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SPECIAL ISSUE PAPER

APPLICATION OF THE PROPORTION OF SEDIMENT-SENSITIVE INVERTEBRATES (PSI) BIOMONITORING INDEX

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ABSTRACT

Sedimentation of river beds is a key pressure impacting riverine ecological communities. Research has identified the need for new approaches to help demonstrate and quantify the impacts of excessive fine sediment deposition on benthic macro-invertebrate populations. To help meet this requirement, PSI (the Proportion of Sediment-sensitive Invertebrates) methodology was developed and has been in operational use in the UK for several years. This paper presents a number of case studies, at both national and local scales, showing how the method can be used to identify point and non-point fine sediment pollution, as well as demonstrating the analysis of a national dataset to describe the relationship between PSI and a channel substrate index. A novel approach to displaying PSI data alongside local ecological and hydrological information is also presented and interpreted, to illustrate how improved understanding of biotic and abiotic relationships and interactions can be readily accomplished. Excessive fine sediment accumulation on river beds results in impaired ecosystem health globally. The case studies and examples presented here will provide confidence that the PSI method can form the basis for evidence-gathering and analysis, both within and beyond the UK. The paper concludes with an overview of the use of PSI in catchment research and management, a consideration of the relationship of the metric with other macro-invertebrate indices and a summary of refinements recently applied to the index.

Key words: Fine sediment, aquatic macro-invertebrates, PSI index, catchment management, multi-metric analysis

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INTRODUCTION

All rivers experience a sediment regime which is controlled by a number of factors, including grain size, sediment supply and flow regime (Rosgen, 1994). Changes in regime may be natural, linked to anthropogenic activity, or combinations of both. Human interventions frequently result in the build-up of excessive fine sediment on river beds and this, in turn, changes the community structure of benthic macro-invertebrates as a direct result of smothering of the substratum and the clogging of interstices (Wood *et al*., 1999; Extence *et al*., 2013). Indirect effects also occur as a consequence of changes in macrophyte and algal communities (Graham, 1990; Ryan, 1991; Parkhill and Gulliver, 2002).

Freshwater macro-invertebrates are reliable indicators of environmental conditions and a range of indices, focussing on individual pressures, have been developed to quantify these. Relatively few univariate metrics have been designed, however, to examine the effects of instream sedimentation on benthic macro-invertebrates, with some notable exceptions (see Zweig and Rabeni, 2001; Bennet *et al*., 2004; Weiss and Reice, 2005). Furthermore, a number of multi-metric methods have included appropriate fine sediment sub-components, such as per cent sediment tolerant individuals (Fore *et al*., 1996), sediment tolerance values (Herlihy *et al*., 2005) and per cent depositional individuals (Weigel *et al*., 2002). Trait-based approaches have also been championed, for example, Richards *et al*. (1997) used catchment and reach scale physical properties, including % fines, to predict the life history and behavioural traits of aquatic insects. In addition, Pollard and Yuan (2010) related the proportion of clinging invertebrates to progressive fine sediment accumulation. Larsen and Ormerod (2010) demonstrated associations between patch-scale sedimentation and the traits of taxon size, length of life cycle and feeding habit and Murphy *et al*. (2017), in a national study focusing on agricultural impacts of fine sediment, demonstrated that the prevalence of resistant eggs in combination with life stage or behavioural traits provided an indicator of fine sediment conditions in streams, akin to outputs from taxonomic based sediment metrics. None of these methods have been adapted for wide scale assessment by regulatory authorities, however, and the need for a new biomonitoring approach for local and national application was highlighted by Clews and Ormerod (2009) and Chadd (2010).

New methodologies for assessing fine sediment deposition and entrainment in rivers were subsequently developed by two independent UK research groups in the early years of the 21st century. Extence *et al.* (2013) used expert judgement and the scientific literature to construct the Proportion of Sediment-sensitive Invertebrates (PSI) Index while Murphy *et al.* (2015) developed the Combined Fine Sediment Index (CoFSI), a wholly empirical approach based on sampled information. Since its initial development, PSI has been applied in a variety of situations (e.g. Poole *et al*., 2013; Glendell *et al*., 2014; Gillespie *et al*., 2015; White *et al*., 2016; Conroy *et al*., 2016; Mathers *et al*., 2016; Bradley *et al*., 2017) and has also been subject to further analysis and development – E-PSI (Turley *et al*., 2014, 2015, 2016), where empirical data and expert judgement are combined to refine the original PSI approach, whilst still being constrained by the original taxon fine sediment associations. The E-PSI adjustments have further strengthened the relationship between river ecology and fine sediment. Meanwhile, PSI has been successfully used in environmental regulation and research over the past few years.

