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Assessing the feasibility and value of employing an ecosystem services approach in chemical environmental risk assessment under the Water Framework Directive

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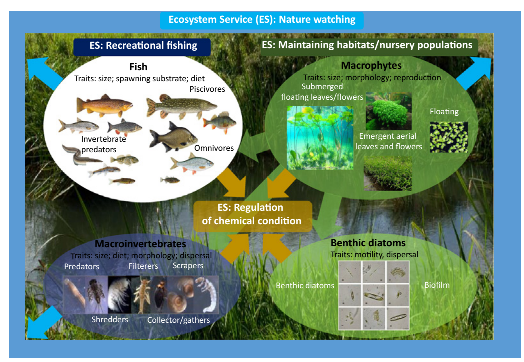
HIGHLIGHTS

- We present a site-specific case study based around the Water Framework Directive (WFD).
- WFD measurement endpoints for ecological receptors were linked to ecosystem services.
- The measured status of ecosystem services was compared to WFD reference values.
- The risk of zinc to ecosystem service delivery was assessed retrospectively.
- Risk to ecosystem service delivery was equivalent or lower than WFD ecological status.

GRAPHICAL ABSTRACT



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ABSTRACT

The feasibility and added value of an ecosystem services approach in retrospective environmental risk assessment were evaluated using a site-specific case study in a lowland UK river. The studied water body failed to achieve good ecological status temporarily in 2018, due in part to the exceedance of the environmental quality standard (annual average EQS) for zinc. Potential ecosystem service delivery was quantified for locally prioritised ecosystem services: regulation of chemical condition; maintaining nursery populations and habitats; recreational fishing; nature watching. Quantification was based on observed and expected taxa or functional groups within WFD biological quality elements, including macrophytes, benthic macroinvertebrates and fish, and on published functional trait data for constituent taxa. Benthic macroinvertebrate taxa were identified and enumerated before, during and after zinc EQS exceedance, enabling a generic retrospective risk assessment for this biological quality element, which was found to have good ecosystem service potential. An additional targeted risk assessment for zinc was based on laboratory-based species sensitivity distributions normalised using biotic-ligand modelling to account for site-specific, bioavailability-corrected zinc exposure. Risk to ecosystem services for diatoms (microalgae) was found to be high, while risks for benthic macroinvertebrates and fish were found to be low. The status of potential ecosystem service delivery (ESD) by fish was equivalent to high ecological status defined

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under the WFD, while ESD was higher for benthic macroinvertebrates than defined by WFD methods. The illustrated ecosystem services approach uses readily available data and adds significantly to the taxonomic approach currently used under the WFD by using functional traits to evaluate services that are prioritised as being important in water bodies. The main shortcomings of the illustrated approach were lack of: representation of bacteria and fungi; WFD predicted species lists for diatoms and macrophytes; site-specific functional trait data required for defining actual (rather than potential) ecosystem service delivery.

1. Introduction

Freshwater ecosystems provide multiple ecosystem services (Aylward et al., 2005), which have an estimated value of £39.5 billion per year in the UK alone (ONS, 2017). However, freshwater biodiversity, which underpins the delivery of many ecosystem services, is under threat. Globally, freshwater species are going extinct more rapidly than terrestrial or marine species, with freshwater vertebrate species declining by an average of 83% since 1970 (WWF, 2018) and a third of freshwater insects being threatened with extinction (Sánchez-Bayo and Wyckhuys, 2019). In Europe, 59% of freshwater molluscs and 40% of freshwater fish are threatened with extinction (IUCN, 2015). The main threats to freshwater ecosystems are habitat loss via changes in land and water use, exploitation of species, changing climates, pollution, harmful algal blooms, and invasive alien species including infectious disease organisms (Reid et al., 2019). These threats are a function of a wider set of indirect pressures that are linked to demographic, socio-cultural, economic, technological and policy drivers, as well as to human conflicts and epidemics (IPBES, 2019). Assessing the risk of these threats to ecosystem service delivery is key to managing freshwater ecosystems for the benefit of people and nature.

The EU Water Framework Directive (WFD; 2000/60/EC) is the primary regulatory tool for assessing and managing the quality of European freshwater ecosystems. The WFD commits European Members States to achieve for all surface water bodies good ecological status, or good ecological potential, reflecting minimal anthropogenic pressure. Following the 'one out all out' approach, ecological status is based on the lowest classification determined for a suite of biological, physico-chemical and hydro-morphological quality elements, prescribed in WFD Annex V (EC, 2000). The WFD has succeeded in standardising assessment methods for identifying key anthropogenic pressures (e.g. hydro-morphological modifications and pollution) and aiding decision making on remedial or mitigatory measures (Voulvoulis et al., 2017). However, implementation of measures has been limited due to insufficient resources, and their effectiveness has been hampered by lack of integration of water- and land- based environmental policy (Carvalho et al., 2019). Consequently approximately 60% of Europe's surface water bodies still fail to achieve good ecological status (EEA, 2018). The WFD's stringent 'one out all out' approach has raised questions among stakeholders over which quality elements are most sensitive to significant pressures and which are most reliable (less prone to uncertainty) (Carvalho et al., 2019). A further key question is which biological quality elements are most important for underpinning ecosystem services? (Vidal-Abarca et al., 2016; Carvalho et al., 2019). The targeting of monitoring and implementation of measures should ideally take account of all current and potential ecosystem services and their interactions with one another, such that trade-offs are maximised and optimal combinations of services are maintained (Blackstock et al., 2015). Targeting of ecosystem services will be river basin- specific and may require a wider or narrower set of quality elements than those currently prescribed under the WFD.

There is increasing recognition that river basins are highly interdependent systems, which require integrated management of environmental, societal and economic systems (Voulvoulis et al., 2017). River Basin Management Plans under the WFD are required to meet a number of potentially competing objectives (e.g. fisheries, water abstraction, nature conservation). This management approach is consistent with the ecosystem services approach (Everard, 2012; Spray and Blackstock, 2013; Vlachopoulou et al., 2014). Although ecosystem services are not explicitly mentioned in the wording of the WFD, there is some connection between the Directive and their delivery via the incorporation of ecosystem function, as well as structure, in the classification of ecological status (Vlachopoulou et al., 2014). Recent advancements in understanding how an ecosystem services approach can be applied to assessing risk to freshwater ecosystems include how ecosystem services map on to the WFD objectives (Vlachopoulou et al., 2014), how WFD indicators may provide information on ecosystem services (Vidal-Abarca et al., 2016) and how the ecosystem services approach can inform WFD river basin management plans (Grizzetti et al., 2016). The potential added value of an ecosystem services approach stems from gaining a more holistic perspective on potential multiple benefits and trade-offs associated with different river basin management options. For example the approach facilitates: i) alignment and co-delivery of complementary policy objectives (e.g. land use, flood risk and water quality management); ii) improved cost-benefit analysis through valuation of and payments for ecosystem services; iii) stakeholder engagement on desired outcomes (including non-statutory goals) (Vlachopoulou et al., 2014; Blackstock et al., 2015). The consideration of cultural and recreational ecosystem services in river basin management planning also serves to emphasise the direct link between environmental and human health and to promote stakeholder participation, which is central to the WFD (Ravenscroft and Church 2011; Ziv et al., 2016).

Chemical pollution is one of the main threats to the ecological status of freshwater ecosystems (Reid et al., 2019) and ecological or environmental risk assessment is the process by which the likelihood of adverse ecological effects of chemical pollutants are evaluated. The potential for an ecosystem services approach to be incorporated in chemical environmental risk assessment has been examined via a series of multi-stakeholder workshops involving industry practitioners, regulators, policy makers, third sector organisations and academic researchers (Maltby et al., 2018; Faber et al., 2019; Maltby et al., in prep.). Key benefits perceived by these stakeholders include: making ecological risk assessment spatially explicit (indicating what ecosystem services to protect and where); improving transparency in communicating risks, identifying trade-offs and synergies as part of environmental decision making; and integrating across multiple stressors, scales, habitats and policies. Key challenges include: dealing with increased complexity; satisfying increased data demands; linking ecological measurement endpoints to impacts on service providing units and final ecosystem services (Faber et al., 2021 in this issue); and establishing an ecosystem services framework for decision making for risk assessors and risk managers (Maltby et al., 2018; Faber et al., 2019). A top priority identified by stakeholders was to define reference values, or normal operating ranges (sensu Kowalchuk et al., 2003), indicating the ecological status of resident species populations and assemblages and then relate these indicators to the functional capacity of service providing units and to ecosystem services delivery. These are prerequisites for observing/discerning any significant deviations from the 'expected' ecological status and for subsequently discriminating any significant pollution effects from other environmental pressures on service providing units and ecosystem services (Faber et al., 2019).

Using a site-specific case study in the UK, we demonstrate the proof of concept of using an ecosystem services approach in retrospective chemical environmental risk assessment. In particular we address the challenges of i) linking measurement endpoints for ecological receptors to ecosystem services; ii) relating measurement endpoints and assessment of potential ecosystem service delivery to reference values; iii) assessing the risk of a specific chemical pollutant to potential ecosystem service delivery. We also compare and contrast WFD- and ecosystem services-based assessments in terms of their protection of freshwater ecosystems in order to iv) evaluate how an ecosystem services approach could be used in conjunction with, and add value to, the current assessment of ecological status under the WFD, including helping to prioritise and identify where remedial measures are likely to have greatest benefit.

2. Materials and methods

The proof of concept case study was based on a small lowland river, located 45 m above sea level, with a mean water flow of 1.1 $m^3\ s^{-1}$. The river (shown in Fig. 1) is a tributary of the River Wey in the Thames catchment and is typical of suburban waterbodies in the UK. Our study utilised WFD monitoring data for the selected water body, incorporating both biological and chemical elements, including zinc. Zinc is an essential metal in many living organisms, but at elevated concentrations in water is classified as a river basin specific pollutant under the WFD. At the case study site, monitoring data showed that dissolved zinc concentrations recently exceeded the environmental quality standard (EQS) of 10.9 μg/L $Zn_{added,bioavailable,}$ and in 2018 this constituted a failure to achieve good ecological status. The source of zinc pollution was a contaminated lagoon at a nearby industrial site, which overflowed intermittently into the river during flood events.

WFD monitoring data for biological quality elements underpinning ecological status were sought from the UK Environment Agency's database BIOSYS for the period before (2014–2017), during (2018) and after (2019) the EQS for zinc was exceeded. Field data collection and processing methods employed by the Environment Agency are outlined in UK Technical Advisory Group guidelines (UK TAG, 2008; Willby et al., 2012; UK TAG, 2014a; UK TAG, 2019). Benthic macroinvertebrate data were available for all time periods (2014–2019), but diatom, macrophyte and fish data were only available for the period before the zinc EQS exceedance (2014–2015). Before the predicted zinc impact (i.e. 2014–2015) the ecological status of benthic diatoms, macrophytes and benthic macroinvertebrates was classified as Moderate, due to lower than expected species diversity and/or the preponderance of nutrient tolerant species (SI Table 1a–e). Benthic macroinvertebrates remained at Moderate status during (2018) and after (2019) the predicted impact. The ecological status of the water body with respect to the local fish community was High in 2014 according to the Fisheries Classification System v.2 (FCS2) (SI Table 1f).

A wide range of ecosystem services described under the UK National Ecosystem Services Assessment (UK NEA, 2014) are provided by the waterbody, of which 12 services (39%) were considered to be at risk according to the local River Basin Plan and accompanying UK Environment Agency's Appraisal Summary Tables (Table 1). The status of prioritised ecosystem services, including any impacts from zinc, was assessed in the following methodological steps, which addressed each of the challenges i-iii) identified at the outset of the study (Fig. 2).

