



A sediment-specific family-level biomonitoring tool to identify the impacts of fine sediment in temperate rivers and streams



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ABSTRACT

Anthropogenic modifications of sediment load can cause ecological degradation in stream and river ecosystems. However, in practice, identifying when and where sediment is the primary cause of ecological degradation is a challenging task. Biological communities undergo natural cycles and variation over time, and respond to a range of physical, chemical and biological pressures. Furthermore, fine sediments are commonly associated with numerous other pressures that are likely to influence aquatic biota. The use of conventional, non-biological monitoring to attribute cause and effect would necessitate measurement of multiple parameters, at sufficient temporal resolution, and for a significant period of time. Biomonitoring tools, which use low-frequency measurements of biota to gauge and track changes in the environment, can provide a valuable alternative means to detecting the effects of a given pressure. In this study, we develop and test an improved macroinvertebrate, family-level and mixed-level biomonitoring tool for fine sediment. Biologically-based classifications of sediment sensitivity were supplemented by using empirical data of macroinvertebrate abundance and percentage fine sediment, collected across a wide range of temperate river and stream ecosystems (model training dataset $n = 2252$) to assign detailed individual sensitivity weights to taxa. An optimum set of weights were identified by non-linear optimisation, as those that resulted in the highest Spearman's rank correlation coefficient between the index (called the Empirically-weighted Proportion of Sediment-sensitive Invertebrates index; E-PSI) scores and deposited fine sediment in the model training dataset. The family and mixed-level tools performed similarly, with correlations with percentage fine sediment in the test dataset ($n = 84$) of $r_s = -0.72$ and $r_s = -0.70$ $p < 0.01$. Testing of the best performing family level version, over agriculturally impacted sites ($n = 754$) showed similar correlations to fine sediment ($r_s = -0.68$ $p < 0.01$). The tools developed in this study have retained their biological basis, are easily integrated into contemporary monitoring agency protocols and can be applied retrospectively to historic datasets. Given the challenges of non-biological conventional monitoring of fine sediments and determining the biological relevance of the resulting data, a sediment-specific biomonitoring approach is highly desirable and will be a useful addition to the suite of pressure-specific biomonitoring tools currently used to infer the causes of ecological degradation.

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1. Introduction

Streambed deposited fine sediment (<2 mm) is an important, natural component of freshwater ecosystems and is critical for habitat heterogeneity and ecological functioning (Owens et al., 2005; Wood and Armitage, 1997; Yarnell et al., 2006). However, anthropogenic activities can alter sediment delivery and dynam-

ics contributing to ecological degradation (Vörösmarty et al., 2003; Walling and Fang, 2003). Fine sediment can directly impact on stream biota by subjecting them to abrasion, scour or burial; by clogging gills or feeding appendages; by limiting light penetration; as well as indirectly by introducing toxic contaminants sorbed to the surface of fine sediment particles, and reducing oxygen concentrations in the substrate (reviewed in Bilotta and Brazier, 2008). In practice, identifying when and where sediment is the primary cause of ecological degradation is a challenging task. Biological communities undergo natural cycles and variation over time, and respond to other physical (e.g. changes to flow and temperature),

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chemical (e.g. pH, heavy metals, pesticides, nutrients) and biological pressures (e.g. invasive species), (Clews and Ormerod, 2009; Liess and Schulz, 1999; Moore and Ramamoorthy, 2012; Sousa, 1984; Townsend, 1996). The use of conventional (non-biological) monitoring approaches to attribute cause and effect necessitates measurement of multiple parameters, at sufficient temporal resolution, and for a significant period of time (Grove et al., 2015; Harris and Heathwaite, 2012). Biomonitoring tools, which use changes in the presence, abundance or behaviour of biota to indicate, gauge and track changes in the environment (Friberg et al., 2011; Gerhardt, 2000; Wright et al., 1993), can provide a valuable lower-cost alternative to conventional monitoring. Ideally, for reasons discussed in more detail in the following paragraphs, biomonitoring tools should, where possible: (i) have a biological basis, (ii) be easily integrated into standardised biological sampling and recording methods, (iii) be developed and tested over sites from the full range of river and stream ecosystems to which they are intended to be applied, and (iv) have a strong relationship with the pressure of concern.

1.1. Biological basis

There is an emerging consensus among those involved in developing biomonitoring tools, that more reliable tools have a biological basis, i.e. they use biological and ecological traits that influence the tolerance of organisms to a given pressure, and are linked to ecological niche theory (Bonada et al., 2006; Friberg et al., 2011). One of the reasons for this is that these types of tools are not vulnerable to statistical artefacts that may affect purely statistical models. The biological traits (e.g. respiration, locomotion, dispersal, feeding) of taxa in a community reflect the spatial and temporal variations in the environmental factors of a habitat, that act as “filters”, with successful combinations of traits enabling survival and reproduction (Poff, 1997; Statzner et al., 2001b; Townsend et al., 1997). Certain traits or combinations of traits can result in sensitivities/tolerances to particular environmental pressures and therefore these traits have the potential to be used in biomonitoring to discriminate between types of human disturbance (Statzner et al., 2001a). If taxa are selected for inclusion and their sensitivities weighted based on the biological and ecological traits, which influence their sensitivity to a given pressure, the resultant biomonitoring tool will have a mechanistic linkage (rather than a purely correlative linkage) between the pressure of concern and the biotic response (Friberg, 2014).

1.2. Easily integrated into standardised biological sampling and recording methods

One of the many benefits of biomonitoring as opposed to conventional monitoring of multiple environmental parameters is that the biological data collected from the same biological sample can be interpreted by an array of different biomonitoring tools to identify potential pressures. In order for this efficiency and cost-effectiveness to be realised, the biomonitoring tools must be able to make use of data collected using standardised biological sampling methods and recorded to standardised, minimum taxonomic levels. Monitoring agencies in different countries commonly carry out invertebrate identification and recording at different taxonomic levels. For example, within Europe the level of invertebrate taxonomic knowledge varies between countries, as do resources, resulting in different taxonomic resolutions and a lack of comparability between data (Hering et al., 2010; Schmidt-Kloiber and Hering, 2015). Within the UK, England and Wales recently moved

from family level invertebrate recording to a mixed level,¹ consisting of family-, genus- and species-level identifications, with the majority of taxa being identified to species (Davy-Bowker et al., 2010). However, Scotland and Northern Ireland currently record at family-level.² A family-level tool is not only essential for those countries that record at this taxonomic level, but is also crucial for those countries which have recently switched to a higher taxonomic resolution, as it will allow for retrospective analysis of historic family-level data. This ability to assess the historic conditions at sites is particularly important when attempting to distinguish between natural temporal variations (Resh et al., 2005) and those caused by anthropogenic activities, as this can require numerous years of baseline data. Furthermore, a lower-cost family-level biomonitoring tool is likely to be highly desirable for non-regulatory work (e.g. river restoration projects, aquatic research and citizen science programmes) where budgets and taxonomic expertise may be more limited.

1.3. Developed and tested over the full range of different rivers and streams

There are a number of reasons why biomonitoring tools should be developed and tested using data from sites across the full spectrum of rivers and streams to which they are designed to be applied. Firstly, biological communities and species distributions vary naturally, partly as a result of environmental gradients, biotic interactions (McGill et al., 2006) and their ecological requirements (Schmidt-Kloiber and Hering, 2015). As such, a biomonitoring tool must incorporate a range of taxa to ensure that any sampled site will have the potential to include a sufficient number of taxa with sensitivity weightings. Secondly, it cannot be assumed that biological communities will respond uniformly to the same pressure in different rivers and streams. Not only may the pressure of concern occur alongside other, different pressures (potentially having synergistic or antagonistic effects), (Folt et al., 1999), but differences in environmental characteristics (e.g. habitat complexity) may also affect a biological community's resistance and resilience to a specific pressure (Dunbar et al., 2010a,b; Lake, 2000).

1.4. Strong relationship with pressure of interest

Given the implications of incorrect assignment of ecological status of streams for both water and land managers (from unjustified burdens being placed on the users of water resources, to environmental damage going undetected), it is important that biomonitoring tools have a strong relationship with their pressure, in addition to the previous three criteria. Nevertheless, a strong correlation does not rule out the possibility of the tool indicating other aquatic pressures or variables that may occur in parallel with the pressure of concern. A statistical approach may yield strong correlations to the pressure of concern, but these can be the result of statistical artefacts e.g. an inadvertent relationship with an associated pressure (Table 1). Although a biological basis provides a mechanistic linkage for a correlation, biological traits are not always unique to a particular pressure, and as such the influence of confounding pressures also cannot be ruled out (Schuwirth et al., 2015).

¹ This mixed level identification is referred to as TL5 by the Environment Agency.

² This family level identification is referred to as TL2 by the Environment Agency.

Table 1
Confounding pressures commonly associated with fine sediment in rivers and streams globally, and their effect on macroinvertebrates.

