where these are at least small) and then to model the relationships between average abundance at these sites and habitat characteristics (Milner, Wyatt, and Broad 1998). An extension to this approach involves modelling data only from sites with time-series data where habitat at a site is related to the long-term mean fish density, rather than the density observed in a single year (Wyatt 2005). The major difficulties with such approaches are in finding reference sites where ova deposition has been sufficient to saturate habitat each year and where there are few or no anthropogenic pressures. In many cases, this restricts reference sites to upland locations, often far from the coast and risks biasing available data to particular environments.

Where habitat is not thought to be saturated, some researchers have suggested modelling an upper percentile of observed abundance data or even maximum observed abundance (Pedersen et al. 2017). Alternatively, with sufficient years of data and stock variability, it may be possible to model fry-to-parr stock–recruitment relationships, allowing model parameters to vary with habitat, in a similar way to that explored for adult assessment methods using catchment scale data (Prévost 2015). A major constraint to all these approaches is the availability of time-series data and the spatial distribution of these data, which are often biased towards favourable sites with high salmonid abundance that have been chosen (ad hoc site selection) by fisheries managers. Furthermore, the choice of very high percentile values or maximum observed recruitment will inevitably result in unattainable river-scale management targets, especially where reference sites are also biased towards the best habitat.

An alternative approach for defining benchmark densities of salmonids involves modelling all the available electrofishing data for which accurate abundance estimates can be obtained, and including metrics of environmental pressures, habitat, time and geography as covariates in the models (Malcolm, Millidine, Glover, et al. 2019). Benchmark predictions can then be made while excluding the effects of pressures or location as appropriate. Benchmark predictions can also be made for a particular year, which could be useful in the case of declining stock levels where contemporary average density could be considered too low.

Setting aside philosophical and data requirements of different habitat–abundance models, there are also a wide range of statistical approaches available. These range from simple linear models such as HABSCORE (Milner, Wyatt, and Broad 1998) to more complex smoother-based approaches (Wyatt 2005; Malcolm, Millidine, Glover, et al. 2019) that allow for flexible fish-environment responses. In recent years, there has also been increasing interest in the use of machine learning approaches that can fit data more rapidly and are less constrained by data assumptions. However, the choice of method would seem less important than the choice of general approach, philosophy and data availability.

4.6 | Comparing Juvenile Benchmarks With Other Biologically Relevant Reference Points

Regardless of the approach taken to develop a juvenile salmonid assessment benchmark, it can be hard to relate these directly to other biological reference points commonly used in management such as maximum sustainable yield (MSY) or maximum production (Smax). The concept of 'Index monitoring rivers' is often used to refer to locations where detailed multilifestage (adult, ova, fry, parr, emigrant) salmonid monitoring data are collected. Such sites are extremely rare due to their costs of operation but are also extremely valuable, allowing the development of stock–recruitment relationships and providing unique insights into population dynamics and bottlenecks. Critically, in the case of juvenile assessment, this presents the opportunity to validate or to rescale benchmarks to other reference points.

Through carefully designed surveys (Malcolm, Millidine, Jackson, et al. 2019) or through appropriate spatial statistical river network modelling (Wyatt 2002; Glover et al. 2018), it is possible to estimate total river-scale juvenile salmonid production. By incorporating estimates of winter survival and migration mortality, these estimates of juvenile abundance can be used to approximate smolt production (Höjesjö et al. 2017). Smolt production estimates can be validated or further calibrated when compared to observed smolt production obtained from fixed or mobile traps (Molin, Kagervall, and Rivinoja 2010).

Where benchmarks are predicted from spatial datasets, it is possible to scale them (Malcolm, Millidine, Glover, et al. 2019) and thus to estimate benchmark densities or production at catchment scales. By investigating relationships between juvenile production and emigrant production (or adult returns, or ova deposition), it is possible to determine where juvenile benchmarks sit relative to other reference points. Because habitat–abundance models typically describe relative intersite differences in production, it would then be possible to rescale benchmarks if required, and if a suitably large number of index rivers were available to do this reliably. If index rivers were available across a large geographic range, it may also be possible to variably rescale on a spatial basis. At present, there are very few index rivers that also contain suitable juvenile data to undertake such an approach. However, this should be investigated further and encouraged where possible to compliment, calibrate and validate spatially extensive juvenile assessment methods.

There is a need for further work to determine the availability of reliable stock–recruitment data from index sites that include juvenile abundance. Often, index rivers have focussed on salmon and frequently only collect data on adult numbers. In fewer cases, emigrant production is also recorded. However, without detailed data on juvenile abundance, size and age, it is not readily possible to make comparisons of juvenile benchmarks and other more traditional stock–recruitment parameters typically derived from data on adult or ova numbers.