2.1. Mapping of biological quality elements to locally prioritised ecosystem services

Mapping of WFD biological quality elements (sensu Vidal-Abarca et al., 2016) was used as a framework for linking measurement endpoints for ecological receptors to ecosystem services. Mapping focused on ecosystem services that were prioritised in the selected water body

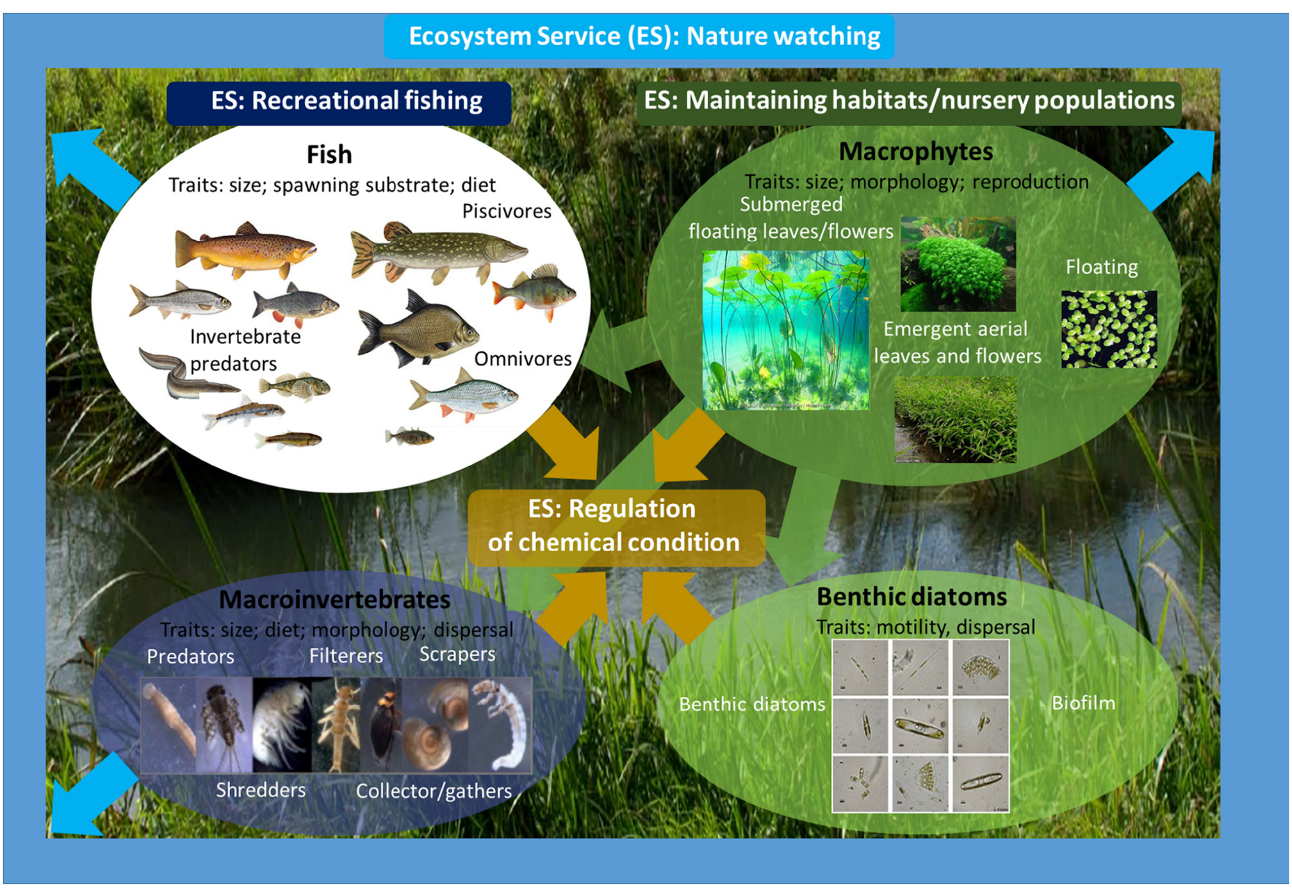

Fig. 1. Prioritised ecosystem services (ES) delivery by WFD biological quality elements at the case study site.

Table 1
Environment Agency Appraisal Summary Table for ecosystem services at the case study site.
Ecosystem services prioritised by the Environment Agency river basin manager are highlighted in bold.
* According to the UK National Ecosystem Assessment (UK NEA, 2014) this ecosystem service is split between two intermediate services: "larval and gamete supply" and "formation of species habitat". We have adopted the summative terminology relating to CICES 2.2.2.3, version 5.1 (Haines-Young and Potschin, 2018).

Ecosystem services	Expected impacts	Possible mitigatory measures
Provisioning services		
Fresh water	Impaired water quality (WQ) e.g. pesticide runoff	Catchment sensitive farming, improve effluent treatment
Water for non-consumptive use	Impaired water flow from hydro-power generation	Install locks, fish passes etc., improve hydromorphology
Food	WQ impacts on fish farming and crayfish harvesting	Catchment sensitive farming, effluent treatment
Regulating services		
Climate regulation (local temperature/precipitation, greenhouse gas sequestration)	Impoundment of water and sediment, reduced riparian woodland & wetland areas	Habitat restoration
Water flow regulation	Flood plain development, reduced interconnectivity	Floodplain reconnection, habitat restoration
Erosion regulation	Agricultural land use, impact on soils, incl. erosion	Catchment sensitive farming, restore channel morphology
Water purification and waste treatment	Impaired WQ e.g. elevated nutrient levels	Fix sewerage misconnection, wastewater treatment
Cultural services		
Cultural heritage (incl. nature watching)	WQ impacts on health of ancient water meadows	Improve WQ and habitats, reduce abstraction
Recreation and tourism (including fishing)	WQ and water flow impacts angling, canoeing, bathing	Improve WQ and habitats, reduce abstraction
Pest regulation	Not described	Not described
Disease regulation	Not described	Not described
Supporting service		
Maintaining nursery populations and habitats*	Moderate WQ, habitat quality & amenity value	Improve WQ, hydromorphology for self cleaning system

according to the local River Basin Plan, Appraisal Summary Tables and local Environment Agency personnel (Table 1). Prioritisation criteria for ecosystem services included perceived amenity value, current baseline, consequence of intervention versus no intervention, cost-benefit analysis and assessment of probability of success of mitigatory intervention(s). Following initial mapping, taxa representing biological quality elements were allocated to each of the prioritised ecosystem services. Where possible, taxa were retained in distinct functional groups (e.g. macroinvertebrate filter feeders, scrapers, shredders, collector/gatherers) within biological quality elements, in order to provide a 'common currency' for comparative evaluation with the results from the WFD assessment of ecological status. The potential contributions from WFD biological quality elements and constituent functional taxa towards ecosystem services delivery were assessed from the published scientific literature, including the review by Vidal-Abarca et al. (2016) and a supplementary review undertaken as part of this case study.

2.2. Assessment of observed versus expected levels of ecosystem service delivery

Observed taxa × abundance data were compared with 'expected' values (corresponding to type-specific reference conditions) for the selected water body, according to a suite of WFD habitat template models. Expected primary producer trophic indices or functional groups were output from the habitat template models 'DARLEQ2' (UK-TAG, 2014c) and 'LEAFPACS' (Willby et al., 2012) for diatoms and macrophytes respectively. Expected taxa × abundance data for benthic macroinvertebrates were output from the 'River Invertebrate Classification Tool - RICT' (UK TAG, 2008) and for fish from the 'Fisheries Classification Scheme - FCS2' (UK TAG, 2019). The ratio of observed/expected values generates Ecological Quality Indices (EQI), which are used conventionally under the WFD to define the ecological status of the water body according to established quality classification boundaries for each biological quality element (Table 2).

In this study quantification of potential ecosystem service delivery from observed and expected functional groups or taxa was based on functional trait data obtained from key publications and databases (Macrophytes i.e. hydrophytes – Willby et al., 2000; Invertebrates (focusing on water-borne/larval life-stages) - Hershey and Lamberti, 2001; Fish – FishBase https://www.fishbase.de/; All taxa https://www.freshwaterecology.info/about.php; http://www.freshwaterplatform.eu/). Key functional traits underpinning ecosystem service delivery by different biological quality elements are specified in SI Table 2. Traits were not readily discernible for benthic diatoms from the literature. Key traits for macrophytes for providing habitats and maintaining nursery populations and for nature watching were size, morphology, reproductive mode (e.g. large emergent macrophytes with aerial or floating flowers). Large submerged or floating macrophytes with rhizomes were considered to play a key role in the regulation of chemical condition, including zinc (Rai, 2009). Key traits for macroinvertebrates with respect to nature watching were size, morphology and dispersal (e.g. larger taxa, visible from the water surface, including taxa with winged adults). Diet, as well as size and morphology, were considered to be key traits for macroinvertebrates for regulating chemical condition (e.g. larger taxa including filter feeders, collector/gatherers and bioturbators) (Leslie and Lamp, 2017). Key traits for fish with respect to recreational fishing and nature watching were size, morphology, trophic position, edible/non-edible (e.g. larger fish including predators and salmonid/game fish). Fish traits considered important for regulation of chemical condition were size and diet (e.g. large predatory fish with significant potential to uptake metals such as zinc) (Andres et al., 2000).

Ecosystem Service Potential (ESP) scores representing the potential contribution of different taxa to delivering each prioritised ecosystem service were determined on a scale of 1 to 5 (low to high), based on expert judgement and multiple criteria assessment for the key traits considered to underpin each service. ESP scores were determined for individual functional taxa recorded present in the water body (i.e. function was assumed based on presence) and scores were peer reviewed by the authors. For example, for macrophytes, maximum ESP scores (5/5) for Nature watching were awarded to emergent macrophytes with aerial or floating flowers, while maximum ESP scores for Regulation of chemical condition were awarded to large submerged or floating macrophytes with rhizomes. Mean ESP scores were calculated for taxonomic assemblages representing macrophytes, benthic macroinvertebrates and fish (SI Table 2). Taxa abundance was also included in the quantification of potential Ecosystem

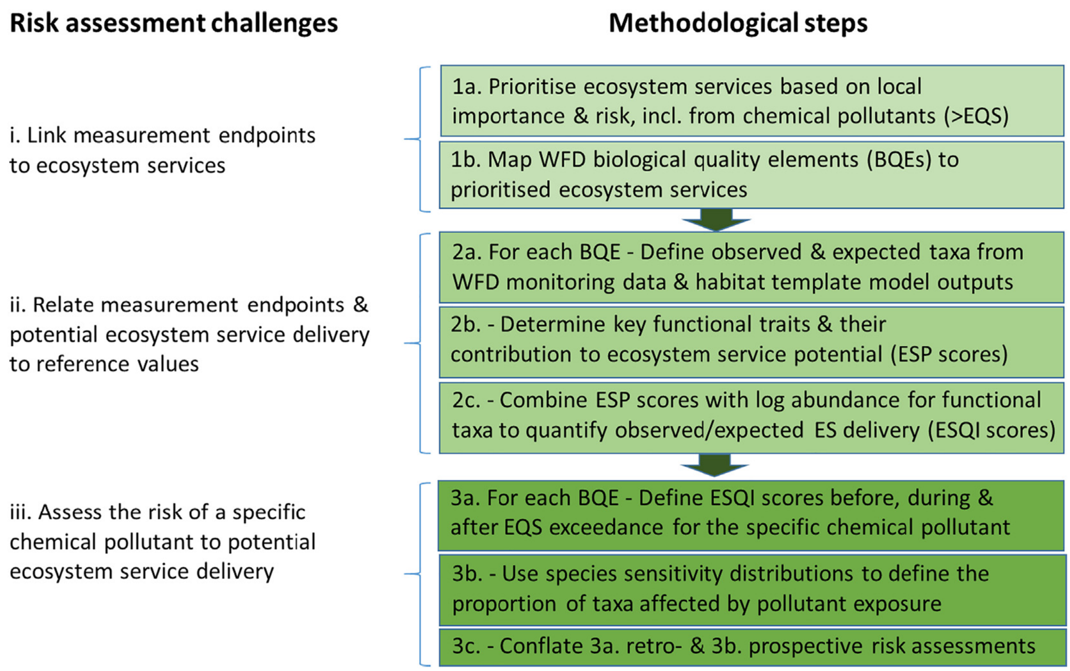

Fig. 2. Flow diagram summarising the process of assessing the risk of a specific chemical pollutant to potential ecosystem service delivery in a WFD water body.