This paper thus focuses on examples and case studies, illustrating how the original PSI index has been used in practice at both national and local scales. An assessment of the utility of the technique at the national scale was accomplished by using monitoring data from England and Wales for the period 2010-2012 to examine how well PSI scores correlated with a channel substrate index designed to describe a gradient between fine and coarse sediment-dominated sites (Naura *et al*., 2016[a]). Examples of the spatial assessment of fine sediment deposition, the integration of PSI within a multi-metric framework designed to detect complex environmental pressures and the detection of an acute pollution event are also included as case studies. The fine sediment pressure issues and examples highlighted in the case studies have wider application beyond the UK and with suitable refinement and adaptation, more widespread application of the PSI approach is possible. In this context Extence *et al*. (2013) showed how the technique was modified for use in the Simandou Mountains, Guinea, successfully demonstrating fine sediment impacts in wet and dry seasons in the context of exploratory iron ore operations. The critical first step in the effective and appropriate management of sediment is the collection of sound and compelling evidence and the primary aim of this paper is to demonstrate how this can be accomplished for a variety of needs in a global context.

METHODS

For full details of PSI calculation, reference should be made to Extence *et al.* (2013). Briefly, taxa (families and species) of UK macro-invertebrates were assigned to one of four fine sediment sensitivity ratings – highly sensitive, moderately sensitive, moderately insensitive or highly insensitive. These designations involved a two stage process in which an extensive literature review was undertaken, followed by a detailed assessment of anatomical, physiological and behavioural traits exhibited by individual taxa. To provide further definition and sensitivity, ratings were abundance weighted and the final PSI score describes the proportion of fine sediment-sensitive invertebrates in the whole sample. Scores range from 0 (entirely silted river bed) to 100 (entirely silt-free river bed).

Standardisation of family level PSI scores and other macro-invertebrate metrics used in the UK is achievable by using reference condition models such as the River InVertebrate Prediction And Classification System - RIVPACS (Clarke *et al.*, 2003) and the River Invertebrate Classification Tool - RICT (Clarke *et al.*, 2011). These models specify the unstressed invertebrate community expected at a site using local physical and chemical characteristics versus a UK-wide reference database of biotic and abiotic data, collected at 835 UK sites covering a wide range of environmental conditions. The ratio of the observed PSI score to the expected score provides an Environmental Quality Index (EQI) and the lower the EQI, the greater the indicated fine sediment stress. Both raw PSI and standardised EQI scores are useful in different contexts and examples of both are included in the case studies and application examples reported here.

Although any appropriate sampling method can be utilised to collect raw data for PSI calculation, Extence *et al.* (2013) recommend that for application in Britain and Ireland, the UK Technical Advisory Group methodology for macro-invertebrate sampling and analysis is used (Murray-Bligh *et al*., 1997; Chadd, 2010). This stipulates timed net/sweep surveys, with different habitats being sampled with effort proportional to their occurrence. All of the examples presented in this paper followed this standard sampling approach.

To assess PSI on a national scale, Environment Agency and Natural Resources Wales data from 3000 sites collected between 2010 and 2012 were used. Samples were excluded from the analysis if they were collected for pollution incident assessment. Where there was more than one sample from a site, the average PSI value across seasons and years of survey was used for the analyses. Sites were not screened for other pressures (i.e. they were not excluded from the analysis if they were subject to another influence, such as channel modification or abstraction) to provide a sufficiently large dataset to enable a review of the full range of scores recorded throughout England and Wales and to assess general national trends. It was accepted at the outset that there would be influences on the data derived from these other pressures and that further, more targeted analysis will be required to more accurately assess the fine sediment pressure gradient at a national scale. The national analysis can be used, however, to identify potential hotspots of impact and aid prioritisation for more focussed investigations.