Associated pressure/contaminant	Details	Country of study	Authors	Effect on macroinvertebrates
Physical Flow	Discharge, velocity	Austria, France, New Zealand, UK	Lefrançois et al. (2007), Matthaei et al. (2010), Petticrew et al. (2007), Slattery and Burt (1997), Tockner et al. (1999), Wood and Armitage (1999)	Decrease/increase in abundance, dependent on the ecological requirements of macroinvertebrates (Extence et al., 1999).
Chemical Nutrients	Nitrogen, phosphorus	China, Korea, New Zealand, UK, USA	Brazier et al. (2007), Carpenter et al. (1998), Jarvie et al. (2006), Kim et al. (2003), Owens et al. (2007), Owens and Walling (2002), Piggott et al. (2015), Sun et al. (2009), Wagenhoff et al. (2011)	Hypoxic conditions following eutropication can reduce the abundance of species with high oxygen requirements. Inputs of nitrogenous compounds, can also lead to ammonia, nitrite and nitrate toxicity (reviewed in Camargo and Alonso, 2006).
Pesticides	Polychlorinated biphenyls, pyrethroid, organochlorines, organophosphate	Australia, Brazil, China, India, Italy, Vietnam, UK, USA	Camusso et al. (2002), McKenzie-Smith et al. (1994), Minh et al. (2007), Torres et al. (2002), Warren et al. (2003), Weston et al. (2004), Zhang et al. (2003)	Various pesticides have been linked to declines in both abundance and richness, through direct toxicity (reviewed in Wijngaarden et al., 2005), which is also likely to influence predator/prey interactions.
Metals	Aluminium, cadmium, chromium, copper, iron, lead, manganese, mercury, zinc	Italy, Pakistan, Turkey, UK, USA	Abernathy et al. (1984), Akcay et al. (2003), Camusso et al. (2002), Dawson and Macklin (1998), Owens et al. (2001), Tariq et al. (1996), Walling et al. (2003)	Responses to metal pollution are varied, but for intolerant groups (e.g. Ephemeroptera), abundance and richness are typically reduced in impacted streams (reviewed in Clements, 1991).
Organic matter		Austria, Luxemburg, Taiwan, UK	Tockner et al. (1999), Von Bertrab et al. (2013), Yu et al. (2001), Glendell and Brazier. (2014b)	Excessive inputs of organic matter and the subsequent biological decomposition, can lead to reduced oxygen levels, impacting those species with high oxygen requirements (e.g. <i>Leuctra</i> sp.), whilst favouring those with low oxygen requirements (e.g. <i>Chironomus</i> sp.), (Friberg et al., 2010).
Other	Polycyclic aromatic hydrocarbons	Canada, China, Malaysia, Taiwan, UK, USA	Ashley and Baker (1999), Doong and Lin (2004), Stevens et al. (2003), Yunker et al. (2002), Zakaria et al. (2002), Zhang et al. (2004)	Polycyclic aromatic hydrocarbons are toxic to many macroinvertebrates, and can lead to reduced richness and abundance (Ankley et al., 1994)
Biological Bacteria/pathogens	<i>Escherichia coli</i> , <i>Salmonella</i> , <i>Cryptosporidium</i>	Austria, Canada, New Zealand, USA	Droppo et al. (2009), Jamieson et al. (2005), Kernegger et al. (2009), Mallin (2000), Muirhead et al. (2004)	The effects of the many sediment associated pathogens on macroinvertebrates is unclear. <i>Cryptosporidium</i> and <i>Giardia</i> accumulate within some macroinvertebrates, but with uncertain consequences (Reboredo-Fernandez et al., 2015).

1.5. The Proportion of Sediment-sensitive Invertebrates (PSI) index

The PSI index is a biomonitoring tool that is designed to identify the degree of sedimentation in rivers and streams (Extence et al., 2011). Invertebrate sensitivity ratings (A-Highly sensitive, B-Moderately sensitive, C-Moderately insensitive, D-Highly insensitive) were assigned following an extensive review of the literature, and using expert knowledge of ecological and biological traits, thus providing the tool with a sound biological basis. Traits that may result in sensitivity to fine sediment include feeding, locomotion and respiratory attributes. For example, scrapers/grazers

and passive filter feeders may experience decreased food availability or damage to feeding appendages as a result of deposited fine sediment (Larsen and Ormerod, 2010; Nerbonne and Vondracek, 2001). The PSI index has both family-level and species-level versions, allowing it to be easily integrated into standardised biological sampling and recording methods. However, an evaluation of these versions of the index across a wide range of temperate river and stream ecosystems (Turley et al., 2014), showed it to have moderate correlations with fine sediment ($r_s = -0.61$, $p < 0.01$ and $r_s = -0.64$, $p < 0.01$ respectively), that although comparable with the average invertebrate-based, pressure-specific tool, used throughout the EU (Birk et al., 2012), limits confidence in its application. Recent work

has demonstrated enhancements to the species-level PSI index through the use of empirical data to supplement the expert knowledge and literature, which were used to determine the original four fine sediment sensitivity ratings (Turley et al., 2015).

The aim of the present study was to investigate if similar empirical-weighting techniques could be used to enhance the performance of family- and mixed-level versions of the PSI index, whilst retaining their biological basis, and ensuring that the tools are compatible with historic and contemporary datasets recorded to family-level and/or mixed-level, and have a strong correlation with fine sediment.

2. Methods

2.1. Data

2.1.1. Minimally-impacted sites

It was important that empirical data from minimally-impacted sites were used in this study, in order to reduce confounding pressures in the process of assigning sensitivity weightings. In this instance, a freely available dataset from the UK was utilised. However, this study can be seen as an exemplar to be used in other countries where such datasets exist. The RIVPACS IV (May 2011 version) dataset (River Invertebrate Prediction and Classification System—NERC [CEH] 2006. Database rights NERC [CEH] 2006 all rights reserved) is described in detail by Wright (2000) and Clarke et al. (2003), but is summarised here. The database contains invertebrate, water quality and catchment characteristics data, recorded at each site over at least one year, between 1978 and 2004. The 835 reference condition sites, on streams and rivers across the United Kingdom (Fig. 1), encompass a wide range of environments, varying in their (i) climate – mean annual precipitation totals between 1961 and 1990 of 430–2930 mm and mean annual temperatures between 1961 and 1990 ranging from 7.93 to 11.45 °C, (ii) geology – varying from catchments dominated by hard igneous rocks to catchments dominated by soft sedimentary rocks and (iii) topography – altitudes at river source varying from 5 to 1216 m above sea level. The stream and river sites also vary in their morphometry with widths ranging from 0.4 to 117 m and average depths ranging from 0.02 to 3.00 m (widths and depths are a mean of three seasonal measurements). All of the sites are considered to be as close to reference condition as it is possible in the United Kingdom, and they have no, or only very minor, anthropogenic alterations to the values of the chemistry and hydromorphology, supporting biota usually associated with such minimally-disturbed conditions.

The 835 sites were sampled for macroinvertebrates using the UK standard method; a standardised three-minute kick sample technique using a 900 µm mesh hand net, followed by a one-minute hand search. All in-stream habitats identified at the site were sampled in proportion to their occurrence (Environment Agency, 2009). The database has abundance records of different macroinvertebrates identified to (i) family level, (ii) mixed level, and (iii) species level (Wright, 2000). Each site has season-specific records of community composition: spring (March–May), summer (June–August) and autumn (September–November). As the technique used for macroinvertebrate sampling was semi-quantitative, model development and testing utilised log abundance data rather than raw abundance data or presence/absence data. This recognises the potential importance of changes in abundance when attempting to identify pressure gradients, without placing spurious confidence in the semi-quantitative sampling technique (Furse et al., 1981).

The fine sediment data within the RIVPACS IV database consisted of visual estimates of the percentage of the substrate cover composed of silt and clay (<0.06 mm) and sand (≥0.06 and <2.00 mm), as an annual average based on three seasonal measure-

ments. The visual estimate method used to collect these data is described in the River Habitat Survey Field Survey Guidance Manual (Environment Agency, 2003). Briefly, it involves the operator carrying out a visual inspection over a given reach, estimating the substrate composition and recording the percentage of bedrock, boulders and cobbles, pebbles and gravel, sand, and silt and clay.

Family and mixed level macroinvertebrate data, and site substrate cover were extracted from the RIVPACS database and compiled in Microsoft Excel. The percentage of the substrate cover consisting of sand, silt and clay, were combined to provide an overall estimate of fine sediment (<2 mm) for each of the 835 minimally-impacted sites. The dataset was then split into a training dataset (751 sites, three seasons, $n=2252$) and test dataset (84 sites, autumn season only, $n=84$) using random allocation, to ensure testing of the indices could be considered independent of the development stages. Similar to Kelly et al. (2012), this 90:10 split was chosen to ensure that the indices were developed using a significant number of sites, whilst leaving a suitable number for independent testing.

2.1.2. Agriculturally impacted sites

The family level indices were also tested across a continuum of agriculturally impacted stream sites (upstream catchment land-use ranging from 0 to 90% arable fields, $n=754$) in England, using part of a dataset described by Pearson et al. (2016). In brief, this dataset included seasonal macroinvertebrate data, seasonal fine sediment data, and water chemistry data (mean of the preceding 12 months) that were collected during routine monitoring by the Environment Agency in 2006. For the present study, more detailed macroinvertebrate abundance data were obtained from the Environment Agency, for samples collected in the autumn, along with fine sediment data based on the mean of spring and autumn visual estimates, and Environmental Quality Ratios (based on the observed and expected scores—calculated using RICT) for an index designed to indicate the effects of organic pollution (WHPT index, Paisley et al., 2014). All 754 sites used in the present study were located on independent water bodies, as detailed by the water body names and grid references provided by the Environment Agency.

2.2. Developing the Empirically-weighted PSI (E-PSI) indices

For both the family level and mixed level E-PSI indices, the *fmincon* function (active-set algorithm), a nonlinear optimisation method of MATLAB (Mathworks, version R2014a), was used to test up to 100,000 iterations of taxon sensitivity weightings, to find an optimum set of weightings that resulted in the strongest Spearman's rank correlation coefficient between index scores and fine sediment in the training dataset. The Spearman's rank correlation was used, as the data were found to be non-normally distributed and show heteroscedasticity, and could not be successfully transformed. In order to maintain the biological basis of the original PSI index, those taxa with PSI ratings of “highly sensitive” and “moderately sensitive” were constrained to sensitivity weights between 0.50 and 1.0, and those rated as “moderately insensitive” and “highly insensitive” between 0 and 0.49. Using a range of weights acknowledges the breadth of ecological niches that macroinvertebrates occupy, as well as their differing potential as indicators. A modified version of the original PSI index's equation was used to calculate E-PSI scores:

$$E-PSI = \frac{\sum (\log A_{sens} \times W)}{\sum (\log A_{all} \times W)} \times 100 \quad (1)$$

Formula used to calculate E-PSI scores. The sum of each sensitive taxon's log abundance ($\log A_{sens}$), multiplied by the corresponding sensitivity weightings (W), is divided by the sum of all taxon log abundances ($\log A_{all}$) multiplied by the corresponding sensitivity