Service Delivery (ESD) for these biological quality elements (i.e. function was assumed to increase asymptotically, in proportion with log abundance). Relative (log) abundance scores were calculated (as follows) and added to the ESP scores for each taxon. Ecosystem Service Delivery per taxon was quantified up to a 'notional' maximum score of 10, based on a maximum ESP score of 5, plus a maximum abundance score of 5, which equated to 100% cover for macrophyte taxa (SI Table 3), or relative abundance scores calculated as $\log_{10}(x + 1)$ abundance per 0.1 m^2 for macroinvertebrate taxa and $\log_2(x + 1)$ abundance per 100 m^2 for fish taxa. Mean Ecosystem Service Delivery scores were then calculated for the observed and expected taxa for benthic macroinvertebrates and for fish. Overall Ecosystem Services Quality Index (ESQI) was determined for each of these biological quality elements based on Observed/Expected mean Ecosystem Services Delivery. The number of functional groups and the mean number of taxa per functional group, per biological quality element were also calculated and used as indicators of functional diversity and functional redundancy respectively (Schmera et al., 2017). This enabled the calculation of additional ESQIs based on Observed/Expected functional diversity and Observed/Expected functional redundancy (including for macrophytes). Ecosystem Services Quality Index boundaries were identical to those established for Ecological Quality Indices for each WFD biological quality element (Table 2).

2.3. Assessment of the impact of zinc on prioritised ecosystem services

Evaluation of the contribution of zinc towards impairment of prioritised ecosystem services in the selected waterbody was based initially on overall mean Ecosystem Services Quality Indices for benthic macroinvertebrates and indices for individual functional groups, determined before (2014–2017), during (2018) and after (2019) exceedance of the environmental quality standard. This assessment of impacts on ecosystem services also included general physico-chemical water quality parameters (e.g. pH, dissolved oxygen, ammonia concentration), in order to account for these potentially confounding factors. A similar approach was used to account for the effect of hydromorphological quality elements (e.g. river flow) according to Environment Agency guidelines (UK TAG, 2014b). Biotic indices/scoring systems routinely employed by the Environment Agency were also used to gauge impacts on macroinvertebrate fauna assemblages from organic enrichment (according to Biological Monitoring Working Party scores - BMWP scores; Hawkes, 1997) or from impaired river flows (according to Lotic invertebrate Indices for Flow Evaluation – LIFE scores; Extence et al., 1999).

Broader evaluation of ecotoxicological risk from zinc in the selected waterbody was performed following the standardised EU-accepted procedure outlined by the UK Technical Advisory Group for the WFD (UK TAG, 2013) and the Technical Guidance Document for Implementing Bioavailability based Environmental Quality Standards for Metals (EU Commission, 2020; in press). The evaluation was based on species sensitivity distributions for zinc, which incorporate microalgae, fish and macroinvertebrates. Annual mean measured, background-corrected, bioavailable zinc concentrations obtained before, during and after exceedance of the environmental quality standard were normalised, based on mean local water chemistry conditions measured in each year (pH, dissolved organic carbon and dissolved calcium), using biotic ligand models M-BAT (v.30) (UK TAG, 2014c), Bio-Met (v.5) (Bio-Met, 2019) and the IZA Full-BLM (EU, 2010). The use of these models enabled a three tier risk assessment (outlined in detail in SI Tables 4–6). A key advantage of the IZA Full-BLM is that it is linked directly to an extensive, curated database of ecotoxicological data for zinc (EU, 2010) and it outputs species sensitivity distributions (SSDs) which were used to

Table 2
Ecological Quality Index boundaries for measures of functional and taxonomic diversity within WFD biological quality elements.

	High/good	Good/moderate	Moderate/poor	Poor/bad	Reference
Benthic diatoms	0.8	0.6	0.4	0.2	UK TAG (2014a)
Macrophytes	0.8	0.6	0.4	0.2	Willby et al. (2012)
Benthic macro-inverts	1.0	0.8	0.6	0.4	UK TAG (2008)
Fish	1.0	0.8	0.55	0.3	FCS2 (2019)

determine local bioavailability-corrected hazardous concentrations (HC5), that are protective of 95% of local aquatic organisms (all taxa, macroinvertebrate taxa, fish taxa, as required).

3. Results

3.1. Mapping of biological quality elements to locally prioritised ecosystem services

Initial mapping, based on literature reviews, indicated that all prioritised ecosystem services could be contributed to by at least one WFD biological quality element in the selected waterbody (Table 3). Regulation of chemical condition is potentially delivered by all biological quality elements, due to their propensity to uptake, adsorb or partition zinc (e.g. via sediment burial or resuspension). Nature watching, is generally considered to be delivered by macrophytes with flowers and emergent leaves, charismatic benthic macroinvertebrates such as odonates (damselflies and dragonflies) and fish such as perch, pike and trout. Maintenance of nursery populations and habitats was considered to be delivered primarily by large rooted, branched macrophytes. A nursery is defined as a habitat that contributes more than the average, compared with other habitats, to the production of individuals of a species that recruit to adult populations (Beck et al., 2001). Recreational fishing was considered to be delivered by all fish, but with a distinction being drawn between more prized, edible game fish and non-edible coarse fish.

3.2. Assessment of observed versus expected levels of ecosystem service delivery

A comparative assessment of potential ecosystem service delivery based on observed and expected taxa was not possible for benthic diatoms, since the outputs from the habitat template model DARLEQ2 are expected trophic indices, rather than expected taxa or functional groups. The habitat template model for macrophytes (LEAFPACS) output expected functional groups, enabling a limited functional assessment of potential ecosystem service delivery. Higher resolution comparative assessment of potential ecosystem service delivery, based on observed versus expected functional taxa, was possible for benthic macroinvertebrates and fish.

There was generally close agreement between our observed and expected Ecosystem Service Potential (ESP) scores, particularly for macroinvertebrates. Macroinvertebrate taxa scored most highly and consistently (pre- and post-zinc EQS exceedance) in terms of regulation of chemical condition, with >95% of taxa having ESP scores of 3 or 4, versus a maximum of 5 (Table 4); highest scoring taxa were filter feeding bivalve molluscs and deposit feeding oligochaete worms. Macroinvertebrate ESP scores for nature watching were notably lower during and post- zinc EQS exceedance compared to pre- exceedance, due to the absence of visible surface dwelling species and emergent winged species. There was a greater discrepancy between observed and expected ESP scores for fish (pre-exceedance), due to the lack of three expected species: eel (*Anguilla anguilla*), dace (*Leuciscus leuciscus*) and trout (*Salmo trutta*). The absence of these fish species had a greater impact on observed versus expected scores for recreational fishing than for regulation of chemical condition and nature watching.

Our ESP scores per taxon (SI Table 2) were summed with the relative abundance of each taxon (SI Table 1) to give potential Ecosystem Service Delivery (ESD) scores (Table 5). For benthic macroinvertebrates expected mean ESD scores ranged from 4.6/10 to 5/10, with highest values being calculated for regulation of chemical condition. Expected ESD scores for fish were highest for Recreational fishing (4.2/10) followed by Nature watching (3.8/10). Observed ESD scores were found to be similar or higher than expected ESD scores, with resulting ESQIs (observed/expected scores) for macroinvertebrates ranging from 0.8 to 0.9 and ESQIs for fish ranging from 1.3 to 1.4 across prioritised ecosystem services.

Resulting ESQIs were high (>0.8) based on observed versus expected numbers of functional groups of macrophytes. There were 8 observed versus 5.5 expected functional groups, giving an ESQI of 1.45 for macrophyte functional diversity, while the observed number of taxa per functional group was 1.5 versus an expected value of 1.6, giving an ESQI of 0.96 for macrophyte functional redundancy. The same approach was applied to benthic macroinvertebrates and fish. Observed versus expected numbers of functional groups of benthic macroinvertebrates were 5 versus 6, giving an ESQI of 0.83 and the mean numbers taxa per functional group were 5.8–6.6 versus 7.2, giving an ESQI of 0.81–0.92. Observed versus expected numbers of functional groups for fish were 5 versus 7, giving an ESQI of 0.71. Observed versus expected numbers of taxa per functional group were 2 versus 1.9, giving an ESQI of 1.08 (Table 5).

Ecosystem service delivery scores and quality indices (observed/expected) were resolved for individual benthic macroinvertebrate functional groups in 2018 (during exceedance of the EQS for zinc), in order to evaluate their comparative status and also levels of functional redundancy in terms of ecosystem service provision (Table 6). Regulation of chemical condition was good (ESQI >0.8) across all functional groups except for collector/gatherers, while nature watching was generally moderate ($0.6 < \text{ESQI} < 0.8$) except for predators and shredders which were classified as good. However, ESQIs based on numbers of taxa per functional group indicated that functional redundancy for predators and scrapers was limited/poor (ESQI < 0.6).

3.3. Assessment of the impact of zinc on prioritised ecosystem services

Annual mean measured environmental concentrations of background-corrected, bioavailable zinc ($\text{MEC} = Zn_{\text{added,bioavailable}}$) exceeded the UK annual average EQS (10.9 μg/L) from 2016 to 2018 (SI Table 5b) and according to standardised EU-accepted risk assessment (UK Technical Advisory Group, 2013) risk was confirmed for 2018 (SI Table 5c). Derived ESQIs for macroinvertebrates (Table 5) indicated that there were no perceptible differences in potential ecosystem service delivery before, during and after exceedance of the environmental quality standard for zinc. In addition there was no indication (from biotic indices/scoring systems employed by the Environment Agency) of impacts

Table 3
Initial assessment of the possible contribution of WFD biological quality elements towards the delivery of prioritised ecosystem services.

Ecosystem services	Biological quality elements and key functional traits for ecosystem service delivery			
	Benthic Diatoms	Macrophytes	Macro invertebrates	Fish
Regulation of chemical condition	All species (nutrient uptake primary production[1,2])	All species (nutrient uptake, photosynthesis)[3]	All species, but especially filter-feeders & collector-gatherers[5,6]	All species (via uptake incl. via food webs)[8,9]
Maintaining nursery populations and habitats		Submerged and emergent species (provide habitat & cover)[3]		
Recreational fishing				Game fish and coarse fish species[10]
Nature watching		Attractive flowering species[4]	Insects, especially odonates[7]	All species[10]

References: 1) Roberts et al., 2007; 2) Lavoie et al., 2008; 3) Aguiar et al., 2013; 4) Hoyle et al., 2017; 5) Covich et al., 1999; 6) Bonada et al., 2006; 7) Lemelin (2007); 8) Holmlund and Hammer, 1999; 9) Andersson et al., 1978, 10) FishBase, 2020.