For each site, Channel Substrate Index (CSI) values were derived using predictive models for both the current condition and for semi-natural conditions. For full details of the approach reference should be made to Naura *et al.* (2016[a]), but, in summary, the CSI is derived from River Habitat Survey (RHS) spot-check data (visual assessment of the dominant river substrate; see Raven *et al.* (1997) for full details of the survey method) and describes a gradient between fine sediment and coarse sediment-dominated sites (CSI at Silt dominated sites = -2.3, gravel-pebble = -0.6, cobble = 0.6 and boulder/bedrock = 0.9). CSI values were calculated for each site within the National RHS database and combined with two predictive models using GIS-derived physical variables (such as geology, altitude and slope) for existing site condition and semi-natural condition. The two models were implemented across the entire river network for England and Wales, thus creating a map of predicted CSI values for observed and semi-natural (reference) conditions. Using the results of the two models, observed and semi-natural CSI scores were derived for each of the biological monitoring sites and compared with the observed and predicted family PSI values respectively, using regression analysis in Minitab 16 (Minitab, 2010), following an approach applied successfully within previous studies (see Naura *et al.*, 2016[b]).

There is great value in using standardised EQIs in the context of presenting multi-metric data. Macro-invertebrate communities are rarely affected by single stressors and in practice a range of often interacting pressures influence resident populations (Friberg, 2010). This is especially pertinent in urban areas and those subject to intensive agricultural practices, where both water quality and hydromorphology influences are commonplace (Gergel *et al.*, 2002). One approach to disentangling this complex information is to examine temporal macro-invertebrate data, expressed in terms of different pressure-specific metrics. The Hydro Ecological Validation (HEV) process was initially developed by the Environment Agency in England and Wales to determine whether impaired ecological quality was due to water resource pressure, or whether confounding factors (such as poor water quality) were in whole or in part contributing to the issue (Klaar *et al*., 2014). In its current form, HEV displays time series data for four macro-invertebrate metrics along with a hydrograph showing the local flow record (daily mean flow). Included are the Biological Monitoring Working Party derived metric, ASPT - Average Score Per Taxon (Chesters, 1980), a long-established UK index primarily responding to organic pollution, the similarly derived NTAXA - Number of scoring TAXA, a general index of environmental health, LIFE - Lotic-invertebrate Index for Flow Evaluation (Extence *et al.,* 1999), a metric designed to assess a site`s ecological response to the antecedent flow regime and PSI (Extence *et al.*, 2013), as described here.

STUDY AREAS AND RESULTS

Application of PSI at a National scale.

The assessment of PSI on a national scale resulted in a regression of observed family PSI with the modelled CSI for contemporaneous conditions. This is shown in Figure 1. There is a significant relationship between the indices, with CSI explaining almost 60% of PSI variability. The scatter shown in Figure 1 suggests the coexistence of other pressures on macro-invertebrate communities, as described previously. The results highlight how well PSI reflects the influence of substrate composition on invertebrate assemblages and the importance of considering confounding pressures when assessing the impacts of fine sediment.

In order to assess how well the predicted PSI values reflect the semi-natural conditions, the predicted PSI values and CSI for semi-natural conditions were compared. The results of the regression are presented in Figure 2. There was a significant relationship between the two indices, with CSI (semi-natural) explaining almost 58% of the variability of predicted PSI. Sites with a CSIexp of -1 or less generally represent sites dominated by gravel-pebble substrates, but with increasing fine sediment content. A group of sites showing low PSI (expected) values for a wide range of CSI (expected) values, generally below CSI=-1, suggests that these sites are being under-predicted.

Having established the clear relationship between both observed and expected PSI and modelled substrate conditions, observed/expected PSI ratios were calculated for the national data set. The values were split into three bands, below predicted (< 0.8), meeting prediction (0.8-1.2) and exceeding prediction (> 1.2). The results are mapped in Figure 3 and presented alongside fine sediment accumulation and agricultural fine sediment risk, modelled within a previous study (see Naura *et al*., 2016^{b}). A visual comparison of the three maps shows an overlap between the areas dominated by PSI O/E <0.8 (red dots) and the areas of high and very high fine sediment accumulation (orange and red). The main areas of concern include central and south eastern areas of the UK. Sites meeting or exceeding predictions predominate in Wales, the South-West and Northern areas. Sites where O/E values exceed predictions indicate a coarser substrate than expected (such as occur, for example, downstream of reservoirs which trap fine sediment), or in some cases the expected scores may be under predicted. The consistent spread of exceeded values across England and Wales suggests that both factors may be an issue.