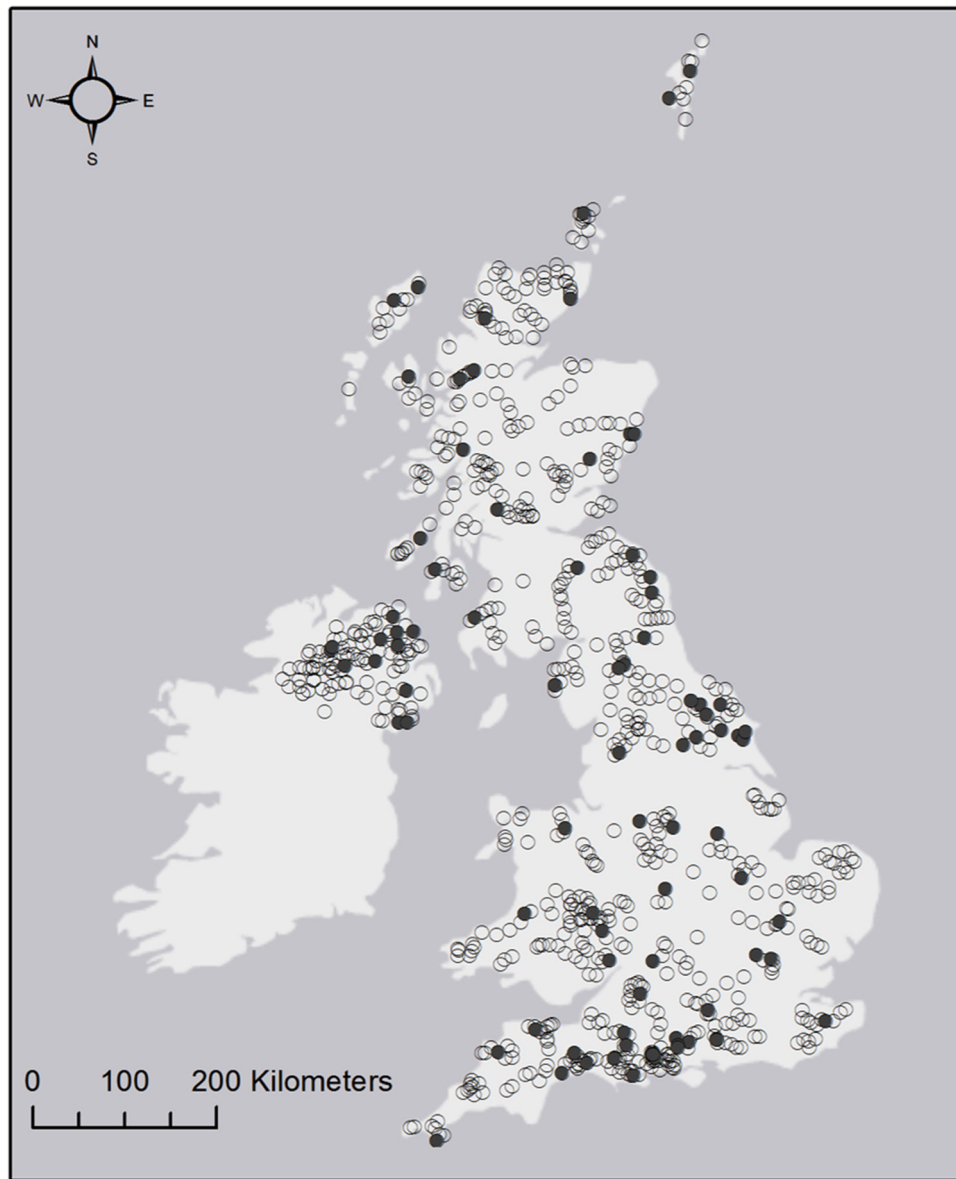


Fig. 1. Distribution of minimally-impacted sites throughout the UK. Light dots are those sites that formed the training dataset ($n = 751$), dark dots are those sites that formed the test dataset ($n = 84$).

weightings (W). This value is then multiplied by 100 to provide the E-PSI score. *Note:* Log abundance categories in E-PSI were simplified to: 1–9 individuals present = 1; 10–99 = 2; 100–999 = 3; 1000+ = 4. E-PSI scores range from 0 to 100; 0 representing a site dominated by fine sediment, 100 representing a site with minimal amounts of fine sediment.

2.2.1. Developing the family level E-PSI index

In total, 79 invertebrate families with PSI sensitivity weightings were present in the family level training dataset. Following the non-linear optimisation procedure, an optimum set of family sensitivity weights formed the E-PSI_{fam} index.

2.2.2. Adjustments for family-level sensitivity variations

In the process of developing the original, family-level PSI index, inevitably some generalisations were made regarding the sensitivities of invertebrate families to fine sediment. Families can comprise numerous genera and species, each with a range of different environmental requirements and sensitivities (Lenat and Resh, 2001), making assignment of sensitivities at family-level a difficult pro-

cess. To determine whether all families identified as sensitive by the original PSI index were correctly identified as useful indicators at this taxonomic level, the empirical data was investigated. Using data from all 835 sites within the RIVPACS dataset, the percentage of fine sediment that corresponded with the 75th percentile of the family's total abundance was calculated for each taxon. This involved using raw abundance invertebrate data, sorting all 2504 samples in order of increasing fine sediment, and calculating the cumulative abundance for each taxon. The 75th percentile of abundance was then calculated for each family, and this value or the next highest value was cross-referenced to obtain the corresponding sediment value for that abundance. This family-specific sediment value, represented the maximum fine sediment percentage at which at least 75% of the families abundance occurred. Similarly to Relyea et al. (2012), this 75% value was used as it recognises the fact that taxa can occur in sub-optimal conditions, often at reduced densities. These family-specific fine sediment values were used to inform the creation of a modified version of the family-level E-PSI index, by excluding taxa whose sediment value was $\geq 33\%$. The families that exceeded this threshold were removed from the index

altogether (instead of being reassigned to the insensitive category) to maintain the biological basis. The removal of these families is not an admission of their tolerance to fine sediment, rather it acknowledges that when considering fine sediment at the reach scale, they are not useful as indicator taxa. The 33% threshold was used as previous research showed that at the patch scale, sediment cover of approximately one third resulted in significant declines in invertebrate richness (Larsen et al., 2009). Once these taxa had been removed from the index, the nonlinear optimisation procedure was performed to return a new set of optimum sensitivity weights based on the new altered community composition/taxon list, forming the E-PSI_{fam69} index

2.2.3. Developing the mixed level E-PSI index

In addition to the family level index, a mixed level version was developed using the same procedures. In total, 355 invertebrate taxa with PSI sensitivity weightings were present in the mixed level training dataset. Following the nonlinear optimisation procedure outlined in Section 2.2, an optimum set of sensitivity weights formed the E-PSI_{mixed} index.

2.3. Testing of the E-PSI indices

The developed E-PSI indices were tested using both the minimally-impacted, independent dataset ($n=84$), as well as the agriculturally impacted dataset ($n=754$), both described in Section 2.1. The performance of these indices were evaluated and compared using their Spearman's rank correlations with fine sediment, due to the non-normally distributed and heteroscedastic data, and the inability to successfully transform. All correlations were interpreted using the Dancy and Reidy (2007) classifications of correlations; 0.1–0.39 = weak, 0.4–0.69 = moderate, 0.7–0.99 = strong.

In order to examine the relationship between E-PSI and fine sediment further, a separate test was conducted on the minimally-impacted (RIVPACS) sites with discharge data ($n=443$). The aim of this exercise was to restrict the potential collinearities between fine sediment and covariates relating to longitudinal gradients, particularly those relating to flow (e.g. discharge, velocity and slope). To achieve this, the sites were grouped by stream power (total stream power per unit stream length, $W m^{-1}$) into five approximately equal groups (see Supplementary material B). The Spearman's rank correlations within each group between the E-PSI indices and fine sediment were then calculated.

Stream power (Ω) was calculated using the following formula:

$$\Omega = \gamma Qs$$

where γ is the specific weight of water ($9810 N m^{-3}$), Q is the average water discharge ($m^3 s^{-1}$), and s is energy slope ($m m^{-1}$, which can be approximated by the river slope), (Knighton, 1999).

2.4. Comparisons with other biomonitoring tools

The Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence et al., 1999) and the Combined Fine Sediment Index (CoFSI) (Murphy et al., 2015) were calculated and their relationship to both fine sediment, PSI and E-PSI indices were assessed in terms of their Spearman's rank correlation coefficients. LIFE was chosen to determine the influence of known interactions between flow and fine sediment deposition and re-suspension (Dewson et al., 2007). Correlations between LIFE, E-PSI indices and fine sediment were also compared within stream power groups.

The recently developed, predominantly species- and genus-level CoFSI was included in this analysis, as it is a purely statistically based index designed to indicate fine sediment pressures (Murphy

Table 2

Spearman's rank correlation coefficients (2 d.p.) between versions of PSI, E-PSI, LIFE, CoFSI and fine sediment, in the minimally-impacted test dataset, and the agriculturally impacted test dataset.

Index ^a	No. of sensitive taxa	Minimally-impacted ($n=84$)	Agriculturally impacted ($n=754$)
Family level			
PSI _{fam}	36	−0.59	−0.66
E-PSI _{fam}	36	−0.66	−0.69
E-PSI _{fam69}	26	−0.72	−0.68
LIFE _{fam}	n/a	−0.57	−0.62
Mixed level			
PSI _{mixed}	139	−0.60	n/a
E-PSI _{mixed}	139	−0.70	n/a
CoFSI	n/a	−0.72	n/a
LIFE _{mixed}	n/a	−0.51	n/a

^a The subscripted text identifies the taxonomic level (family or mixed) of the data and the number of taxa used in the calculations of the index. All correlations are significant at the 0.01 level (2-tailed). Mixed level indices could not be tested over the agriculturally impacted sites, as data at this taxonomic level were not recorded.

et al., 2015). All mixed-level indices were calculated using data of the same taxonomic detail, at the level used for Water Framework Directive reporting.

3. Results

3.1. Biological indices and fine sediment

The best performing family-level and mixed-level E-PSI indices were strongly, negatively correlated to fine sediment metrics in the minimally-impacted dataset and similarly, but moderately correlated in the agriculturally impacted test dataset (Table 2). The agriculturally impacted test dataset had WHPT scores of 2.5–8.0 and Environmental Quality Ratios of between 0.4 and 1.3 (see Supplementary material C) indicating that the sites were, at the very least, impacted to varying degrees by organic pollution.