Table 4

Ecosystem Services Potential scores (ESP scores) for observed/expected lists of key functional taxa (presence/absence).

Mean ecosystem services potential scores (ESP scores) are recorded in SI Table 2. For diatoms and macrophytes there was no indication of expected taxa from the respective habitat template models, therefore ecosystem service delivery could not be bench-marked. Macrophytes were the only biological quality element responsible for maintaining nursery populations and habitats, so this ecosystem service is omitted below.

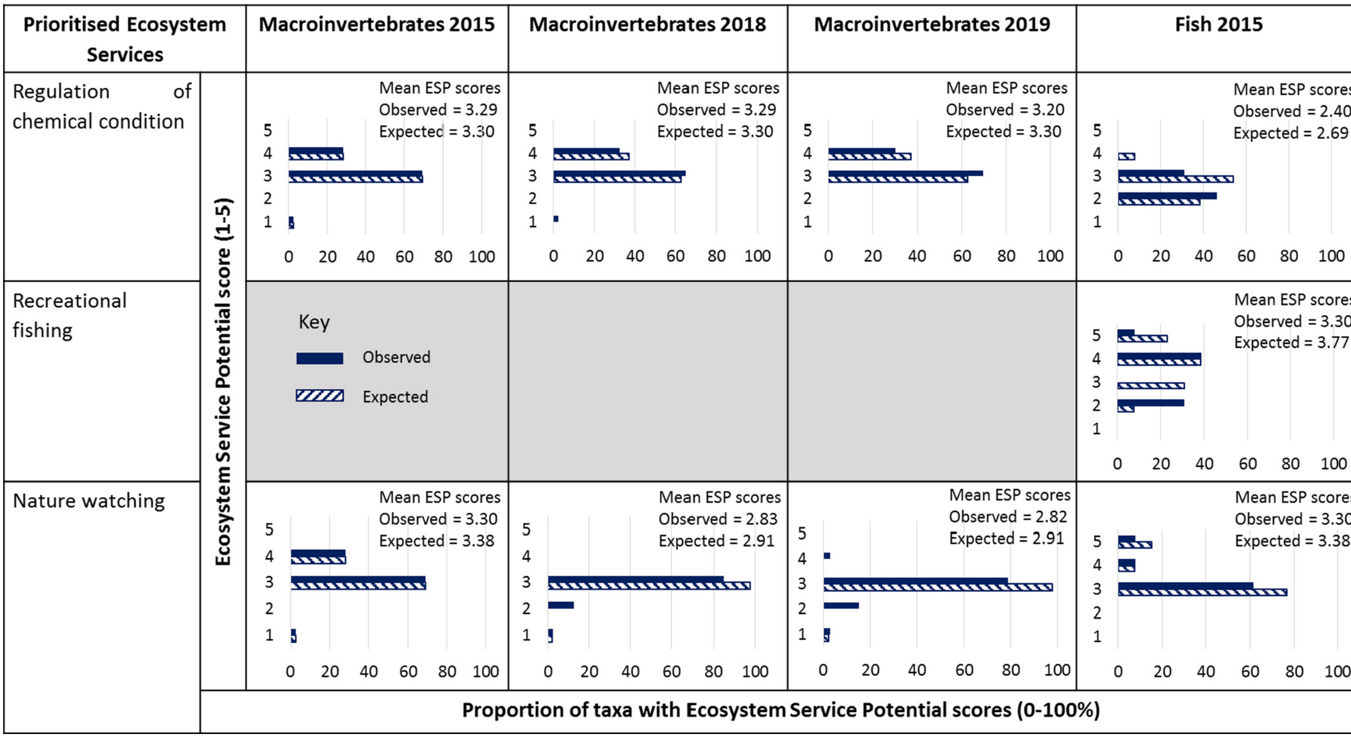

Table 5

Ecosystem Service Quality Index (ESQI) based on observed and expected functional groups, taxa or derived metrics for each biological quality element.

For diatoms there was no indication of expected taxa from the habitat template model, therefore ecosystem service delivery could not be bench-marked. Benchmarking was possible for macrophytes (based on expected numbers of functional groups) and for benthic macroinvertebrates and fish (based on expected numbers of taxa, their Ecosystem Service Potential scores (ESP scores) recorded in SI Table 2 and their relative abundance scores recorded in SI Table 1). Data for macroinvertebrates were available before (2015), during (2018) and after (2019) the EQS exceedance for zinc. Ecosystem Service Delivery = relative abundance score + ESP score (scores provided in SI Tables 1 and 2, respectively). Mean Ecosystem Service Delivery was calculated for all taxa within each biological quality element. Ecosystem Service Quality Index (ESQI) = Observed / Expected; Blue indicates where ESQIs meet high status (>1.0); Green indicates where ESQIs meet good status (0.8–1.0); Orange indicates where ESQIs meet moderate status.

Biological quality element	Metric	Functional diversity (number of functional groups)	Functional redundancy (mean number of taxa per functional group)	Potential ecosystem service delivery			
				Recreational fishing	Nature watching	Maintaining nursery populations and habitats	Regulation of chemical condition
Macrophytes	Expected	5.5	1.6	–	–	–	–
	Observed 2015	8	1.5	–	3.68	4.04	4.18
	ESQI 2015	1.45	0.96	–	–	–	–
Benthic macro-invertebrates	Expected	6	7.2	–	4.6	–	5.0
	Observed 2015	5	6.2	–	3.95	–	4.47
	ESQI 2015	0.83	0.86	–	0.86	–	0.89
	Observed 2018	5	6.6	–	3.66	–	4.15
	ESQI 2018	0.83	0.92	–	0.80	–	0.83
	Observed 2019	5	5.8	–	3.75	–	4.23
	ESQI 2019	0.83	0.81	–	0.82	–	0.84
Fish	Expected	7	1.9	4.2	3.8	–	3.1
	Observed 2014	5	2	5.3	4.9	–	4.2
	ESQI 2014	0.71	1.08	1.26	1.28	–	1.38

Table 6
Ecosystem Service Quality Index (ESQI) based on observed and expected taxa within macroinvertebrate functional groups in 2018.
Ecosystem Service Quality Index (ESQI) = Observed / Expected; Blue indicates where ESQIs meet high status (>1.0); Green indicates where ESQIs meet good status (0.8–1.0); Orange indicates where ESQIs meet moderate status. The functional group 'Piercers' (incl. Corixidae and Hydroptilidae) were expected in the selected waterbody, but not observed.

Macroinvertebrate functional groups	Metric	Functional redundancy (number of taxa per functional group)	Potential ecosystem service delivery			
			Recreational fishing	Nature watching	Maintaining nursery populations and habitats	Regulation of chemical condition
Collector/gathers	Expected	9	–	4.86	–	5.09
	Observed	13	–	3.55	–	3.93
	ESQI	1.44	–	0.73	–	0.77
Filter feeders	Expected	3	–	5.47	–	6.47
	Observed	4	–	4.19	–	5.19
	ESQI	1.33	–	0.77	–	0.80
Predators	Expected	12	–	4.02	–	5.18
	Observed	6	–	3.71	–	4.71
	ESQI	0.50	–	0.92	–	0.91
Scrapers	Expected	8	–	4.48	–	4.48
	Observed	4	–	3.43	–	3.68
	ESQI	0.50	–	0.77	–	0.82
Shredders	Expected	9	–	4.59	–	4.59
	Observed	6	–	3.86	–	3.86
	ESQI	0.67	–	0.84	–	0.84

on macroinvertebrates from organic enrichment or from impaired river flows. EQSIs could not be derived for other biological quality elements, since taxonomic and/or function group data were only available before exceedance of the environmental quality standard for zinc.

According to species sensitivity distributions (SSDs) output from the IZA Full-BLM (SI Table 6b), the risk of bioavailable zinc causing toxicity was highest for microalgae (c.f. diatoms) with 2/2 (100%) of species in the SSD found to be at risk i.e. exhibiting no observed effect concentrations (NOECs) below the maximum annual average measured environmental concentration (MEC_{max}) in 2018. According to the SSD and recent EFSA guidance (EFSA, 2019) risk for diatoms could be equated to a potential large effect on ecosystem service delivery. However this assessment is based on only two algal species from the SSD (*Raphidocelis subcapitata*, *Chlorella* sp.), neither of which are diatoms. For benthic macroinvertebrates 2/6 taxa were found to be at risk (c.f. the Fraction Affected from the SSD model FA = 31%) and 1/8 fish was found to be at risk (c.f. FA = 16.7%) (Fig. 3). Risk for benthic macroinvertebrates and fish was equated to a potential small effect (10–35% of functional taxa affected). Furthermore, there was no indication of any systematic variation in the sensitivity of functional groups within these biological quality elements (Fig. 3). The relatively short-term exposure-effects data (for taxa with short life-cycles), on which the SSDs are based, are appropriate for the risk assessment, given the intermittently elevated concentrations of zinc in 2018.

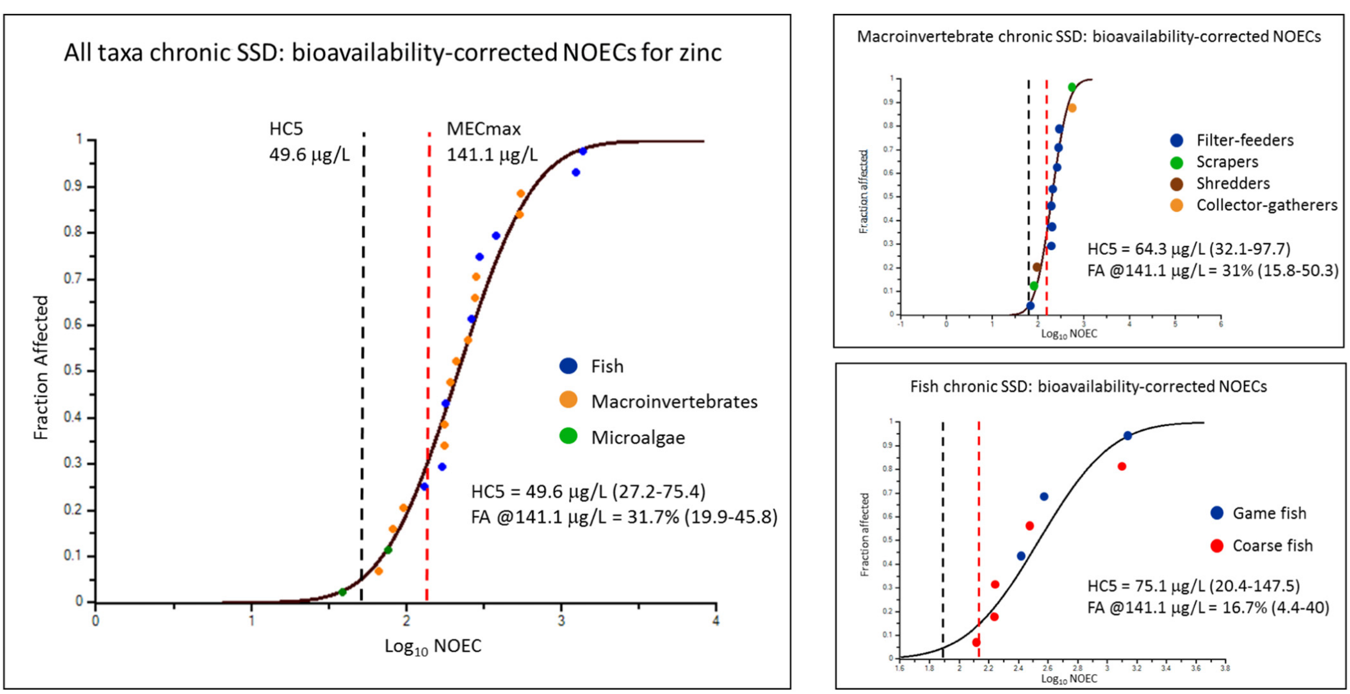

Fig. 3. Species sensitivity distributions of site-specific (bioavailability-corrected) chronic NOECs for zinc. NOECs were bioavailability-corrected based on local water chemistry conditions for the case study site using the IZA Full-BLM (SI Table 7b) The hazardous concentration affecting 5% of taxa (HC5) and maximum annual average measured environmental concentration (MECmax = 141.1 µg/L) from 2018 are indicated by a black dashed line and a red dashed line respectively. The fraction of taxa affected at 141.1 µg/L zinc is indicated for each taxonomic group = median value (5–95% confidence limits). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Macrophytes were not included in the species sensitivity distributions for zinc, due to lack of availability of curatable data for inclusion in the IZA database i.e. data which meet all quality criteria defined in the EU Technical Guidance Document (EC, 1994) and additional considerations set out in the European Risk Assessment for zinc (EU, 2010).