Due to the coexistence of other pressures acting on these data, this is not intended to indicate that excess fine sediment is solely responsible for environmental impacts, but to identify areas for future investigations. A comparison with the agricultural sediment risk map shows some spatial alignment of areas of high risk, with sediment accumulation and biological impact (PSI O/E <0.8). There are some obvious exceptions, such as major urban conurbations including London (indicated by the circle on the map), suggesting that other pressures may be having an influence. In this case, urban sediment sources and channel modifications are likely influences. This approach demonstrates how applying a pressure specific biotic index within a national assessment can be used to identify and prioritise more targeted investigations.

The spatial assessment of fine sediment deposition in the Barlings Eau catchment.

Once a fine-sediment pressure has been indicated, a more detailed and specific investigation is needed to confirm the scale and extent of the issue. The Barlings Eau is a short tributary of the River Witham located to the north-east of Lincoln. The river rises as three streams which merge near Cold Hanworth (Lat: 53.340028, Long: -0.44247246) before flowing for 17 km to the River Witham at Short Ferry. Numerous small tributaries join the main river from both the east and west and these streams characteristically have gravel beds interspersed with slower, fine sediment-dominated sections. The nature of the main river channel is similar, although extensive habitat modification – widening, straightening and deepening - has led to a general decline in habitat heterogeneity. The western tributaries drain the Lincolnshire limestone aquifer, while those to the east drain deeper superficial deposits, although seepages from the scarp edges of sandstone and chalk outcrops can provide minor flow input. This part of Lincolnshire is predominantly agricultural, with arable land comprising a large proportion of the Barlings Eau catchment. There is no industry of note, but a number of small sewage treatment works discharge to some of the feeder streams. The catchment is currently failing to achieve Water Framework

Directive (WFD) good status due largely to the interacting impacts of low flow, river re-sectioning and sedimentation, the latter mainly resulting from a lack of erosion control from the surrounding farmland.

In order to map and quantify the extent of sedimentation pressures in the Barlings Eau catchment, macro-invertebrate surveys were undertaken at 11 sites in the autumn of 2013 and from these samples PSI family scores and observed to expected ratios (EQIs) were calculated. The location of the survey sites and the results obtained are shown in Figure 4. The PSI EQI varies from 0.0 for Faldingworth Beck at Freisthorpe Bridge, to 1.16 for the Dunholme site on Welton Beck. Most of the PSI EQIs shown in Figure 4 are below 0.6, indicating severe and widespread pressure arising from excessive fine sediment deposition on *in situ* macro-invertebrate communities.

Multi-metric analysis.

In order to confirm sediment as an issue, a multi-metric approach is advocated. Macro-invertebrate data are presented as EQIs in HEV plot examples from Eye Brook at Caldecott – Lat: 52.532, Long: -0.722 (Figure 5) and from the River Ancholme at Cadney – Lat: 53.512, Long: -0.492 (Figure 6). The horizontal target lines for the ASPT and NTAXA sub-plots represent the UK WFD boundaries for Good: Moderate status. LIFE and PSI are not yet used for WFD classification and the horizontal target lines for these two metrics are currently only used for local operational management purposes. The 0.94 LIFE EQI standard is derived from the Environment Agency`s Catchment Abstraction Management Strategy (CAMS) and the 0.70 target for PSI is currently based on expert judgement. Work by the UK Technical Advisory Group is underway, however, to calibrate and integrate these two metrics into future WFD reporting and to derive more defensible and widely applicable standards.

The Eye Brook at Caldecott site is located 1.5 km downstream from the 155 hectare on-line Eyebrook Reservoir. Eye Brook drains an agricultural clay catchment of 67 km^2 and water quality is regarded as good throughout. This condition is confirmed by the Good or better WFD classification derived in part from ASPT and NTAXA EQIs. Conversely, it is clear from the LIFE and PSI plots (Figure 5) that sedimentation, linked to insufficient compensation flow from the reservoir and a lack of flow variation, exacerbated by channel modification, has resulted in the build-up of fine sediment at the Caldecott site. There also appears to be a trend of increasing sedimentation through time here and this may be due in part to the gradual disappearance of flushing flows from the reservoir over the 1988 to 2015 period (see Figure 5 hydrograph).