5 | Conclusions

Trout data have been collected for a wide range of purposes, including WFD compliance and as a by-product of salmon assessment. These data are rarely collected according to a formal statistical survey design and are, thus, susceptible to a variety of potential biases. In an ideal world, where resources were not constraining, a range-wide juvenile assessment programme for

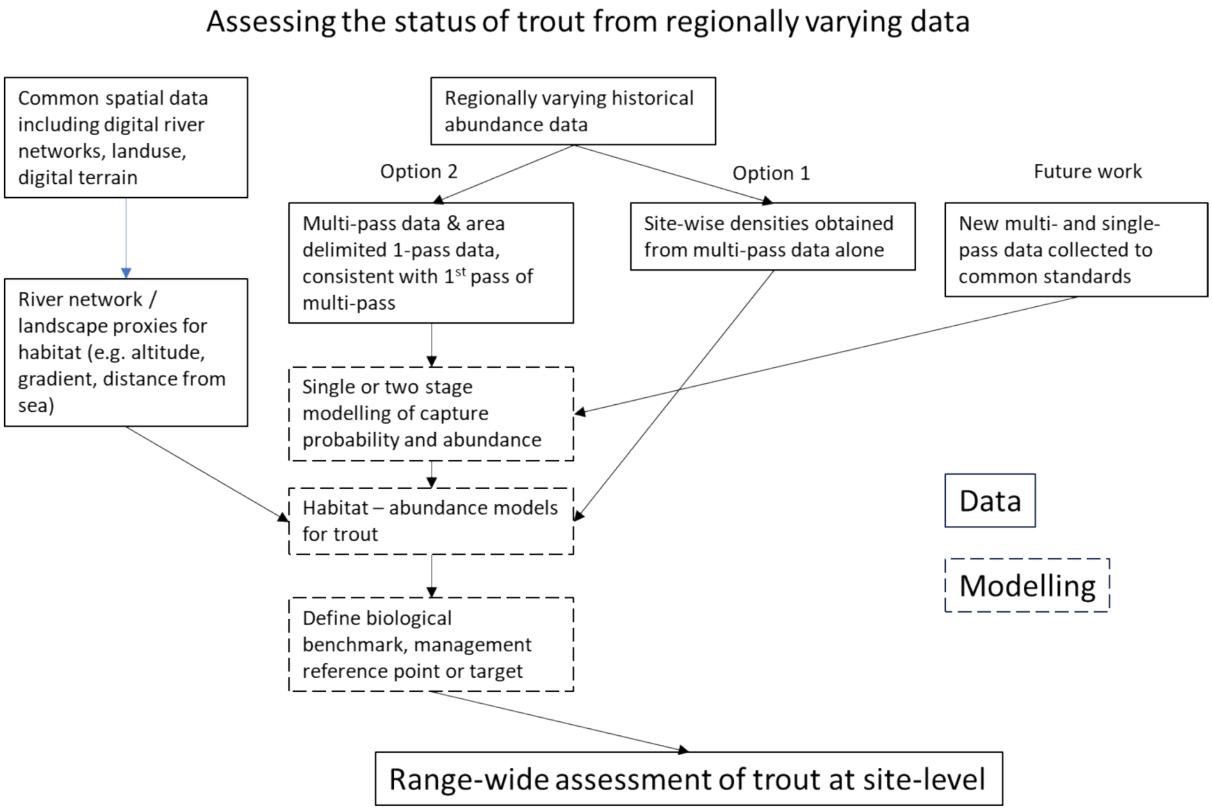

FIGURE 6 | Schematic diagram showing potential approaches for developing a range-wide trout assessment model from regionally varying abundance and habitat data. Also shown is the potential to harmonise future data collection methods, having established the relative importance of different habitat predictors from analyses of existing data.

trout, following standard methods and conducted using a formal statistical survey design, would present the best opportunities for obtaining a spatially comparable, unbiased assessment of resource. However, such an approach would appear unrealistic at present, especially given the results obtained from the survey conducted in this study, where several differences between and within countries were identified and there was no noticeable spatial pattern in applied protocols. This result suggests that administrative units override ecological ones, which can represent a severe impediment to large scale data set compilations, like the ones we need for the assessment of species of wide-range distribution such as the sea trout.

Despite these constraints, there are opportunities to make significant progress towards the development of range-wide habitat–abundance and assessment models as identified in this paper (Figure 6). Abundance estimates could be obtained from multipass sites, which provide estimates of capture probability and single-pass area delimited sites. This could involve two-stage capture probability models (e.g., Malcolm et al. 2023) or single-stage Bayesian approaches (e.g., Dauphin et al. 2019). In the case of habitat data, there are opportunities in the short term to make use of covariates derived from spatial data, that is, habitat proxies. In future, analyses from internally consistent NEPS and WGBAST programmes will identify the most important locally derived habitat variables, thereby prioritising data collection and standardisation in future work. As juvenile assessment methods are developed, it will be important that these can be related, compared and calibrated to other assessment approaches, for example, using adult data. In this context, index rivers (including subcatchments of larger rivers), where measurements of numbers, sizes and ages of adults, in-river juvenile production and emigrant production are available, are of the utmost importance.

Author Contributions

CMA, KM, AW, JH and IM were involved in conceptualisation; CMA and SS conducted questionnaire data analyses; IM, AW and JH provided suggestion on analyses; CMA, KM, PL, JH, MB and IM developed manuscript figures; CMA, SS and AW prepared and reviewed Supporting Information; CMA, MB and IM wrote the original draft; all coauthors reviewed and edited the manuscript.

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Ethics Statement

The authors have nothing to report.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

Data used on this manuscript are available in 10.5281/zenodo.14290125.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.