4. Discussion and conclusions

The results of our study demonstrate that it is feasible to employ an ecosystem services approach in site-specific chemical environmental risk assessment by i) linking WFD measurement endpoints for ecological receptors to ecosystem services; ii) relating WFD measurement endpoints and assessment of potential ecosystem service delivery to reference values; iii) assessing the risk of a specific chemical pollutant to potential ecosystem service delivery. However, our work highlights a number of limiting factors for each of these three steps, which ultimately constrain the evaluation of ecosystem service delivery. Our study also serves to demonstrate iv) the added value of ecosystem services approaches compared to the conventional evaluation of ecological status under the WFD.

4.1. Linking WFD measurement endpoints for ecological receptors to ecosystem service delivery

Measuring and predicting changes in ecosystem services associated with improving or deteriorating ecological status remains a key challenge for designing and implementing effective water policy and regulation (Blackstock et al., 2015). Key questions, which so far remain unanswered are “Does improving ecological status (defined under the WFD) result in increased capacity/synergy for the delivery of ecosystem services?” and “Does Good ecological status equate to Good ecosystem service delivery?” (Maes et al., 2018). The answers to these questions may not be universally applicable, since they will depend on local species assemblages, locally prioritised ecosystem services and spatio-temporal variation in environmental pressures. For example, the prioritization of ecosystem services may vary considerably between River Basin Districts and these services may not always be adequately represented by WFD biological quality elements (Vidal-Abarca et al., 2016; Kagalou and Latinopoulos, 2020).

The present case study showed that assessment of the ecological status of WFD biological quality elements (benthic diatoms, macrophytes, benthic macroinvertebrates and fish) provides a solid (but somewhat narrow) foundation for linking ecological receptors to locally prioritised ecosystem services. Elsewhere, WFD biological quality elements have been linked to several key regulatory ecosystem services such as bioremediation, disease control, environmental flow regimes, flood mitigation and nutrient cycling (Burkhard and Müller, 2008). However, a recent mapping exercise concluded that less than 50% of all freshwater ecosystem services were adequately represented by macrophytes and fish, and fewer still by benthic diatoms and macroinvertebrates (Vidal-Abarca et al., 2016). WFD biological quality elements exclude bacteria, fungi, non-fish vertebrates, non-benthic invertebrates and non-diatom algae, all of which contribute to key ecosystem services. Bacteria, fungi and non-diatom algae are particularly important in the regulation of chemical condition through biodegradation of organic pollutants/matter, which can in turn affect the bioavailability of non-biodegradable metals such as zinc (EU, 2010; Faburé et al., 2015). Non-fish vertebrates and non-benthic invertebrates have also been shown to be important for regulating services (e.g. pest control, pollination and seed dispersal) (Hevia et al., 2017). Published literature have generally focused until now on establishing links to regulating services from taxa, functional groups, service providing units and/or WFD biological elements, while links to other ecosystem services have received less study (Burkhard and Müller, 2008; Hevia et al., 2017).

In addition to addressing the incomplete coverage of ecosystem services and underlying service providing units by WFD biological quality elements, there is a need for evidence of quantitative (rather than semi-quantitative) linkages between ecosystem structure and function and ecosystem service delivery. Establishing these quantitative linkages via ecological production functions is recognised as a priority under Common Implementation Strategy for the WFD (Reyjol et al., 2014). Measurement of the impairment of ecosystem service delivery is currently based on generic/arbitrary criteria for evaluating the magnitude of effects on population(s), functional group(s), biodiversity (EFSA, 2019).

In our study we considered both the presence/absence (diversity) and also the relative abundance of functional groups, since both are essential for the quantification of ecosystem services delivery (Luck et al., 2009; Winfree et al., 2015). We also relied on our own expert judgement in assigning values to key traits that contribute to Ecosystem Service Potential (ESP) scores for functional taxa within WFD biological quality elements (SI Table 2). Key functional traits were considered to be size/biomass, morphology, diet and reproductive strategy. Valuation of these key traits could be validated through wider expert peer review. Our approach could be further refined by including other traits, such as rates of food consumption, metabolism, somatic and population growth. These additional traits are highly interdependent and variable with local conditions, season, life-stage and consequently less extrapolatable to other situations (Faber et al., 2021 in this issue). Accounting for spatial and temporal variability in functional traits is acknowledged as being a key challenge for quantifying ecosystem service delivery (Balvanera et al., 2014). When modelling the effects of environmental stressors on ecosystem service delivery through the use of functional traits, it is helpful to differentiate ‘response traits’, which determine responses to stressors, from ‘effect traits’, which are more clearly linked to the provision of ecosystem services - the focus of our adopted approach (De Bello et al., 2010; Días et al., 2013; Lavorel, 2013; Valiente-Banuet et al., 2015). Nevertheless response traits, which quantify species responses to different environmental factors (Friberg et al., 2011) can serve as diagnostic tools to identify specific cause(s) of ecological impairment and prioritise remedial or mitigatory measures (Baattrup-Pedersen et al., 2019). For some regulating ecosystem services, such as regulation of chemical condition, it is important to appreciate that an association detected between a prospective service providing unit and an ecosystem service could imply dependency rather than causality. For instance, fish may be more diverse in clean waters as a result of good water quality rather than the other way around (Vidal-Abarca et al., 2016).

Further refinements to our methodology could be made by better understanding and quantifying the relative efficiencies of different taxa within service providing units (c.f. biological quality elements in our study) (Luck et al., 2009). Further work could also be devoted to acquiring chemical exposure-response data to quantify the effects of chemicals (like zinc) on population numbers and growth rates for key functional taxa. Quantifying recovery rates for functional taxa and linking this through to recovery in ecosystem service delivery will also be critically important for informing pollution remediation options. These refinements need to be considered in the future development of ecological production functions, which currently provide only limited linkages between functional diversity and ecosystem delivery, and between functional redundancy and ecosystem service resilience (Nyström, 2006; Tilman et al., 2006; Faber et al., 2021 in this issue).

It is also important to gain further evidence and understanding (in general) for quantifying of the delivery of cultural ecosystem services (UK NEA, 2014). A cursory evaluation of cultural ecosystem service delivery was made in our study by considering the morphological and functional traits of WFD biological quality elements that are most likely to appeal to nature watchers and recreational fishers (anglers), such as colourful and charismatic flora and fauna (McGinlay et al., 2017). Ideally our evaluation of locally prioritised cultural services, as well as the actual use, should be validated through local stakeholder engagement, which is an integral part of the river basin management planning review process.

Finally, the importance of traits underpinning ecosystem service delivery should ideally be considered in conjunction with ecosystem structure. One way this can be achieved is to adopt an analytical framework that considers the magnitude, spatial scale, sustainability and resilience of ecosystem structure and function versus historical baselines and/or future expectations concerning ES delivery (Maes et al., 2018). The downside of such an approach is the need for extensive monitoring data. In our case study this data need is partially met by WFD monitoring data. Nevertheless, it is acknowledged that it would be advantageous if a future revision of the WFD could include a wider array of quality elements and ecosystem services than are currently included (Kagalou and Latinopoulos, 2020).

4.2. Relating WFD measurement endpoints and assessment of potential ecosystem service delivery to reference values

The comparison of observed and expected functional taxa or functional groups within WFD biological quality elements enabled quantification of potential ecosystem service delivery in relation to expected reference values, via the calculation of Ecosystem Service Quality Indices (ESQIs), which are analogous to WFD Ecological Quality Indices (EQI). The use of historic habitat type-specific reference conditions in the assessment of ecological status under the WFD is consistent with approaches used in conservation and restoration management, in which historical conditions remain the cornerstone for target setting (Kopf et al., 2015). However, the use of static, historical reference conditions contradicts a central tenet in ecosystem-based approaches, which is that ecological (e.g. successional and evolutionary) changes are inevitable within all ecosystems, regardless of anthropogenic influences, leading to continually shifting baselines (UK NEA, 2014). This realisation has led conservationists to consider that the management of human-dominated ecosystems must move beyond historical constraints towards new points of reference dictated by social–ecological sustainability (Kopf et al., 2015) and a changing world (Bouleau and Pont, 2015; Moomaw et al., 2018). Taking account of shifting baselines is not the same as allowing standards to slide. The adoption of new baselines should be justified by long-term trends and extenuating circumstances, regardless of whether they lie above or below historical baselines.

In practice, accounting for shifting baselines in the evaluation of ecosystem service delivery by WFD biological quality elements will require periodic updating of WFD habitat template models and/or the definition of site-specific, rather than habitat-specific, reference conditions (Nõges et al., 2015), or the use of reference-free indicators (Tweedley et al., 2017). A further practical issue highlighted by the present study is that (in the UK) WFD habitat template models for benthic diatoms and macrophytes are limited to outputting metrics, such as Trophic Diatom Index and River Macrophyte Nutrient Index, or number of functional groups. Lists of taxa and their abundances expected under type-specific reference conditions (e.g. provided by benthic macroinvertebrates and fish habitat template models) offer far more scope for assessing potential ecosystem service delivery. This scope could also be provided for diatoms and macrophytes via the use of internationally calibrated taxonomic lists shown to be indicative of type-specific reference conditions (Kelly et al., 2009; Birk and Willby, 2010). It has been demonstrated in terrestrial and aquatic systems that ecosystem service delivery rarely increases linearly with taxonomic diversity and abundance. For example, ecosystem service delivery can plateau with increasing biodiversity, but more importantly there may be a precipitous reduction in ecosystem service delivery when biodiversity declines below a certain threshold (Balvanera et al., 2014). It is also important to note that in a range of terrestrial systems, the composition of functional taxa and the abundance of key taxa within functional groups have been shown to have much greater bearing on ecosystem service delivery than simply the number of functional taxa (richness) (Hooper and Vitousek, 1997; Heemsbergen et al., 2004).