The River Ancholme in North Lincolnshire also drains an agricultural catchment and runs in a northerly direction for 68 km before discharging to the River Humber at South Ferriby. The river is strategically important, being part of the Trent-Witham-Ancholme Transfer Scheme in which water pumped from the River Trent at Torksey augments River Witham flows, before being re-abstracted and moved by pipeline to the headwaters of the River Ancholme. The invertebrate monitoring site at Cadney (Figure 6) is located a short distance upstream from the abstraction point for Cadney reservoir, which supplies industrial and potable water to the South Humber bank area. No major water quality problems are apparent from the ASPT and NTAXA plots in Figure 6 and there are no identifiable flow issues, discharge being higher than natural as a result of the inter basin transfer. However, Figure 6 clearly shows pronounced impacts from the build-up of excessive fine sediment on the river bed.

The detection of acute fine sediment pollution.

In addition to screening for chronic sediment pressure, PSI can be used to identify acute issues. The River Gwash in Rutland was dammed in the early 1970s to create Rutland Water, at 12.6 km^2 surface area, the largest man-made lake in the UK. The reservoir provides drinking water for large areas of the East Midlands and beyond. Raw water is mainly pumped in from the Rivers Welland and Nene, although there is also some natural input from the upper Gwash and surrounding basin.

PSI EQIs for the period 1984 to 2015 are shown for a routine biomonitoring site at Belmesthorpe (Lat: 52.680, Long: -0.460), located 20 km downstream from the reservoir (Figure 7). Also shown is the gauged flow recorded at Church Bridge (Lat: 52.666, Long: -0.4665) located close to the Belmesthorpe site. The regulated nature of the River Gwash is illustrated by the consistently elevated base flow throughout the study period, though tributary feeds provide some flow variability following rainfall events. It is clear from Figure 7 that the River Gwash at Belmesthorpe does not normally experience excessive sedimentation of the river bed, although deposition of fine sediment does increase (lower PSI scores) when flushing flows are absent or reduced. A dramatic fall in PSI EQI occurred, however, in the autumn of 2011, concomitant with field observations of heavy sedimentation within the coarse substratum. This event followed statutory relief valve testing at the reservoir, when on this occasion a plug of sediment was flushed out from the reservoir’s profundal zone. The water quality metrics NTAXA and BMWP-ASPT were barely affected at this time. The ecological effect of the sediment pollution was still evident, however, in the spring of 2012, full recovery being indicated only in the autumn of that year.

DISCUSSION

Since the original PSI methodology was proposed as a simple and cost effective way of assessing the extent of river bed fine sedimentation, its use by the UK regulatory Agencies has been widespread. The case studies presented in this paper illustrate some of the ways in which the methodology has improved diagnostic and investigative capabilities and in an international context, there are many common scenarios which are also experienced in the UK that lead to the excessive accumulation of fine sediment on river beds (see Extence *et al*., 2013). PSI will be effective as an evidence gathering tool in all of these situations.

Macro-invertebrate indices have been used comprehensively in the past to detect and quantify organic pollution (e.g. Dahl *et al*., 2004), but in comparison, fine sediment pollution has received less attention both in practice and in terms of management action (Extence, 1978; Braccia and Reese, 2006). In this context, the impact of poor working practice during valve testing at Rutland Water in 2011 was clearly demonstrated and steps were immediately taken by the Water Company concerned to prevent any reoccurrence. Ongoing river monitoring has provided compelling evidence that the measures taken following this event have been successful.

The spatial monitoring and assessment of diffuse fine sediment pollution has been equally successful. In the Barlings Eau example, EQIs indicate a wide range of effects but it is clear that the eastern tributaries are the most sedimented (mean PSI O/E 0.28, range 0.00 - 0.74). The main river sites show moderate impacts throughout (EQIs 0.50, 0.53 and 0.46) and clearly also need attention. The western limestone-fed tributaries are less affected, with a mean PSI O/E of 0.88 and a range of 0.35 - 1.16. In this case, these results have helped in targeting catchment walkovers, aimed at identifying sources of fine sediment pollution and the PSI overview has been of critical importance in determining priorities when limited catchment resources are available. While these results are of local interest only, this case study demonstrates the wider principles of evidence-gathering, which apply globally, to quantify impacts and steer management action.

The last two decades have also seen a substantial rise in the amount of river restoration work undertaken in the UK (Smith *et al*., 2014) and elsewhere (e.g. Kail *et al.,* 2015), but frequently, no attempt is made to monitor the success or otherwise of rehabilitation schemes (Kondolf, 1995). Feedback on river restoration successes detected by invertebrates (e.g. Friberg *et al.,* 1994; Extence *et al*. 2013), partial successes (e.g. Kail *et al*., 2015; Tetu *et al*., 2016) and failures (e.g. examples cited by Hammond *et al.,* 2011) is essential to inform and influence the future design and planning of river restoration projects. The application of a sediment sensitive index, such as PSI, alongside structural and functional approaches, may improve our understanding of the biological response to hydromorphological change, which has proved difficult to detect in the past (Feld *et al.,* 2014).