The E-PSI indices with the strongest correlation with fine sediment in the minimally-impacted test dataset were the E-PSI_{fam69} and E-PSI_{mixed} ($r_s = -0.72$ and $r_s = -0.70$, $p < 0.01$, respectively). The optimisation process was responsible for improving the correlations between fine sediment and the E-PSI_{fam} index compared to the PSI_{fam} index, in the minimally-impacted test dataset ($r_s = -0.66$, $p < 0.01$ compared to $r_s = -0.59$, $p < 0.01$). The largest improvement following the optimisation process was for the E-PSI_{mixed} index in the minimally-impacted test dataset, which represented an improvement of 10 percentage points ($r_s = -0.70$, $p < 0.01$ compared to $r_s = -0.60$, $p < 0.01$). LIFE_{fam}, and LIFE_{mixed} had moderate correlations with fine sediment ($r_s = -0.57$ and $r_s = -0.51$, $p < 0.01$, respectively) and CoFSI had a strong correlation with fine sediment ($r_s = -0.72$, $p < 0.01$).

Within the stream power groups (Table 3), the strongest correlation between E-PSI_{fam69} and fine sediment was $r_s = 0.71$, $p < 0.01$ in the group of sites with low stream power, with moderate correlations in all other groups (ranging from $r_s = 0.59$ to $r_s = 0.68$, $p < 0.01$).

3.2. Adjustments for family-level sensitivity variations

By including only those taxa whose corresponding sediment value was $<33\%$, the strength of the correlation between the E-PSI_{fam} index and fine sediment was improved in the unimpacted test dataset, but had essentially the same correlation over the impacted sites. The exclusion process resulted in 10 families being removed entirely from the calculation of index scores. These sensitive families had corresponding sediment values of between 36% and 83% fine sediment. Three of these families had taxa rated as

Table 3

Spearman's rank correlation coefficients between the best performing E-PSI indices, LIFE and fine sediment, in the minimally-impacted dataset, using autumn data, and all sites with discharge data ($n = 443$). Sites are grouped by total stream power per unit length (W m^{-1}).

Stream power group ^a	E-PSI _{fam69}	E-PSI _{mixed}	LIFE _{fam}	LIFE _{mixed}	E-PSI _{fam69}	E-PSI _{mixed}
	Vs fine sediment				Vs LIFE _{mixed}	
All sites ($n = 443$)	-0.73	-0.72	-0.58	-0.56	0.80	0.84
Group 1 ($0.25\text{--}10.59 \text{ W m}^{-1}$)	-0.71	-0.73	-0.71	-0.72	0.92	0.96
Group 2 ($10.93\text{--}29.34 \text{ W m}^{-1}$)	-0.59	-0.56	-0.46	-0.51	0.84	0.89
Group 3 ($29.41\text{--}67.01 \text{ W m}^{-1}$)	-0.63	-0.65	-0.49	-0.53	0.81	0.85
Group 4 ($67.92\text{--}167.95 \text{ W m}^{-1}$)	-0.65	-0.63	-0.39	-0.32	0.65	0.68
Group 5 ($168.34\text{--}6935.42 \text{ W m}^{-1}$)	-0.68	-0.65	-0.44	-0.19 (ns)	0.47	0.55

^a Stream power groups 1 and 5 ($n = 88$), stream power groups 2,3 and 4 ($n = 89$). All correlations are significant at the 0.01 level (2-tailed), unless otherwise indicated, (ns) not statistically significant.

insensitive in the mixed level tool: Limnephilidae, Scirtidae and Gammaridae (for further information see [Appendix A](#)).

3.3. Assigned sediment-sensitivity weightings

The sensitivity weightings that formed the best performing E-PSI indices (E-PSI_{fam69} and E-PSI_{mixed}) in the minimally-impacted test dataset are shown in [Appendix B and C](#). An index calculator is also provided in Supplementary material A.

3.4. Restricting the collinearities between fine sediment and longitudinal gradients

The Spearman's rank correlations between E-PSI and fine sediment within each of the five stream power groups were moderate to strong ($r_s = -0.56$ to $r_s = -0.73$) ([Table 3](#)).

3.5. Comparisons between biomonitoring tools

In addition to comparing the E-PSI indices to fine sediment, the relationships between indices were considered. The various indices had strong correlations with each other, ranging from $r_s = 0.79$ to $r_s = 0.98$, $p < 0.01$ ([Table 4](#)). All versions of the E-PSI index had weaker correlations with LIFE_{fam} and LIFE_{mixed}, compared to the original versions of the PSI index.

The correlations between E-PSI and LIFE decreased in successively increasing stream power groups ([Table 3](#)), from $r_s = 0.92$ to $r_s = 0.47$, $p < 0.01$ for E-PSI_{fam69} versus LIFE_{mixed}. The correlation between LIFE_{mixed} and fine sediment was strongest ($r_s = 0.72$, $p < 0.01$) in the group of sites with low stream power, and was weakest at sites with high stream power.

4. Discussion

4.1. Identifying fine sediment pressures

The results of this study show that the use of empirical data to assign sensitivity weightings within the PSI index's original biologically-based sensitivity ratings, has improved the performance of the tools, in terms of their correlation with fine sediment. As a result, both the family-level and mixed-level E-PSI indices have correlation coefficients with fine sediment, that are stronger than the average invertebrate-based, pressure-specific biomonitoring tool used throughout Europe in the implementation of the WFD ([Birk et al., 2012](#)). Furthermore, by including only those taxa whose 75th percentile of abundance corresponded with a fine sediment value of $< 33\%$, the E-PSI_{fam69} index has a strong correlation with fine sediment similar to that of the mixed-level E-PSI index (E-PSI_{fam69} versus fine sediment: $r_s = -0.72$, $p < 0.01$). Applying this threshold to the family-level index was intended to acknowledge the likelihood of significant within family variation in terms of sensitivity to fine sediment, and identify any families that were

not useful as indicators at the reach scale. This exercise appears to have had the desired effect, reducing some of the variation in the relationship that may have been caused by varying sensitivities within families. Testing of the E-PSI_{fam69} index also showed similar improvements and correlation ($r_s = -0.68$, $p < 0.01$) over the agriculturally impacted sites, suggesting that the index performs similarly over sites that are impacted by the multitude of pressures associated with agricultural land-use.

Nevertheless, correlations do not prove causality and they should be interpreted with some caution, partly because many in-stream influences are associated with each other in riverine environments ([Allan, 2004](#)). The benefit of biologically-based indices such as the PSI and E-PSI index is that the biological basis provides a mechanistic linkage between index scores and fine sediment conditions. The PSI index utilises existing knowledge on biological and ecological traits and sensitivities to fine sediment, to assign sensitivity ratings, which the E-PSI index combines with empirical data to assign more detailed sensitivity weightings (0–1).

Based on the available data on the ecological preferences of macroinvertebrates from [Schmidt-Kloiber and Hering \(2015\)](#), those families, which have been assigned the highest sensitivity weightings in E-PSI_{fam69} (Heptageniidae, Perlidae, Aphelocheiridae, Chloroperlidae, Lepidostomatidae, Leuctridae, and Perlodidae), are dominated by species with a strong affinity to coarse substrate microhabitats, with the exception of Lepidostomatidae that have some affinity to coarse substrates, but are more often associated with woody debris. Habitat preferences are determined by an invertebrates biological traits or combination of traits. These preferences relate not only to the direct physical properties of the habitat (e.g. the substrate), but also the flow velocities, hydrological and thermal regimes, resource availability and biotic interactions associated with them. These taxa with the greatest E-PSI sensitivity weightings are mainly clingers, with tarsal claws allowing them to “grip” the surface of pebbles, cobbles and boulders in shearing flows ([Pollard and Yuan, 2010](#)). Nevertheless, they depend on the heterogeneity of coarse substrates to provide refugia from these shearing flows, as well as from predators. Previous research has shown reduced densities and richness of clingers, with increasing fine sediment cover ([Pollard and Yuan, 2010](#); [Rabeni et al., 2005](#)). Additionally, the turbulence caused by coarse substrate and the resulting oxygenation is likely to be important for many of these sensitive families. For example, *Aphelocheirus aestivalis* is a plastron-breather and so requires well-oxygenated waters ([Seymour et al., 2015](#)), and fine sediment may also disrupt its respiratory functioning.

4.2. Confounding pressures

A considerable challenge for research carried out over large spatial extents is the occurrence of collinearity between the variable of interest and other environmental variables ([Pearson et al., 2016](#)). Relatively few studies have considered the interactions and co-occurrence of in-stream pressures, which are likely to affect the

Table 4
Spearman's rank correlation coefficients (2 d.p.) between biological indices, in the minimally-impacted, autumn test dataset ($n=84$).

Index ^a	PSI _{fam}	E-PSI _{fam}	E-PSI _{fam69}	LIFE _{fam}	PSI _{mixed}	E-PSI _{mixed}	CoFSI _{mixed}
Family level							
PSI _{fam}		0.95	0.93	0.92	0.96	0.93	0.87
E-PSI _{fam}	0.95		0.98	0.85	0.94	0.96	0.93
E-PSI _{fam69}	0.93	0.98		0.85	0.91	0.96	0.94
LIFE _{fam}	0.92	0.85	0.85		0.89	0.86	0.79
Mixed level							
PSI _{mixed}	0.96	0.94	0.91	0.89		0.95	0.88
E-PSI _{mixed}	0.93	0.96	0.96	0.86	0.95		0.92
CoFSI _{mixed}	0.87	0.93	0.94	0.79	0.88	0.92	
LIFE _{mixed}	0.88	0.83	0.79	0.85	0.91	0.85	0.80

^a The subscripted text identifies the taxonomic level (family or mixed) of the data and the number of taxa used in the calculations of the index. All correlations are significant at the 0.01 level (2-tailed).

performance of biomonitoring tools. A recent study of 9330 sites throughout 14 European countries, found 47% of rivers (90% of lowland rivers) throughout Europe to be subject to multiple pressures relating to hydrology, morphology, water quality and connectivity (Schneegger et al., 2012). This highlights the importance of a mechanistic linkage between indices and sediment conditions, given the potential for these multiple pressures to confound biomonitoring approaches.