4.3. Assessing the risk of a specific chemical pollutant to potential ecosystem service delivery

There were no perceptible impacts on potential ecosystem service delivery before, during or after exceedance of the EQS for zinc. Based on WFD monitoring data, potential ecosystem service delivery by macroinvertebrates was consistently found to be good for both nature watching and regulation of chemical condition. Data for macrophytes and fish were only available before exceedance of the zinc EQS, preventing a targeted risk assessment for these biological quality elements. However, nutrient enrichment was indicated in the studied waterbody by the presence of nutrient-tolerant benthic diatoms, macroinvertebrates and macrophytes, and corresponded with locally elevated concentrations of ortho-phosphate (EA, 2019). This highlights that the attribution of chemical-specific effects on ecological status and ecosystem service delivery can be complicated by the co-existence of multiple pressures, including nutrient enrichment and variable water flows (EEA, 2018). We controlled for these extraneous pressures by using laboratory-derived species sensitivity distributions for zinc, bioavailability-corrected for the studied waterbody. Accordingly, risks to locally prioritised ecosystem services were related primarily to microalgae, although toxicity data were only available for two green algae (Chlorophyta: *Raphidocelis subcapitata* and *Chlorella* sp.). If these green algae were considered suitable surrogates for diatoms (Bacillariophyceae in the phylum Ochrophyta), they would represent a small fraction (~3%) compared to the 61 benthic diatom species recorded in the selected water body (Table 1a). The represented fraction of microalgae is anticipated to be far lower compared to the full range of microalgae likely to be present. Limitations in assessing ecological risk using small numbers of standard test species are offset to some extent by the use of species which are physiologically sensitive and naïve, i.e. not adapted to local chemical exposures (SCHER, SCENIHR, SCCS, 2011). On the limited basis of the SSD results zinc was assumed to present a risk to microalgae, and this was equated to a large effect according to draft EFSA guidelines (EFSA, 2019). Microalgae are key components of biofilms, which are the predominant microbial life-form in rivers and play a major role in biogeochemical cycling and the associated ecosystem service of water purification (Sabater et al., 2002; Faburé et al., 2015; Battin et al., 2016). It is also important to appreciate, however, that multiple trophic/functional groups often underpin ecosystem services, and, in this study, all four WFD biological quality elements were recognised as contributors to the regulation of chemical condition. Macrophytes in particular are likely to play a prominent role in the selected water body, since they are generally highly tolerant and have considerable potential to uptake and adsorb dissolved metals including zinc, which has led to their growing use in heavy metal 'phyto-remediation' (Rai, 2009; Li et al., 2015). Most importantly, these plants possess rhizomes, which are capable of precipitating and concentrating waterborne heavy metals by 'rhizo-filtration' (Dushenkov et al., 1995), and can accumulate up to 60% of their dry weight as toxic metals (Salt, 1995). Very high bio-concentration factors (BCFs) for zinc have been reported in the rhizomes and roots of water fern, *Azolla filiculoide* (12,000) (Sela et al., 1989); and pondweed, *Potamogeton* spp. (6600) (Hutchinson and Stokes, 1975). Uptake of metals, such as zinc, can also occur in the shoots and leaves, through direct uptake or translocation from other plant parts, including the rhizomes (Rai, 2009).

When evaluating the results of ecotoxicity data for multiple species, such as the species sensitivity distributions for zinc, it is important to appreciate how species sensitivity varies within taxonomic and functional groups composing broad trophic groups or biological quality elements (algae, invertebrates, fish). Sensitivity can vary considerably (e.g. some species within a given trophic group may have very high NOECs, while others have a relatively low NOECs (Larras et al., 2012). Ultimately the loss of some species may not necessarily lead to a net loss of function (due to functional redundancy) or a decline in ecosystem service delivery. Our study highlights several uncertainties around quantifying the impacts of specific pollutants such as zinc on ecosystem services. Uncertainties can be reduced by when similar results are generated from independent risk assessment approaches, employing field versus laboratory data, and/or biological versus chemical classifications. However, in reality mismatches are often observed. For example, biological status now often exceeds physico-chemical status, following the recent imposition of more stringent environmental quality standards (EQSs) for chemicals under the WFD (EA, 2019; EA, 2020). These mismatches and the results from our case study raise the following key questions: How can we develop a more integrated systems-based approach to water management? Are remedial programmes of measures justifiable or necessary in situations in which ecosystem service delivery is maintained at or above reference levels, while ecological and/or chemical status fail to meet reference levels prescribed under the WFD? How can the holistic ecosystem services approach be reconciled with the "one out all out" approach enshrined in the WFD (Voulvoulis et al., 2017; Giakoumis and Voulvoulis, 2019).

4.4. Added value of ecosystem services approaches compared to the conventional evaluation of ecological status under the WFD

Adopting an ecosystem services approach in which ecosystem services are prioritised by local stakeholders can greatly increase the focus and relevance of retrospective environmental risk assessment processes, aid decision making and help direct mitigation measures where they are needed/valued most. For example ecosystem service-based risk assessments may indicate that some services are more sensitive or conversely more resilient than indicated by current WFD assessments of ecological status. In the present case study, the status of locally important (prioritised) ecosystem services was equivalent to or higher than the measured ecological status of underlying WFD biological quality elements. This is because ecological status is largely dependent on taxonomic (rather than functional) diversity and may therefore be influenced substantially by the presence/absence of rarer, often more sensitive taxa. Alternatively, ecosystem service delivery often depends on multiple functional taxa, potentially leading to high levels of functional redundancy (Schmera et al., 2017). This redundancy was 'modelled' in our study by calculating mean potential ecosystem service delivery (as ESP and ESQI scores) for individual WFD biological quality elements and/or underlying functional groups. However, it is important to note that taxonomic diversity can be highly important and has been linked with increasing provision and resilience of a range of ecosystem services to multiple environmental stressors (Oliver et al., 2015; Whittingham, 2011). Assessment of potential ecosystem services delivery versus taxonomic diversity should ideally have wide geographical coverage. EU wide inter-calibration of taxonomic based assessments of the ecological status of WFD biological quality elements provides an excellent foundation for additional calibration of ecosystem services delivery versus taxonomic diversity. Our UK-based study provides a proof of concept that could be developed and tested more widely.

The selected site-specific case study illustrated the potential for exploiting synergies in ecosystem services delivery. There were numerous instances in which there was strong inter-dependence between ecosystem services, which rely on essentially the same service providing units. For example recreational fishing and nature watching both rely directly upon the presence of diverse and abundant assemblages of fish, invertebrates, and also macrophytes, which contribute to the maintenance of nursery populations and habitats. These services in turn rely upon the regulation of chemical condition of the water body. These inter-relationships highlight the fundamental importance of regulatory services (Burkhard and Müller, 2008), as well as potential synergies in ecosystem service delivery, for example when increasing taxonomic and functional diversity contribute to better regulation of chemical condition (Smith et al., 2017; Pienkowski et al., 2019). Our study also served to illustrate trade-offs in ecosystem service delivery. For example, benthic macroinvertebrates play prominent roles in sequestering or burying metal pollutants and regulating chemical condition (e.g. filter feeding bivalve molluscs, deposit feeding insect larvae and bioturbating worms), but are less appealing to nature watchers. Conversely more aesthetic/charismatic invertebrate and vertebrate fauna, such as odonate larvae (i.e. Cordulegasteridae dragonfly larvae), are generally less functional and/or pollutant tolerant, and therefore less important in regulating pollution (Jacob et al., 2017). Nevertheless, the lack of ecosystem service representation by WFD biological quality elements limits the scope for making trade-offs and also assessments of the costs versus benefits of different remediation options within programmes of measures under the Directive.

As we have shown, WFD measurement endpoints can be extrapolated using functional trait data to link ecological status (based on ecological structure) to potential ecosystem service delivery. A further extension to implementing this ecosystem services approach is to consider the actual use (exploitation) and socio-economic value of ecosystem services (Vallecillo et al., 2019). The EU Environment Action Programme (7th EAP) called for the integration of natural capital and ecosystem services evaluation into accounting and reporting systems at European Union and national levels (EU, 2013). The evaluation of ecosystem services is now being undertaken within EU Member States (Vallecillo et al., 2019) and in the UK, following HM Government Guidance on appraisal and evaluation for national investments (HM Treasury, 2018). More specifically, ecosystem services are considered in assessing the costs and benefits for River Basin Management Planning (UK EA, 2013). Thus the environmental, social and economic appraisal of WFD programmes of measures includes (*inter alia*): local importance and extent of use of ecosystem services; magnitude of change in ecosystem services expected over time, with and without remedial measures. Fulfilling this level of options appraisal requires the development and use of ecological and economic modelling tools for projecting future outcomes. These models will require site-specific data, which could be derived in part from WFD monitoring data using the approach outlined in this study.

4.5. Declaration-of-competing-interests

A Ross Brown devised the methodology, undertook the data analysis and writing of the manuscript. Stuart Marshall, Chris Cooper and Paul Whitehouse helped with data acquisition, analysis, reporting and preparation of the manuscript. Paul Van den Brink and Jack Faber helped supervise the data analysis, reporting and reviewing of the manuscript. Lorraine Maltby was responsible for funding acquisition, study design and supervision, data analysis and editing of the manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References

[HM] Her Majesty's Treasury (2018). Green Book: Central Government Guidance on Appraisal and Evaluation. URL (accessed December 2019): https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/685903/The_Green_Book.pdf

Aguiar, F.C., Segurado, P., Urbanič, G., et al., 2013. Comparability of river quality assessment using macrophytes: a multi-step procedure to overcome biogeographical differences. Sci. Total Environ. 476–477, 757–767.

Andersson, G., Berggren, H., Cronberg, G., Gelin, C., 1978. Effects of planktivorous and benthivorous fish on organisms and water chemistry in eutrophic lakes. Hydrobiologia 59 (1), 9–15.

Andres, S., Ribeyre, F., Tourencq, J.N., Boudou, A., 2000. Interspecific comparison of cadmium and zinc contamination in the organs of four fish species along a polymetallic pollution gradient (Lot River, France). Sci. Total Environ. 248 (1), 11–25.

Aylward, B., Bandyopadhyay, J., Belausteguigotia, J.-C., Börkey, P., Cassar, A., Meadors, L., Saade, L., Siebentritt, M., Stein, R., Tognetti, S., Tortajada, C., Allan, T., Bauer, C., Bruch, C., Guimaraes-Pereira, A., Kendall, M., Kiersch, B., Landry, C., Rodriguez, E.M., Meinzen-Dick, R., Moellendorf, S., Pagiola, S., Porras, I., Ratner, B., Shea, A., Swallow, B., Thomich, T., Voutchkov, N., Chopra, K., Leemans, R., Kumas, P., Simons, H., 2005. Freshwater ecosystems services. Ecosystems and Human Well-Being: Policy Responses. vol. 3. Millenium Ecosystem Assessment, Island Press Washington, pp. 215–258.

Baattrup-Pedersen, A., Larsen, S.E., Rasmussen, J.J., Riis, T., 2019. The future of European water management: demonstration of a new WFD compliant framework to support sustainable management under multiple stress. Sci. Total Environ. 654, 53–59.

Balvanera, P., Siddique, I., Dee, L., Paquette, A., Isbell, F., Gonzalez, A., Byrnes, J., O'Connor, M.I., Hungate, B.A., Griffin, J.N., 2014. Linking biodiversity and ecosystem services: current uncertainties and the necessary next steps. Bioscience 64, 49–57.

Battin, T.J., Besemer, K., Bengtsson, M.M., Romani, A.M., Packmann, A.I., 2016. The ecology and biogeochemistry of stream biofilms. Nat. Rev. Microbiol. 14 (4), 251–263.

Beck, M.W., Heck, K.L., Able, K.W., Childers, D.L., Eggleston, D.B., Gillanders, B.M., Halpern, B., et al., 2001. The identification, conservation and management of estuarine and marine nurseries for fish and invertebrates. Bioscience 51, 633–641.