Regarding improved data analysis, Clews and Ormerod (2009) demonstrated the diagnostic capabilities of single univariate indices (in this case BMWP/ASPT, LIFE and the acidification metric AWIC) when used together in the relatively unpolluted River Wye catchment in the UK. They went on to advocate the development of further pressure specific metrics including those responsive to morphological modifications, sedimentation and metal impacts. The concept of a graphical display of multi-metric information, including a sediment-sensitive index, evolved from these recommendations, the end product being the HEV plots now being used throughout the Environment Agency of England and by other UK environmental regulators, such as Natural Resources Wales.

Although HEV provides only a simple presentation of information, this initial step can be crucial in determining key pressures, their interactions and their relationship with hydrology at monitoring sites. Figures 5 and 6, for example, clearly demonstrate good prevailing water quality for both the Eye Brook and River Ancholme. In both these cases, however, fine sedimentation is revealed as a key pressure. For the Eye Brook this can be linked to regulated low flows, (exacerbated by habitat modification), whereas in the River Ancholme, excessive fine sediment deposition is occurring despite flows being higher than natural. A combination of factors are at play here, including erosion and diffuse fine- sediment input from the arable catchment and a deepened, straightened and widened river channel encouraging the deposition and accumulation of fine sediment. The Ancholme is not achieving good WFD status for phosphate, diatoms, macrophytes and fish and all of these failing elements can be linked to sedimentation. Wherever possible, site specific physio-chemical data should be collected alongside ecological information, as this is essential for accurate analysis and interpretation.

A further example of the exploration of relationships between macro-invertebrate metrics, to enable a better understanding of how key pressures interact, is provided by Bradley *et al*. (2017). These authors demonstrated that for a number of West Midlands streams, low flows linked to groundwater abstraction were adversely affecting *in situ* invertebrate communities and LIFE scores. However, at some sites PSI analysis showed that excessive fine sediment impacts were acting independently of flow to override abstraction effects. This understanding is important, as the information identifies where abstraction and excessive fine sediment impacts need to be mitigated separately or together.

In terms of national application, this is the first time PSI data has been analysed at this scale. Results are promising, showing a relationship between PSI and modelled substrate composition with PSI score proving an adequate descriptor of channel substrate composition for semi-natural sites with coarse substrata. There is, however, evidence of under-prediction of PSI in natural condition for gravel dominated sites (see Figure 2 where low expected values of PSI are observed for a wide range of CSI expected values). The reason for this may involve the inclusion of width, depth and substrate as predictor values within the current system of obtaining expected values for the River Invertebrate Classification Tool (Clarke *et al*., 2011). All such values are directly affected by the pressure being assessed.

The national map of the distribution of PSI O/E values, suggests that the main areas for concern include the central and south eastern parts of the UK, where there are significant areas of intensive agriculture and extensive urbanisation. Analyses of this sort are extremely useful, since fine sediment input is one of the less well defined pressures and reasons for failure investigations and catchment walkover evidence have highlighted that this can make diffuse sediment sources difficult to identify (Environment Agency, 2010). It should be noted, however, that EQIs are measures of relative impact, so when making comparisons between sites, relative impacts may vary unless the reference denominator is the same. Plotting an absolute score, i.e. PSIobs – PSIexp, may also be effective when making spatial comparisons, but in practice, this is not essential.

One area in which PSI has proved particularly effective has been in demonstrating environmental improvements following change in land management practices. The Catchment Sensitive Farming (CSF) initiative is designed to reduce water pollution, including river bed sedimentation, caused by farming activities. Davey *et al.* (2013) have reported the results of a monitoring and evaluation programme to determine if CSF has achieved its aim. After controlling for confounding factors, there was evidence that ecological status improved throughout England after the introduction of CSF activity based on CSF advice, especially at sites with previously elevated phosphate levels. Several invertebrate metrics were tested, but the strongest response was shown by PSI. The report concluded that the PSI and (to a lesser extent) ASPT metrics were sensitive to the impacts of diffuse water pollution and provided the best indication of ecological recovery following CSF interventions.