In the present study, sites were grouped using site-specific stream power to restrict the collinearities between fine sediment and longitudinal gradients, particularly those relating to flow. Our results show that flow (indicated by LIFE) was related to fine sediment cover, and as such LIFE and E-PSI are strongly correlated. However, this relationship between fine sediment cover and flow weakens as stream power increases, likely because after the critical thresholds for particle entrainment and transport have been exceeded, further increases in stream power cease to result in much further entrainment, transport and removal of fine sediment from the river bed (though the increases in stream power may trigger entrainment and transport of coarser particles as bed load). This is likely to be the reason why the relationship between LIFE scores and E-PSI, and LIFE and fine sediment weaken in successively increasing stream power groups. These results add confidence to our interpretation that although fine sediment cover is related to stream power (for physical reasons), and there will therefore be a moderate to strong relationship between scores derived from sediment-specific and flow-specific biomonitoring tools; these relationships are not fixed, and thus where one pressure (e.g. flow) is stable, whilst the other is variable (e.g. sediment cover), the scores derived from the biomonitoring tools will diverge and provide valuable information on the likely cause of deviation from reference-condition community composition. In the case of LIFE and E-PSI, a similar approach to model development for LIFE could result in even greater decoupling of the two tools.

4.3. Statistically based sediment-specific tools

As noted, other studies have attempted to develop fine sediment-specific indices, using empirical data and statistical methods to select taxa for inclusion and to assign sensitivity weights to these taxa, to achieve a correlation between a tool's score and the pressure of concern. These approaches have achieved moderate-strong correlations with fine sediment but lack a biological basis and mechanistic linkage, which are important for disentangling the multiple pressures in rivers and streams (Friberg, 2014). Murphy et al. (2015) developed CoFSI, an index that endeavours to take account of both the amount and organic content of fine sediment. The authors used partial canonical correspondence analysis to rank taxa in terms of their sensitivity to fine sediment, based on empirical data from 179 stream sites. The re-suspension method of quantifying fine sediment was used, averaging samples from two erosional and two depositional patches to achieve a contemporane-

ous reach-scale average. Despite this technique necessitating some subjective assessment of "erosional" and "depositional" patches, moderate to strong correlations ($r_s = -0.54$ to -0.70 , $p < 0.05$) were observed between the resultant sediment data and CoFSI in their test dataset (Murphy et al., 2015). Due to the difficulties of selecting sites with minimal confounding pressures that at the same time represent a gradient of sediment pressures, their resulting test dataset ($n=83$) is somewhat geographically restricted and is focused on agricultural streams, which may be impacted by an array of different pressures typically associated with agricultural practices (Allan, 2004; Matthaehi et al., 2010; Weston et al., 2004). In the present study, CoFSI is shown to have a similar correlation with fine sediment compared to the best performing family and mixed-level E-PSI indices. CoFSI is a mixed-level (predominantly genus- and species-level) biomonitoring tool, and as such cannot be applied to present and historic datasets recorded at family-level.

Most recently, in North America, Hubler et al. (2016) developed a statistically based index for Oregon streams (the Biological Sediment Tolerance Index; BSTI) using weighted averaging to assign tolerance values, and found a moderate correlation ($r^2 = 0.58$) between the index and fine sediment in their independent test dataset ($n=50$). Similarly, Zweig and Rabeni (2001) found a moderate correlation ($r_s = -0.59$, $P < 0.01$) between their Deposited Sediment Biotic Index (DSBI) and visual estimates of fine sediment, using data from four streams in Missouri. The authors used the sediment values that corresponded with each taxon's median abundance, to assign tolerance ratings, and used the same data for testing, therefore lacking an independent test dataset. In addition, the DSBI is currently limited by the small development dataset, which is likely to restrict its spatial applicability. A further example from North America is the Fine Sediment Biotic Index (FSBI), which again used empirical data to assign sensitivity weights to taxa (Releya et al., 2012). The FSBI showed a large amount of variation, with the data forming a wedge-shaped response that has since been suggested as a typical response to multiple pressures (Friberg 2014); when sediment pressure is low, it has minimal impact on the invertebrate community and other pressures dominate. This wedge-shaped response is also observed in PSI, E-PSI and CoFSI (see Supplementary material D), supporting this generalisation. Some of the variation in these relationships is also inevitably associated with invertebrate sampling and quantifying fine sediment pressures. For example the standardised, three minute kick-sample method has been shown to collect only approximately 50% of the species and 60% of the families present in six replicate samples (Furse et al., 1981). The challenges of quantifying fine sediment pressures in a way that is biologically relevant and accurately represents spatial variation is a further limitation to the development and/or testing of sediment-specific biomonitoring tools. Visual estimates of substrate composition were used in this study due to the data being from an existing database, which had the benefits of a wide geographic coverage. Although the technique is subjective

(Clapcott et al., 2011) it provides a measure of fine sediment surface drape, which is likely to directly influence macroinvertebrate communities and as such is likely to be a biologically relevant metric (Conroy et al., 2016), relating to niche theory and habitat suitability (Hirzel and Le Lay 2008) over the entire reach.

4.4. Taxonomic resolution

In contrast to previous findings (Relyea et al., 2012; Turley et al., 2014) the taxonomic resolution had little effect on the indices correlations with fine sediment; E-PSI_{fam69} and E-PSI_{mixed} had similar strong correlations with fine sediment. Relyea et al. (2012) found that sensitivities varied within invertebrate families and concluded that family level was insufficient for pressure-specific tools. This within-family variation of sensitivities to fine sediment was expected to result in large differences in terms of the correlation between family and mixed level indices and fine sediment. Despite the similar correlations observed, it is likely that the mixed level tool will provide more diagnostic information, particularly at more impacted sites. The taxonomic sufficiency is a key consideration when developing an invertebrate-based biomonitoring tool, as the taxonomic resolution can affect the accuracy of predictions (Schmidt-Kloiber and Nijboer, 2004) as well as the costs associated with processing biotic samples (Jones, 2008; Marshall et al., 2006). The higher the taxonomic resolution, the more resource intensive biomonitoring becomes (identification requires more time and expertise/training). For macroinvertebrates, some studies have shown that family level is often sufficiently able to detect environmental change (Marshall et al., 2006; Mueller et al., 2013), whilst others conclude that a higher resolution is preferential, particularly for genera-rich families or when attempting to identify subtle environmental changes (Monk et al., 2012; Waite et al., 2004). Furthermore, although our knowledge of species level biology is incomplete, intra-generic environmental requirements have been shown to vary significantly for various families of macroinvertebrates (reviewed in Lenat and Resh, 2001). In terms of a pressure-specific biomonitoring tool, the optimum taxonomic resolution in terms of model predictions is likely to be dependent on the scale or gradient of environmental pressure, but will inevitably be a compromise between the costs associated with high taxonomic resolution and the ecological information lost at lower resolutions. Nevertheless, the development of new pressure-specific tools should ideally align with national/international monitoring agencies, to ensure that they can be applied to existing protocols and historical data.

In addition, as the E-PSI index is not a purely statistically based tool, it is able to include relatively rare taxa (low abundances or small distribution range), which are often removed in the development stages of statistically based indices (Murphy et al., 2015; Relyea et al., 2012). These rare taxa, specifically those with small distribution ranges, may be important indicators of subtle environmental change as they are likely to be more specialised, having a narrower range of optimal conditions, and as such may be the first taxa to indicate the effects of a pressure (Cao et al., 1998; Schmidt-Kloiber and Nijboer, 2004). Whilst these assumptions are intuitive, there is some debate as to the suitability of rare taxa in multivariate analyses particularly relating to their redundancy and their influence on observed versus expected scores (Cao et al., 2001; Marchant 2002; Van Sickle et al., 2007). Furthermore, when conservation issues become a concern for these rare species, their collection should be carefully considered. The E-PSI index only uses taxa that are collected in routine samples by the monitoring agencies, and as such makes the most use of existing data. The River Invertebrate Classification Tool (RICT) used in the UK to derive expected index scores for sites, includes rare species, calculating scores based on their probability of occurrence (Clarke et al., 2003).

4.5. Application of the E-PSI index

In application, most indices should be used within an Ecological Quality Index (EQI) framework, based on observed versus expected invertebrate community composition (Glendell et al., 2014a), particularly when being used for regulatory purposes. This requires a predictive model that is independent of the pressure of concern. At present, in the UK, the predictive model “RICT” (previously RIVPACS IV) is not independent of fine sediment, as substrate composition is a predictor variable (Clarke et al., 2011). Furthermore, it is over-simplistic to assume that deviation of observed from expected community composition (and therefore index score) can be explained by a single measurement at one point in time. Aside from the fact that the structure of macroinvertebrate communities is determined not simply by the contemporaneous fine sediment, but by local sediment dynamics spanning the preceding days, weeks and months, there are multiple reasons for this not being a simple cause-effect relationship. Firstly, fine sediment has numerous direct and indirect effects on invertebrates (reviewed in Bilotta and Brazier, 2008), many of which are poorly understood. Secondly, not only are there often dissociated pressures present, but studies have also shown a number of common sediment-associated physical, chemical and biological pressures (Table 1) that depend on catchment land use, in-stream influences, geomorphology, hydrology and so on. For example, fine sediment pressure is commonly associated with stream flow, due to the influence of stream velocity on the transport, deposition and re-suspension of fine sediment (Wood and Armitage, 1997). Additionally, depending on the sorptive properties of the sediment (i.e. organic carbon content, particle size, clay content and cation exchange capacity) hydrophobic contaminants often become associated via adsorption or absorption, potentially acting as confounding pressures (Warren et al., 2003). Lastly, in disturbed sites, the biological communities have often been subjected to this range of pressures over an extended period of time and therefore the observed biological community composition at any point in time is partly reflective of the legacy of these pressures (Allan, 2004).

The common occurrence of diverse pressures in rivers and streams, places a greater importance on a mechanistic linkage and understanding of biotic response to the pressure of concern. As a result, it is likely that most pressure-specific indices will be limited in terms of their ability to make accurate predictions of the degree of impact across all river and stream ecosystems. Instead, a more achievable and realistic goal is to use a suite of different pressure-specific indices to identify potential areas of ecological degradation and the likely causes of any deviations. The family and mixed level E-PSI indices are two such tools, which could be used alongside other indices to inform more targeted monitoring and mitigation measures.

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Appendix A.