Bio-Met (2019). Bioavailability Tool User Guide (version 5.0), Guidance document on the use of the bio-met bioavailability tool. URL (accessed December 2019): www.bio-met.net

Birk S, Willby N (2010). Towards harmonization of ecological quality classification: establishing common grounds in European macrophyte assessment for rivers. Hydrobiologia 652, 149–163 (2010). https://doi.org/10.1007/s10750-010-0327-3

Blackstock, K.L., Martin-Ortega, J., Spray, C.J., 2015. Implementation of the European Water Framework Directive: what does taking an ecosystem services-based approach add? In: Martin-Ortega, J., Ferrier, R.C., Gordon, I.J., Khan, S. (Eds.), Water Ecosystem Services: A Global Perspective. International Hydrology Series. Cambridge University Press , pp. 57–64. http://www.cambridge.org/bn/academic/subjects/earth-and-environmental-science/hydrology-hydrogeology-and-water-resources/water-ecosystem-services-global-perspective

Bonada, N., Prat, N., Resh, V.H., Statzner, B., 2006. Developments in aquatic insect biomonitoring: a comparative analysis of recent approaches. Annu. Rev. Entomol. 51, 495–523.

Bouleau, G., Pont, D., 2015. G. Did you say reference conditions? Ecological and socio-economic perspectives on the European Water Framework Directive. Environ. Sci. Pol. 47, 32–41.

Burkhard, B., Müller, F., 2008. Indicating ecosystem health and integrity. In: Denhardt, A., Petschow, U. (Eds.), Sustainability in River Basins—a Question of Governance. Ökom Verlag, München, pp. 35–57.

Carvalho, L., Mackay, E.B., Cardoso, A.C., Baattrup-Pedersen, A., Birk, S., Blackstock, K.L., et al., 2019. Protecting and restoring Europe's waters: an analysis of the future development needs of the Water Framework Directive. Sci. Total Environ. 658, 1228–1238.

Covich, A.P., Palmer, M.A., Crowl, T.A., 1999. The role of benthic invertebrate species in freshwater ecosystems. Zoobenthic species influence energy flows and nutrient cycling. Bioscience 49 (2), 119–127.

De Bello F, Lavorel S, Díaz S, Harrington R, Bardgett R, Berg M et al. (2010) Functional traits underlie the delivery of ecosystem services across different trophic levels. Biodivers. Conserv. 143: 2873–2893.

Días, A.T.C., Berg, M.P., De Bello, F., Van Oosten, A.R., Bila, K., Moretti, M., 2013. An experimental framework to identify community functional components driving ecosystem processes and services delivery. J. Ecol. 101, 29–37.

Dushenkov, V., Kumar, P.B.A.N., Motto, H., Raskin, I., 1995. Rhizofiltration—the use of plants to remove heavy metals from aqueous streams. Environ. Sci. Technol. 29 (5), 1239–1245.

EA Environment Agency (2019). 2021 river basin management plans Chemicals Published: October 2019. URL (accessed November 2020): https://consult.environment-agency.gov.uk/environment-and-business/challenges-and-choices/user_uploads/chemicals-in-the-water-environment-challenge-rbmp-2021-2.pdf

EA Environment Agency (2020). Catchment Data Explorer. URL (accessed November 2020): https://environment.data.gov.uk/catchment-planning/

EC (1994). Technical Guidance Document on risk assessment, TGD Part 2. URL (accessed December 2020):https://echa.europa.eu/documents/10162/16960216/tgdpart2_2ed_en.pdf.

EC European Communities (2000). Ecoregions for rivers and lakes. Annex XI, Water Framework Directive, in Official Journal of the European Communities. URL (accessed December 2019): https://www.eea.europa.eu/data-and-maps/data/ecoregions-for-rivers-and-lakes.

EEA, European Environment Agency (2018). European waters Assessment of status and pressures 2018. EEA Report No 7/2018. URL (accessed September 2020): file:///C:/Users/arb213/Downloads/European%20waters%20assessment%202018.pdf

EFSA European Food Standards Authority, 2019. DRAFT scientific opinion addressing the state of the science on risk assessment of plant protection products for in-soil organisms. EFSA Panel on Plant Protection Products and their Residues (PPR). European Food Safety Authority (EFSA), Parma, Italy.

EU (2010) European Union risk assessment report zinc metal. Office for Official Publications of the European Communities, Luxembourg. URL: (accessed September 2020): http://publications.jrc.ec.europa.eu/repository/handle/JRC61245.

EU (2013). Decision no 1386/2013/EU of the european parliament and of the council of 20 November 2013 on a general union environment action programme to 2020 'living well, within the limits of our planet'. Off. J. Eur. Union L 354/171.

EU Commission (2020). Technical Guidance Document for Implementing Bioavailability based Environmental Quality Standards for Metals. In Press

Everard, M., 2012. Why does "good ecological status" matter? Water Environ. J. 26, 165–174.

Extence, C.A., Balbi, D.M., Chcadd, R.P., 1999. River flow indexing using British benthic macro-invertebrates: a framework for setting hydro-ecological objectives. Regul. Rivers: Res. Manage. 15, 543–574.

Faber, J.H., Marshall, S., van den Brink, P.J., Maltby, L., 2019. Priorities and opportunities in the application of the ES concept in risk assessment for chemicals in the environment. Sci. Total Environ. 651, 1067–1077.

Faber JH, Marshall S, Brown AR, Holt A, van den Brink PJ, Maltby L (2021). Identifying ecological production functions for use in ecosystem services-based environmental risk assessment of chemicals. Sci. Total Environ., Accepted manuscript STOTEN-D-21-00219R1.

Faburé, et al., 2015. Impact of an urban multi-metal contamination gradient: metal bioaccumulation and tolerance of river biofilms collected in different seasons. Aquat. Toxicol. 159, 276–289.

FCS2 (2019). Annex 6 of FCS2 and fish barrier tool. URL (accessed December 2019): https://www.wfduk.org/sites/default/files/Media/Environmental%20standards/Annex%206%20Rivers%20Fish%20FCS2%20%26%20Fish%20Barrier.pdf

FisBase (2020). URL (accessed December 2020): https://www.fishbase.se/search.php.

Friberg, N., Bonada, N., Bradley, D.C., Dunbar, M.J., Edwards, F.K., Grey, J., Hayes, R.B., Hildrew, A.G., Lamouroux, N., Trimmer, M., Woodward, G., 2011. Biomonitoring of human impacts in freshwater ecosystems: the good, the bad and the ugly. Adv. Ecol. Res. 44, 1–68.

Giakoumis, T., Voulvoulis, N., 2019. Water Framework Directive programmes of measures: lessons from the 1st planning cycle of a catchment in England. Sci. Total Environ. 668, 903–916.

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A., Cardoso, A.C., 2016. Assessing water ecosystem services for water resource management. Environ. Sci. Pol. 61, 194–203. https://doi.org/10.1016/j.envsci.2016.04.008.

Haines-Young R, Potschin MB (2018). Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure. URL (accessed December 2020): www.cices.eu

Hawkes, H.A., 1997. Origin and development of the biological monitoringworking party score system. Water Res. 32 (3), 964–968.

Heemsbergen, D.A., Berg, M.P., Loreau, M., van Haj, J.R., Faber, J.H., Verhoef, H.A., 2004. Biodiversity effects on soil processes explained by interspecific functional dissimilarity. Science 306, 1019–1020.

Hershey, A.E., Lamberti, G.A., 2001. Stream macroinvertebrate communities. In: Naiman, Bilby (Eds.), River Ecology and Management: Lessons from the Pacific Coastal Ecoregion, Pages 169–199, Publ. 2001. Springer, New York.

Hevia, V., Martín-López, B., Palomo, S., García-Llorente, M., de Bello, F., González, J.A., 2017. Trait-based approaches to analyze links between the drivers of change and ecosystem services: synthesizing existing evidence and future challenges. Ecol. Evol. 7 (3), 831–844.

Holmlund, C., Hammer, M., 1999. Ecosystem services generated by fish populations. Ecol. Econ. 29, 253–268.

Hooper, D.U., Vitousek, P.M., 1997. The effects of plant composition and diversity on ecosystem processes. Science 277, 1302–1305.

Hoyle, H., Hitchmough, J., Jorgensen, A., 2017. All about the 'wow factor'? The relationships between aesthetics, restorative effect and perceived biodiversity in designed urban planting. Landsc. Urban Plan. 164, 109–123.

Hutchinson, T.C., Stokes, P.M., 1975. Metal Toxicity and Algal Bioassays Water Quality Parameters. American Society for Testing and Materials, Philadelphia, Pa.

IPBES (2019) Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IUCN International Union for the Conservation of Nature (2015). European species under threat: Overview of European Red Lists results. URL (accessed July 2020): https://www.iucn.org/downloads/red_list_overview_new_1.pdf

Jacob S, Thomas AP, Manju EK (2017). Odonata (dragonflies and damselflies) as bio-indicators of water quality Int. J. Innov. Res. Sci. Eng. Technol. 6(9): 19464-74. DOI: 10.15680/IJIRSET.2017.0609144 19464

Kagalou, I., Latinopoulos, D., 2020. Filling the gap between ecosystem services concept and River Basin Management Plans: the case of Greece in WFD 20+. Sustainability 12 (18), 7710.

Kelly, M.G., Bennett, C., Coste, M., Delgado, C., Delmas, F., Denys, L., Ector, L., Fauville, C., Ferreol, M., Golub, M., Jarlman, A., Kahlert, M., Lucey, J., Ni Chathain, B., Pardo, I., Pfister, P., Picinska-Faltynowicz, J., Rosebery, J., Schranz, C., Schaumburg, J., van Dam, H., Vilbaste, S., 2009. A comparison of national approaches to setting ecological status boundaries in phytobenthos assessment for the European Water Framework Directive: results of an intercalibration exercise. Hydrobiologia 621, 169–182.