Table A1

Sensitive macroinvertebrate families removed from E-PSI calculations, based on a 33% threshold being applied to their corresponding sediment values.

Removed sensitive families	Number of sensitive taxa in E-PSI _{mixed}	Number of insensitive taxa in E-PSI _{mixed}	Corresponding sediment value (%)
Sisyridae	0	0	83
Piscicolidae	1	0	73
Limnephilidae	5	15	46
Beraeidae	3	0	45
Scirtidae	1	1	45
Dixidae	n/a	n/a	42
Goeridae	3	0	38
Gammaridae	2	3	37
Polycentropodidae	7	0	37
Psychomyiidae	5	0	36

Appendix B.

Table B1

Sensitivity weights (*W*) for the family-level, Empirically-weighted Proportion of Sediment-sensitive Invertebrates (E-PSI) index.

Sensitive taxa	<i>W</i>
Heptageniidae	1
Perlidae	1
Aphelocheiridae	0.99
Chloroperlidae	0.99
Lepidostomatidae	0.99
Leuctridae	0.99
Perlodidae	0.99
Baetidae	0.75
Brachycentridae	0.75
Capniidae	0.75
Hydraenidae	0.75
Hydropsychidae	0.75
Taeniopterygidae	0.75
Planorbidae (genus <i>Ancylus</i> only)	0.51
Glossosomatidae	0.51
Leptophlebiidae	0.51
Rhyacophilidae	0.51
Elmidae	0.5
Ephemerellidae	0.5
Niphargidae	0.5
Odontoceridae	0.5
Philopotamidae	0.5
Potamanthidae	0.5
Sericostomatidae	0.5
Simuliidae	0.5
Tipulidae, Limoniidae, Cylindrotomidae and Pediciidae	0.5
Insensitive taxa	<i>W</i>
Aeshnidae	0.49
Bithyniidae	0.49
Calopterygidae	0.49
Cordulegastridae	0.49
Corixidae	0.49
Ephemeridae	0.49
Hydrophilidae, Helophoridae, Georissidae and Hydrochidae	0.49
Libellulidae	0.49
Noteridae	0.49

Table B1 (Continued)

Insensitive taxa	<i>W</i>
Stratiomyidae	0.49
Halplidae	0.48
Hydrobiidae	0.48
Nepidae	0.48
Neritidae	0.48
Phryganeidae	0.48
Platycnemididae	0.48
Ptychopteridae	0.48
Sialidae	0.48
Sphaeriidae	0.48
Syrphidae	0.48
Tabanidae	0.48
Unionidae	0.48
Crangonyctidae	0.25
Dytiscidae	0.25
Physidae	0.25
Planariidae	0.25
Siphonuridae, Ameletidae	0.25
Valvatidae	0.25
Asellidae	0.24
Glossiphoniidae	0.24
Viviparidae	0.24
Dendrocoelidae	0.01
Dryopidae	0.01
Erpobdellidae	0.01
Hirudinidae	0.01
Lymnaeidae	0.01
Molannidae	0.01
Caenidae	0
Corophiidae	0
Dugesidae	0
Nemouridae	0
Planorbidae (excluding genus <i>Ancylus</i>)	0
Psychodidae	0

Appendix C.

Table C1

Sensitivity weights (*W*) for the mixed-level, Empirically-weighted Proportion of Sediment-sensitive Invertebrates (E-PSI) index.

Sensitive taxa	<i>W</i>
<i>Amphinemura sulcicollis</i>	(Stephens, 1836) 1
<i>Apatania muliebris</i>	(McLachlan, 1866) 1
<i>Baetis rhodani</i>	(Pictet, 1843–1845) 1
<i>Baetis scambus</i> group	1
<i>Brachycentrus subnubilus</i>	(Curtis, 1834) 1
<i>Brachyptera putata</i>	(Newman, 1838) 1
<i>Ceraclea albimaculata</i>	(Rambur, 1842) 1
<i>Ceraclea annulicornis</i>	(Stephens, 1836) 1
<i>Cheumatopsyche lepida</i>	(Pictet, 1834) 1
<i>Chimarra marginata</i>	(Linnaeus, 1761) 1
<i>Chloroperla tripunctata</i>	(Scopoli, 1763) 1
<i>Crenobia alpina</i>	(Dana, 1766) 1
<i>Dinocras cephalotes</i>	(Curtis, 1827) 1
<i>Diplectrona felix</i>	(McLachlan, 1878) 1
<i>Ecclisopteryx guttulata</i>	(Pictet, 1834) 1
<i>Ecdyonurus</i> sp.	1
<i>Electrogena lateralis</i>	(Curtis, 1834) 1
<i>Glossosoma</i> sp.	1
<i>Hydraena gracilis</i>	(Germar, 1824) 1
<i>Hydraena nigrita</i>	(Germar, 1824) 1
<i>Hydroporus ferrugineus</i>	(Stephens, 1829) 1
<i>Hydropsyche contubernalis</i>	(McLachlan, 1865) 1
<i>Hydropsyche pellucidula</i>	(Curtis, 1834) 1
<i>Isoperla grammatica</i>	(Poda, 1761) 1
<i>Lepidostoma hirtum</i>	(Fabricius, 1775) 1
<i>Leuctra fusca</i>	(Linnaeus, 1758) 1

Table C1 (Continued)

Sensitive taxa		W
<i>Leuctra hippopus</i>	(Kempny, 1899)	1
<i>Leuctra inermis</i>	(Kempny, 1899)	1
<i>Leuctra moselyi</i>	(Morton, 1929)	1
<i>Macronychus quadrituberculatus</i>	(Müller, 1806)	1
<i>Neureclipsis bimaculata</i>	(Linnaeus, 1758)	1
<i>Nigrobaetis digitatus</i>	(Bengtsson, 1912)	1
<i>Normandia nitens</i>	(Müller, 1817)	1
<i>Oecetis notata</i>	(Rambur, 1842)	1
<i>Oreodytes davisii</i>	(Curtis, 1831)	1
<i>Oreodytes septentrionalis</i>	(Gyllenhal, 1826)	1
<i>Perla bipunctata</i>	(Pictet, 1833)	1
<i>Perlodes microcephalus</i>	(Pictet, 1833)	1
<i>Plectrocnemia geniculata</i>	(McLachlan, 1871)	1
<i>Polycentropus flavomaculatus</i>	(Pictet, 1834)	1
<i>Polycentropus kingi</i>	(McLachlan, 1881)	1
<i>Protonemura praecox</i>	(Morton, 1894)	1
<i>Psychomyia pusilla</i>	(Fabricius, 1781)	1
<i>Rhithrogena</i> sp.		1
<i>Siphonoperla torrentium</i>	(Pictet, 1841)	1
<i>Wormaldia</i> sp.		1
<i>Alainites muticus</i>	(Linnaeus, 1758)	0.88
<i>Heptagenia sulphurea</i>	(Müller, 1776)	0.88
<i>Hydropsyche siltalai</i>	(Döhler, 1963)	0.88
<i>Protonemura meyeri</i>	(Pictet, 1841)	0.88
<i>Rhyacophila dorsalis</i>	(Curtis, 1834)	0.88
<i>Beraea maurus</i>	(Curtis, 1834)	0.87
<i>Beraeodes minutus</i>	(Linnaeus, 1761)	0.87
<i>Ephemerella notata</i>	(Eaton, 1887)	0.87
<i>Orectochilus villosus</i>	(O.F. Müller, 1776)	0.87
<i>Procloeon pennulatum</i>	(Eaton, 1870)	0.87
<i>Ceraclea senilis</i>	(Burmeister, 1839)	0.75
<i>Leuctra geniculata</i>	(Stephens, 1836)	0.63
<i>Oecetis testacea</i>	(Curtis, 1834)	0.63
<i>Paraleptophlebia submarginata</i>	(Stephens, 1835)	0.63
Tipulidae		0.63
<i>Melampophylax mucoreus</i>	(Hagen, 1861)	0.62
<i>Agabus guttatus</i>	(Paykull, 1798)	0.5
<i>Agapetus</i> sp.		0.5
<i>Allotrichia pallicornis</i>	(Eaton, 1873)	0.5
<i>Ameletus inopinatus</i>	(Eaton, 1887)	0.5
<i>Ancyclus fluvialis</i>	(O.F. Müller, 1774)	0.5
<i>Aphelocheirus aestivalis</i>	(Fabricius, 1794)	0.5
<i>Athripsodes albifrons</i>	(Linnaeus, 1758)	0.5
<i>Athripsodes bilineatus</i>	(Linnaeus, 1758)	0.5
<i>Athripsodes cinereus</i>	(Curtis, 1834)	0.5
<i>Athripsodes commutatus</i>	(Rostock, 1874)	0.5
<i>Baetis buceratus</i>	(Eaton, 1870)	0.5
<i>Baetis vernus</i>	(Curtis, 1834)	0.5
<i>Beraea pullata</i>	(Curtis, 1834)	0.5
<i>Brachyptera risi</i>	(Morton, 1896)	0.5
<i>Calopteryx virgo</i>	(Linnaeus, 1758)	0.5
<i>Capnia atra</i>	(Morton, 1896)	0.5
<i>Capnia bifrons</i>	(Newman, 1839)	0.5
<i>Ceraclea dissimilis</i>	(Stephens, 1836)	0.5
<i>Ceraclea fulva</i>	(Rambur, 1842)	0.5
<i>Ceraclea nigronervosa</i>	(Retzius, 1783)	0.5
<i>Crunoecia irrorata</i>	(Curtis, 1834)	0.5
<i>Cynurus trimaculatus</i>	(Curtis, 1834)	0.5
<i>Deronectes latus</i>	(Stephens, 1829)	0.5
<i>Diura bicaudata</i>	(Linnaeus, 1758)	0.5
Dixidae		0.5
<i>Drusus annulatus</i>	(Stephens, 1837)	0.5
<i>Elmis aenea</i>	(Müller, 1806)	0.5
<i>Elodes</i> sp.		0.5
<i>Gammarus duebeni</i>	(Liljeborg, 1852)	0.5
<i>Gammarus pulex</i>	(Linnaeus, 1758)	0.5
<i>Goera pilosa</i>	(Fabricius, 1775)	0.5
<i>Hydraena riparia</i>	(Kugelann, 1794)	0.5
<i>Hydraena rufipes</i>	(Curtis, 1830)	0.5
<i>Hydropsyche angustipennis</i>	(Curtis, 1834)	0.5

Table C1 (Continued)

Sensitive taxa		W
<i>Hydropsyche fulvipes</i>	(Curtis, 1834)	0.5
<i>Hydropsyche instabilis</i>	(Curtis, 1834)	0.5
<i>Hydropsyche saxonica</i>	(McLachlan, 1884)	0.5
<i>Labiobaetis atrebatinus</i>	(Eaton, 1870)	0.5
<i>Lepidostoma basale</i>	(Kolenati, 1848)	0.5
<i>Leuctra nigra</i>	(Olivier, 1811)	0.5
<i>Limnius volckmari</i>	(Panzer, 1793)	0.5
Limoniidae		0.5
<i>Margaritifera margaritifera</i>	(Linnaeus, 1758)	0.5
<i>Metalype fragilis</i>	(Pictet, 1834)	0.5
<i>Micronecta</i> sp.		0.5
Micropterna group		0.5
<i>Nemoura cambrica</i> group		0.5
<i>Nigrobaetis niger</i>	(Linnaeus, 1761)	0.5
<i>Niphargus aquilex</i>	(Schiodte, 1855)	0.5
<i>Odontocerum albicorne</i>	(Scopoli, 1763)	0.5
<i>Oreodytes sanmarkii</i>	(C.R. Sahlberg, 1826)	0.5
<i>Paraleptophlebia cincta</i>	(Retzius, 1835)	0.5
Pediciidae		0.5
<i>Phagocata vitta</i>	(Duges, 1830)	0.5
<i>Philopotamus montanus</i>	(Donovan, 1813)	0.5
<i>Piscicola geometra</i>	(Linnaeus, 1761)	0.5
<i>Platambus maculatus</i>	(Linnaeus, 1758)	0.5
<i>Plectrocnemia conspersa</i>	(Curtis, 1834)	0.5
<i>Polycentropus irroratus</i>	(Curtis, 1835)	0.5
<i>Potamanthus luteus</i>	(Linnaeus, 1767)	0.5
<i>Potamophylax</i> group		0.5
<i>Protonemura montana</i>	(Kimmins, 1941)	0.5
<i>Rhyacophila fasciata</i>	(Hagen, 1859)	0.5
<i>Rhyacophila munda</i>	(McLachlan, 1862)	0.5
<i>Rhyacophila obliterata</i>	(McLachlan, 1863)	0.5
<i>Riolus cupreus</i>	(Müller, 1806)	0.5
<i>Riolus subviolaceus</i>	(Müller, 1817)	0.5
<i>Sericostoma personatum</i>	(Spence in Kirby and Spence, 1826)	0.5
<i>Serratella ignita</i>	(Poda, 1761)	0.5
<i>Sialis fuliginosa</i>	(Pictet, 1836)	0.5
<i>Silo nigricornis</i>	(Pictet, 1834)	0.5
<i>Silo pallipes</i>	(Fabricius, 1781)	0.5
Simuliidae		0.5
<i>Stictonectes lepidus</i>	(Olivier, 1795)	0.5
<i>Tinodes dives</i>	(Pictet, 1834)	0.5
<i>Tinodes unicolor</i>	(Pictet, 1834)	0.5
<i>Tinodes waeneri</i>	(Linnaeus, 1758)	0.5
Insensitive taxa		W
<i>Aeshna</i> sp.		0.49
<i>Agabus didymus</i>	(Olivier, 1795)	0.49
<i>Agabus paludosus</i>	(Fabricius, 1801)	0.49
<i>Alboglossiphonia heteroclita</i>	(Linnaeus, 1761)	0.49
<i>Anabolia nervosa</i>	(Curtis, 1834)	0.49
<i>Anacaena globulus</i>	(Paykull, 1829)	0.49
<i>Anisus (Anisus) leucostoma</i>	(Millet, 1813)	0.49
<i>Anisus (Disculifer) vortex</i>	(Linnaeus, 1758)	0.49
<i>Bdellocephala punctata</i>	(Pallas, 1774)	0.49
<i>Bithynia (Bithynia) tentaculata</i>	(Linnaeus, 1758)	0.49
<i>Bithynia (Codiella) leachii</i>	(Sheppard, 1823)	0.49
<i>Brychius elevatus</i>	(Panzer, 1793)	0.49
<i>Caenis horaria</i>	(Linnaeus, 1758)	0.49
<i>Caenis robusta</i>	(Eaton, 1884)	0.49
<i>Callicorixa praeusta</i>	(Fieber, 1848)	0.49
<i>Calopteryx splendens</i>	(Harris, 1782)	0.49
<i>Centroptilum luteolum</i>	(Müller, 1776)	0.49
<i>Cloeon dipterum</i>	(Linnaeus, 1761)	0.49
<i>Cloeon simile</i>	(Eaton, 1870)	0.49
<i>Cordulegaster boltonii</i>	(Donovan, 1807)	0.49
<i>Corixa dentipes</i>	(Thomson, 1869)	0.49
<i>Corixa punctata</i>	(Illiger, 1807)	0.49
<i>Corophium</i> sp.		0.49
<i>Dytiscus semisulcatus</i>	(O.F. Müller, 1776)	0.49

Table C1 (Continued)

Insensitive taxa		W
<i>Ephemera danica</i>	(Müller, 1764)	0.49
<i>Erpobdella testacea</i>	(Savigny, 1812)	0.49
<i>Galba truncatula</i>	(O.F. Müller, 1774)	0.49
<i>Glyptotaelius pellucidus</i>	(Retzius, 1783)	0.49
<i>Gomphus vulgatissimus</i>	(Linnaeus, 1758)	0.49
<i>Graptodytes pictus</i>	(Fabricius, 1787)	0.49
<i>Habrophlebia fusca</i>	(Curtis, 1834)	0.49
<i>Haemopsis sanguisuga</i>	(Linnaeus, 1758)	0.49
<i>Halesus</i> sp.		0.49
<i>Haliplus flavicollis</i>	(Sturm, 1834)	0.49
<i>Haliplus fluviatilis</i>	(Aubé, 1836)	0.49
<i>Haliplus ruficollis</i>	(DeGeer, 1774)	0.49
<i>Haliplus sibericus</i>	(Motschulsky, 1860)	0.49
<i>Helophorus (Helophorus) minutus</i>	(Fabricius, 1775)	0.49
<i>Helophorus (Meghelophorus) aequalis</i>	(Thomson, 1868)	0.49
<i>Helophorus (Meghelophorus) grandis</i>	(Illiger, 1798)	0.49
<i>Helophorus (Rhopalohelophorus) brevipalpis</i>	(Bedel, 1881)	0.49
<i>Hesperocorixa linnaei</i>	(Fieber, 1848)	0.49
<i>Hesperocorixa sahlbergi</i>	(Fieber, 1848)	0.49
<i>Hydatophylax infumatus</i>	(McLachlan, 1865)	0.49
<i>Hydraena pulchella</i>	(Germar, 1824)	0.49
<i>Hydraena testacea</i>	(Curtis, 1831)	0.49
<i>Hydrobius fuscipes</i>	(Linnaeus, 1758)	0.49
<i>Hydroporus memnonius</i>	(Nicolai, 1822)	0.49
<i>Hydroporus pubescens</i>	(Gyllenhal, 1808)	0.49
<i>Hydroporus tessellatus</i>	(Drapiez, 1819)	0.49
<i>Hygrotus (Hygrotus) inaequalis</i>	(Fabricius, 1777)	0.49
<i>Hygrotus (Hygrotus) versicolor</i>	(Schaller, 1783)	0.49
<i>Hyphyrus ovatus</i>	(Linnaeus, 1761)	0.49
<i>Ilybius</i> sp.		0.49
<i>Kageronia fuscogrisea</i>	(Retzius, 1783)	0.49
<i>Laccobius (Laccobius) minutus</i>	(Linnaeus, 1758)	0.49
<i>Limnebius nitidus</i>	(Marsham, 1802)	0.49
<i>Limnephilus decipiens</i>	(Kolenati, 1848)	0.49
<i>Limnephilus extricatus</i>	(McLachlan, 1865)	0.49
<i>Limnephilus flavicornis</i>	(Fabricius, 1787)	0.49
<i>Limnephilus lunatus</i>	(Curtis, 1834)	0.49
<i>Limnephilus rhombicus</i>	(Linnaeus, 1758)	0.49
<i>Lymnaea stagnalis</i>	(Linnaeus, 1758)	0.49
<i>Molanna angustata</i>	(Curtis, 1834)	0.49
<i>Mystacides longicornis</i>	(Linnaeus, 1758)	0.49
<i>Nebrioporus depressus</i>	(Fabricius, 1775)	0.49
<i>Nemoura cinerea</i>	(Retzius, 1783)	0.49
<i>Nemurella pictetii</i>	(Klapálek, 1900)	0.49
<i>Noterus clavicornis</i>	(DeGeer, 1774)	0.49
<i>Notidobia ciliaris</i>	(Linnaeus, 1761)	0.49
<i>Ochthebius dilatatus</i>	(Stephens, 1829)	0.49
<i>Ochthebius minimus</i>	(Fabricius, 1792)	0.49
<i>Oecetis lacustris</i>	(Pictet, 1834)	0.49
<i>Oecetis ochracea</i>	(Curtis, 1825)	0.49
<i>Oulimnius rivularis</i>	(Rosenhauer, 1856)	0.49
<i>Paraleptophlebia werneri</i>	(Ulmer, 1919)	0.49
<i>Phryganea</i> sp.		0.49
<i>Physa fontinalis</i>	(Linnaeus, 1758)	0.49
<i>Pisidium</i> sp.		0.49
<i>Planorbium corneum</i>	(Linnaeus, 1758)	0.49
<i>Planorbis (Planorbis) carinatus</i>	(O.F. Müller, 1774)	0.49
<i>Planorbis (Planorbis) planorbis</i>	(Linnaeus, 1758)	0.49
<i>Platycnemis pennipes</i>	(Pallas, 1771)	0.49
<i>Proasellus meridianus</i>	(Racovitza, 1919)	0.49
<i>Procloeon bifidum</i>	(Bengtsson, 1912)	0.49
Ptychopteridae		0.49
<i>Scarodytes halensis</i>	(Fabricius, 1787)	0.49
<i>Segmentina nitida</i>	(O.F. Müller, 1774)	0.49
<i>Sialis lutaria</i>	(Linnaeus, 1758)	0.49
<i>Sigara (Pseudovermicorixa) nigrolineata</i>	(Fieber, 1848)	0.49
<i>Sigara (Sigara)</i> sp.		0.49
<i>Sigara (Subsigara) falleni</i>	(Fieber, 1848)	0.49
<i>Sphaerium</i> sp.		0.49
<i>Stagnicola palustris</i>	(O.F. Müller, 1774)	0.49
Stratiomyidae		0.49