Kopf, R.K., Finlayson, C.M., Humphries, P., Sims, N.C., Hladyz, S., 2015. Anthropocene baselines: assessing change and managing biodiversity in human-dominated aquatic ecosystems. BioScience 65 (8), 798–811.
Kowalchuk, G.A., Bruinsma, M., van Veen, J.A., 2003. Assessing responses of soil microorganisms to GM plants. Trends Ecol. Evol. 18, 403–410.
Larras, F., Bouchez, A., Rimet, F., Montuelle, B., 2012. Using bioassays and species sensitivity distributions to assess herbicide toxicity towards benthic diatoms. PLoS One 7 (8), e44458.
Lavoie I, Campeau S, Darchambeau F et al. (2008). Are diatoms good integrators of temporal variability in stream water quality? Freshw. Biol. 53:827–841.
Lavorel, S., 2013. Plant functional effects on ecosystem services. J. Ecol. 101 (1), 4–8.
Lemelin, R., 2007. Finding beauty in the dragon: the role of dragonflies in recreation and tourism. J. Ecotour. 6, 139–145.
Leslie, A.W., Lamp, W.O., 2017. Taxonomic and functional group composition of macroinvertebrate assemblages in agricultural drainage ditches. Hydrobiologia 787, 99–110. https://doi.org/10.1007/s10750-016-2947-8.
Li J, Yu H, Luan Y (2015). Meta-analysis of the copper, zinc, and cadmium absorption capacities of aquatic plants in heavy metal-polluted water. Int J Environ Res Public Health 12(12): 14958–14973.
Luck, G.W., Harrington, R., Harrison, P.A., Kremen, C., et al., 2009. Quantifying the contribution of organisms to the provision of ecosystem services. BioScience 59 (3), 223–235.
Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., Condé, S., Somma, F., Orgiazzi, A., Jones, A., Zulian, A., Vallecilo, S., Petersen, J.E., Marquardt, D., Kovacevic, V., Abdul Malak, D., Marin, A.I., Czúcz, B., Mauri, A., Loffler, P., Bastrup-Birk, A., Biala, K., Christiansen, T., Werner, B., 2018. Mapping and Assessment of Ecosystems and their Services: An Analytical Framework for Ecosystem Condition. Publications office of the European Union, Luxembourg.
Maltby, L., van den Brink, P.J., Faber, J.H., Marshall, S., 2018. Advantages and challenges associated with implementing an ecosystem services approach to ecological risk assessment for chemicals. Sci. Total Environ. 621, 1342–1351.
McGinlay, J., Parsons, D.J., Morris, J., Hubatova, M., Graves, A., Bradbury, R.B., Bullock, J.M., 2017. Do charismatic species groups generate more cultural ecosystem service benefits? Ecosyst. Serv. 27, 15–24.
Moomaw, W.R., Chmura, G.L., Davies, G.T., et al., 2018. Wetlands in a changing climate: science, policy and management. Wetlands 38, 183–205. https://doi.org/10.1007/s13157-018-1023-8.
Nõges, P., van de Bund, W., Cardoso, A.C., Solimini, A.G., Heiskanen, A.S., 2015. Assessment of the ecological status of European surface waters: a work in progress. Hydrobiologia 633 (1), 197–211. https://doi.org/10.1007/s10750-009-9883-9.
Nyström, M., 2006. Redundancy and response diversity of functional groups: implications for the resilience of coral reefs. AMBIO J. Hum. Environ. 35 (1), 30–35.
Oliver, T.H., Heard, M.S., Isaac, N.J.B., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C.D.L., Petchey, O.L., Proença, V., Raffaelli, D., Suttle, K.B., Mace, G.M., Martín-López, B., Woodcock, B.A., Bullock, J.M., 2015. Biodiversity and resilience of ecosystem functions. Trends Ecol. Evol. 30 (11), 673–684. https://doi.org/10.1016/j.tree.2015.08.009.
ONS – Office for National Statistics (2017). UK natural capital: ecosystem accounts for freshwater, farmland and woodland. URL (accessed February 2021): https://www.ons.gov.uk/economy/environmentalaccounts/bulletins/uknaturalcapital/landandhabitatecosystemaccounts
Pienkowski T, Bickersteth S, Milner-Gulland EJ (2019). Evidencing links between biodiversity and health: a rapid review with a water quality case study. A report prepared for the Secretariat of the Rockefeller Foundation Economic Council on Planetary Health at the Oxford Martin School. URL (accessed September 2020): https://www.planetaryhealth.ox.ac.uk/wp-content/uploads/sites/7/2019/05/Biodiversity-and-Health-for-web-v2.pdf
Rai, P.K., 2009. Heavy metal phytoremediation from aquatic ecosystems with special reference to macrophytes. Crit. Rev. Environ. Sci. Technol. 39 (9), 697–753.
Ravenscroft, N., Church, A., 2011. The attitudes of recreational user representatives to pollution reduction and the implementation of the European Water Framework Directive. Land Use Pol. 28 (1), 167–174. https://doi.org/10.1016/j.landusepol.2010.05.009.
Reid, A.J., Carlson, A.K., Creed, I.F., Erika, J., Eliason, E.J., Gell, P.A., Johnson, P.T.J., Kidd, K.A., MacCormack, T.J., Olden, J.D., Ormerod, S.J., Smol, J.P., Taylor, W.W., Tockner, K., Vermaire, J.C., Dudgeon, D., Cooke, S.J., 2019. Emerging threats and persistent conservation challenges for freshwater biodiversity. Biol. Rev. 94, 849–873.
Reyjol, Y., Argillier, C., Bonne, W., Borja, A., Buijse, A.D., Cardoso, A.C., Daufresne, M., Kernan, M., et al., 2014. Assessing the ecological status in the context of the European water framework directive: where do we go now? Sci. Total Environ. 497–498, 332–344.
Roberts, K., Granum, E., Leegood, R.C., Raven, J.A., 2007. Carbon acquisition by diatoms. Photosynth. Res. 93, 79–88.
Sabater, et al., 2002. The effect of biological factors on the efficiency of river biofilms in improving water quality. Hydrobiologia 469 (1), 149–156.
Salt, D.E., 1995. MgATP-dependent transport of phytochelatins across the tonoplast of oat roots. Plant Physiol. 107, 1293–1301.
Sánchez-Bayo, F., Wyckhuys, K.A.G., 2019. Worldwide decline of the entomofauna: a review of its drivers. Biol. Conserv. 232, 8–27.
SCHER, SCENIHR, SCCS (2011). Scientific Committee on Health and Environmental Risks SCHER Scientific Committee on Emerging and Newly Identified Health Risks SCENIHR, Scientific Committee on Consumer Safety SCCS. Toxicity and Assessment of Chemical Mixtures. doi:https://doi.org/10.2772/21444
Schmera, D., Heino, J., Podani, J., Erös, T., Dolédec, S., 2017. Functional diversity: a review of methodology and current knowledge in freshwater macroinvertebrate research. Hydrobiologia 787, 27–44.
Sela, M., Gary, J., Tel-Or, E., 1989. Accumulation and the effect of heavy metals on the water fern *Azolla filiculoides*. New Phytol. 112, 7–12.
Smith, A.C., Harrison, P.A., Soba, M.P., et al., 2017. How natural capital delivers ecosystem services: a typology derived from a systematic review. Ecosyst. Serv. 26, 111–126.
Spray C, Blackstock K (2013). Optimising Water Framework Directive River Basin Management Planning Using an Ecosystem Services Approach, CD2012_17.
Tilman, D., Reich, P.B., Knops, J.M., 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature 441 (7093), 629–632.
Tweedley, J.R., Warwick, R.M., Hallett, C.S., Potter, I.C., 2017. Fish-based indicators of estuarine condition that do not require reference data. Estuar. Coast. Shelf Sci. 191, 209–220.
UK EA Environment Agency (2013). Water Appraisal Guidance; Assessing Costs and Benefits for River Basin Management Planning. Final draft report, May 2013. URL (accessed December 2019): http://www.ecrr.org/Portals/27/Publications/Water%20Appraisal%20Guidance.pdf
UK NEA (2014) The UK National Ecosystem Assessment: synthesis of the Key Findings. UNEP WCMC, LWEC, UK. URL (accessed September 2020): http://randd.defra.gov.uk/Document.aspx?Document=12099_01.UKNEAFOSynthesis.pdf
UK TAG (2008). River Assessment Methods - Benthic Invertebrate Fauna, River Invertebrate Classification Tool (RICT) URL (accessed December 2019): http://www.wfduk.org/sites/default/files/Media/Characterisation%20of%20the%20water%20environment/Biological%20Method%20Statements/river%20invertebrates.pdf
UK TAG (2013). Development and use of the zinc bioavailability assessment tool (Draft). Published by Water Framework Directive – United Kingdom Technical Advisory Group. URL (accessed October 2020): https://wfduk.org/sites/default/files/Media/Zinc%20bioavailability%20assessment%20tool%20-%20UKTAG%20DRAFT_0.pdf
UK TAG (2014a). UK TAG River Assessment Method Macrophytes and Phytobenthos, Phytobenthos – Diatoms for Assessing River and Lake Ecological Quality (River DARLEQ2) by Water Framework Directive – United Kingdom Technical Advisory Group (WFD-UKTAG). https://www.wfduk.org/sites/default/files/Media/Characterisation%20of%20the%20water%20environment/Biological%20Method%20Statements/River%20Phytobenthos%20UKTAG%20Method%20Statement.pdf URL (accessed December 2019).
UK TAG (2014b). UK TAG Guide to ecological indicators of severe water resources pressures in rivers. Water Framework Directive – United Kingdom Technical Advisory Group. URL (accessed October 2020): https://www.wfduk.org/sites/default/files/Media/Characterisation%20of%20the%20water%20environment/Biological%20Method%20Statements/Ecological%20Indicators%20of%20severe%20flow%20pressures%20UKTAG%20Method%20Statement.pdf
UK TAG (2014c). River & Lake Assessment Method Specific Pollutants (Metals): Metal Bioavailability Assessment Tool (M-BAT). Water Framework Directive – United Kingdom Technical Advisory Group. URL (accessed December 2019): www.wfduk.org
UK TAG (2019). Annex 6 – RIVERS – Fish – FCS2 and Fish Barrier Tool. URL (accessed December 2019): https://www.wfduk.org/sites/default/files/Media/Environmental%20standards/Annex%206%20Rivers%20Fish%20FCS2%20%26%20Fish%20Barrier.pdf
Valiente-Banuet, A., Aizen, M.A., Alcantara, J.M., Arroyo, J., Cocucci, A., Galetti, M., Garcıa, M.B., Garcia, D., Gomez, J.M., Jordano, P., Medel, R., Navarro, L., Obeso, J.R., Oviedo, R., Ramırez, N., Rey, P.J., Traveset, A., Verdu, M., Zamora, R., 2015. Beyond species loss: The extinction of ecological interactions in a changing world. Funct. Ecol. 29, 299–307. https://doi.org/10.1111/1365-2435.12356.
Vallecillo, S., La Notte, A., Ferrini, S., Maes, J., 2019. How ecosystem services are changing: an accounting application at the EU level. Ecosyst. Serv. 40, 101044. https://doi.org/10.1016/j.ecoser.2019.101044.
Vidal-Abarca, M.R., Santos-Martín, F., Martín-López, B., Sánchez-Montoya, M.M., Suárez Alonso, M.L., 2016. Exploring the capacity of water framework directive indices to assess ecosystem services in fluvial and riparian systems: towards a second implementation phase. Environ. Manag. https://doi.org/10.1007/s00267-016-0674-6.
Vlachopoulou, M., Coughlin, D., Forrow, D., Kirk, S., Logan, P., Voulvoulis, N., 2014. The potential of using the ecosystem approach in the implementation of the EU Water Framework Directive. Sci. Total Environ. 470–471, 684–694.
Voulvoulis, N., Arpon, K.D., Giakoumis, T., 2017. The EU Water Framework Directive: from great expectations to problems with implementation. Sci. Total Environ. 575, 358–366.
Whittingham, M.J., 2011. The future of agri-environment schemes: biodiversity gains and ecosystem service delivery? J. Appl. Ecol. 48 (3), 509–513.
Willby, N.J., Abernethy, V.J., Demars, B.O.L., 2000. Attribute-based classification of European hydrophytes and its relationship to habitat utilization. Freshw. Biol. 43, 43–74.
Willby NJ, Pitt J-A, Phillips GL (2012). The ecological classification of UK rivers using aquatic macrophytes. Science Report SC010080/R1. Environment Agency, Bristol. URL (accessed December 2019): https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/291478/LIT_7379_8fe63b.pdf
Winfree, R., Fox, J.W., Williams, N.M., Reilly, J.R., Cariveau, D.P., 2015. Abundance of common species, not species richness, drives delivery of a real-world ecosystem service. Ecol. Lett. 18, 626–635.
WWF, Grooten, M., Almond, R.E.A., 2018. Living Planet Report - 2018: Aiming Higher. WWF, Gland, Switzerland.
Ziv, G., Mullin, K., Boeuf, B., Fincham, W., Taylor, N., Villalobos-Jiménez, G., et al., 2018. Water quality is a poor predictor of recreational hotspots in England. PLoS One 11 (11), e0166950. https://doi.org/10.1371/journal.pone.016695.