Table C1 (Continued)

Insensitive taxa		W
Syrphidae		0.49
<i>Theodoxus fluviatilis</i>	(Linnaeus, 1758)	0.49
<i>Unio</i> sp.		0.49
<i>Valvata (Valvata) cristata</i>	(O.F. Müller, 1774)	0.49
<i>Viviparus viviparus</i>	(Linnaeus, 1758)	0.49
<i>Glossiphonia complanata</i>	(Linnaeus, 1758)	0.48
<i>Ephemera vulgata</i>	(Linnaeus, 1758)	0.47
<i>Amphinemura standfussi</i>	(Ris, 1902)	0.37
<i>Asellus aquaticus</i>	(Linnaeus, 1758)	0.37
<i>Athripsodes aterrimus</i>	(Stephens, 1836)	0.37
<i>Callicorixa wollastoni</i>	(Douglas and Scott, 1865)	0.37
<i>Hydroporus obscurus</i>	(Sturm, 1835)	0.37
<i>Mystacides azurea</i>	(Linnaeus, 1761)	0.37
<i>Mystacides nigra</i>	(Linnaeus, 1758)	0.37
<i>Polycelis felina</i>	(Dalyell, 1814)	0.37
<i>Potamopyrgus antipodarum</i>	(J.E.Gray, 1843)	0.37
Psychodidae		0.37
Tabanidae		0.37
<i>Valvata (Cincinna) piscinalis</i>	(O.F. Müller, 1774)	0.37
Agrypnia obsoleta group		0.25
<i>Anacaena bipustulata</i>	(Marsham, 1802)	0.25
<i>Aplexa hypnorum</i>	(Linnaeus, 1758)	0.25
<i>Brachytron pratense</i>	(Müller, 1764)	0.25
<i>Corixa panzeri</i>	(Fieber, 1848)	0.25
<i>Dytiscus marginalis</i>	(Linnaeus, 1758)	0.25
<i>Ecnomus tenellus</i>	(Rambur, 1842)	0.25
<i>Ephemera lineata</i>	(Eaton, 1870)	0.25
<i>Haliplus heydeni</i>	(Wehncke, 1875)	0.25
<i>Haliplus lineolatus</i>	(Mannerheim, 1844)	0.25
<i>Helophorus (Helophorus) strigifrons</i>	(Thomson, 1868)	0.25
<i>Laccobius (Laccobius) colon</i>	(Stephens, 1829)	0.25
<i>Laccobius (Macrolaccobius) sinuatus</i>	(Motschulsky, 1849)	0.25
<i>Laccobius (Macrolaccobius) striatulus</i>	(Fabricius, 1801)	0.25
<i>Limnephilus binotatus</i>	(Curtis, 1834)	0.25
<i>Limnephilus politus</i>	(McLachlan, 1865)	0.25
<i>Limnephilus vittatus</i>	(Fabricius, 1798)	0.25
<i>Paracymus scutellaris</i>	(Rosenhauer, 1856)	0.25
<i>Porhydrus lineatus</i>	(Fabricius, 1775)	0.25
<i>Enochrus testaceus</i>	(Fabricius, 1801)	0.24
<i>Nebrioporus assimilis</i>	(Paykull, 1798)	0.24
<i>Anacaena lutescens</i>	(Stephens, 1829)	0.12
Anodonta group		0.12
<i>Brachycercus harrisellus</i>	(Curtis, 1834)	0.12
<i>Caenis luctuosa</i> group		0.12
<i>Erpobdella octoculata</i>	(Linnaeus, 1758)	0.12
<i>Gyraulus (Armiger) crista</i>	(Linnaeus, 1758)	0.12
<i>Gyraulus (Gyraulus) albus</i>	(O.F. Müller, 1774)	0.12
<i>Haliplus laminatus</i>	(Schaller, 1783)	0.12
<i>Haliplus lineatocollis</i>	(Marsham, 1802)	0.12
<i>Helobdella stagnalis</i>	(Linnaeus, 1758)	0.12
<i>Hemiclepsis marginata</i>	(O.F.Müller, 1774)	0.12
<i>Hippeutis complanatus</i>	(Linnaeus, 1758)	0.12
<i>Hydroporus palustris</i>	(Linnaeus, 1761)	0.12
<i>Ilybius chalconatus</i>	(Panzer, 1796)	0.12
<i>Leptophlebia marginata</i>	(Linnaeus, 1767)	0.12
<i>Leptophlebia vespertina</i>	(Linnaeus, 1758)	0.12
<i>Nemoura avicularis</i>	(Morton, 1894)	0.12
<i>Oulimnius major</i>	(Rey, 1889)	0.12
<i>Oulimnius tuberculatus</i>	(Müller, 1806)	0.12
<i>Sialis nigripes</i>	(Pictet, 1865)	0.12
<i>Sigara (Subsigara) distincta</i>	(Fieber, 1848)	0.12
<i>Stictotarsus duodecimpustulatus</i>	(Fabricius, 1792)	0.12
<i>Laccophilus minutus</i>	(Linnaeus, 1758)	0.01
<i>Agabus bipustulatus</i>	(Linnaeus, 1767)	0
<i>Agabus sturmii</i>	(Gyllenhal, 1808)	0
<i>Anacaena limbata</i>	(Fabricius, 1792)	0
<i>Bathymphalus contortus</i>	(Linnaeus, 1758)	0
<i>Caenis pseudorivulorum</i> group		0
<i>Caenis pusilla</i>	(Navás, 1913)	0
<i>Caenis rivulorum</i>	(Eaton, 1884)	0

Table C1 (Continued)

Insensitive taxa		W
<i>Corixa affinis</i>	(Leach, 1817)	0
<i>Crangonyx pseudogracilis</i>	(Bousfield, 1958)	0
<i>Cyphon</i> sp.		0
<i>Dendrocoelum lacteum</i>	(O.F.Müller, 1774)	0
<i>Dina lineata</i>	(O.F.Müller, 1774)	0
<i>Dryops</i> sp.		0
<i>Dugesia polychroa</i> group		0
<i>Dugesia tigrina</i>	(Girard, 1850)	0
<i>Esolus parallelepipedus</i>	(Müller, 1806)	0
<i>Gammarus lacustris</i>	(Sars, 1863)	0
<i>Gammarus tigrinus</i>	(Sexton, 1939)	0
<i>Gammarus zaddachi</i>	(Sexton, 1912)	0
<i>Glossiphonia paludosa</i>	(Carena, 1824)	0
<i>Glossiphonia verrucata</i>	(Fr. Müller, 1844)	0
<i>Gyraulus (Torquis) laevis</i>	(Alder, 1838)	0
<i>Halipus confinis</i>	(Stephens, 1828)	0
<i>Halipus immaculatus</i>	(Gerhardt, 1877)	0
<i>Helophorus (Helophorus) flavipes</i>	(Fabricius, 1792)	0
<i>Helophorus (Helophorus) obscurus</i>	(Mulsant, 1884)	0
<i>Helophorus (Rhopalohelophorus) arvernensis</i>	(Mulsant, 1846)	0
<i>Hydrochus angustatus</i>	(Germar, 1824)	0
<i>Hydroporus discretus</i>	(Fairmaire and Brisout, 1859)	0
<i>Hydroporus nigrita</i>	(Fabricius, 1792)	0
<i>Hydroporus planus</i>	(Fabricius, 1782)	0
<i>Laccobius (Macrolaccobius) atratus</i>	(Rottenburg, 1874)	0
<i>Laccobius (Macrolaccobius) ytenensis</i>	(Sharp, 1910)	0
<i>Laccophilus hyalinus</i>	(DeGeer, 1774)	0
<i>Limnebius truncatellus</i>	(Thunberg, 1794)	0
<i>Limnephilus bipunctatus</i>	(Curtis, 1834)	0
<i>Limnephilus fuscicornis</i>	(Rambur, 1842)	0
<i>Limnephilus marmoratus</i>	(Curtis, 1834)	0
<i>Nepa cinerea</i>	(Linnaeus, 1758)	0
<i>Ochthebius bicolon</i>	(Germar, 1824)	0
<i>Ochthebius exsculptus</i>	(Germar, 1824)	0
<i>Orthetrum</i> sp.		0
<i>Oulimnius troglodytes</i>	(Gyllenhal, 1827)	0
<i>Physella (Costatella) acuta</i>	(Draparnaud, 1805)	0
<i>Planaria torva</i>	(Müller, 1774)	0
<i>Polycelis nigra</i> group		0
<i>Radix auricularia</i>	(Linnaeus, 1758)	0
<i>Radix balthica</i>	(Linnaeus, 1758)	0
<i>Sigara (Retrocorixa) semistriata</i>	(Fieber, 1848)	0
<i>Sigara (Retrocorixa) venusta</i>	(Douglas and Scott, 1869)	0
<i>Sigara (Subsigara) fossarum</i>	(Leach, 1817)	0
<i>Sigara (Vermicorixa) lateralis</i>	(Leach, 1817)	0
<i>Siphonurus lacustris</i>	(Eaton, 1870)	0
<i>Taeniopteryx nebulosa</i>	(Linnaeus, 1758)	0
<i>Theromyzon tessellatum</i>	(O.F.Müller, 1774)	0
<i>Trocheta bykowski</i>	(Gedroyc, 1913)	0
<i>Trocheta subviridis</i>	(Dutrochet, 1817)	0
<i>Valvata (Tropidina) macrostoma</i>	(Morch, 1864)	0

Appendix D. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2016.05.040>.

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