Glendell *et al.* (2014) similarly reported that PSI was more strongly related to % fine bed sediment than either LIFE or Ephemeroptera-Plecoptera-Tricoptera (EPT) % abundance in two contrasting catchments (the Aller and Horner) located on the edge of the Exmoor National Park, UK. These authors went on to conclude that PSI and % fine bed sediment cover have the potential to provide simple, sensitive and effective tools for setting dual ecological and physical sedimentation targets, as well as adding additional exploratory power to the existing suite of macro-invertebrate metrics. In a second study validating the index and focused on the upper Thames and its tributaries, Poole *et al.* (2013) showed that the proportion of macro-invertebrates intolerant of sedimentation increased with high proportions of woodland within 100m or 500m of river channels in the upstream catchment. Furthermore, the concentration of beneficial agri-environment scheme river options (e.g. improved/sympathetic tillage practices) within the same distance were positively correlated with higher PSI scores.

Fine sediment impacts resulting from a range of anthropogenic activities adversely affect rivers worldwide. Before appropriate action can be taken, however, sound evidence needs to be collected to quantify the nature and extent of the problem. From the examples provided and discussed here, we believe that a persuasive case can be made for using PSI as a basis for evidence gathering and analysis. In this context and following publication of the original PSI methodology, Turley *et al.* (2014) undertook a comprehensive assessment of PSI across a wide range of UK reference sites and also concluded that PSI was more strongly related to fine-sediment pressures than a number of other macro-invertebrate indices. This research group went on to propose refinements of the PSI approach (E-PSI) at both species (Turley *et al.*, 2015) and family level (Turley *et al.,* 2016), whereby empirical data relating measures of deposited sediment to metric output were used to supplement the expert knowledge and published information underpinning the original method. These changes thus incorporate the inherent mechanistic independence of the original index with a direct measure of response. An encouraging improvement in correlation between E-PSI and deposited fine sediment was subsequently reported. E-PSI can thus be regarded as a hybrid approach, which bridges the polarity of the original PSI and CoFSI (Murphy *et al*., 2015) methods and in consequence offers exciting possibilities for future data analysis and exploration.

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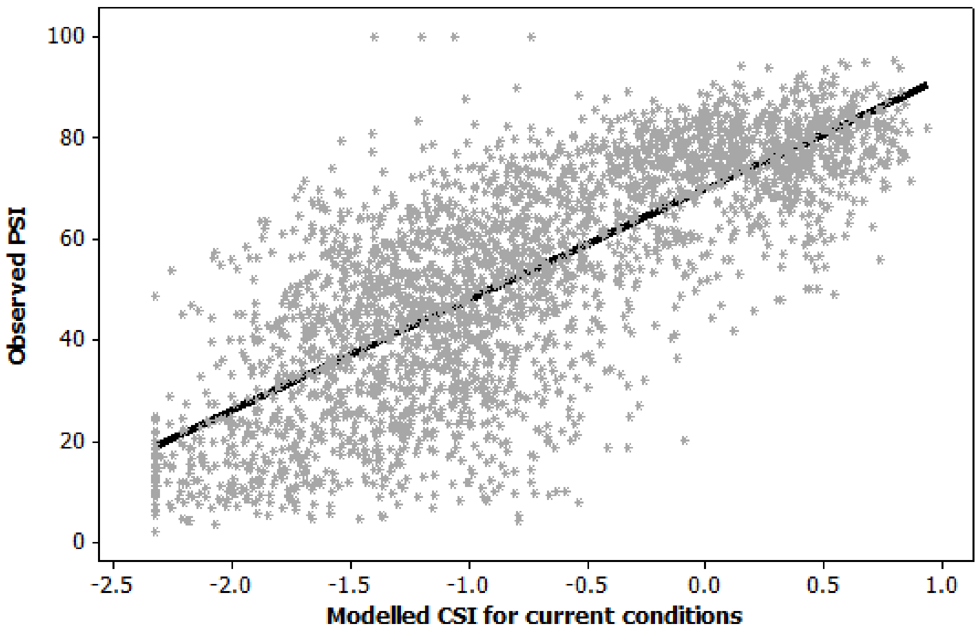

Figure 1: Observed PSI values compared with Modelled CSI for current conditions (PSI = 69.9 + 21.7 CSI R-Sq = 59.7%, n= 2976).

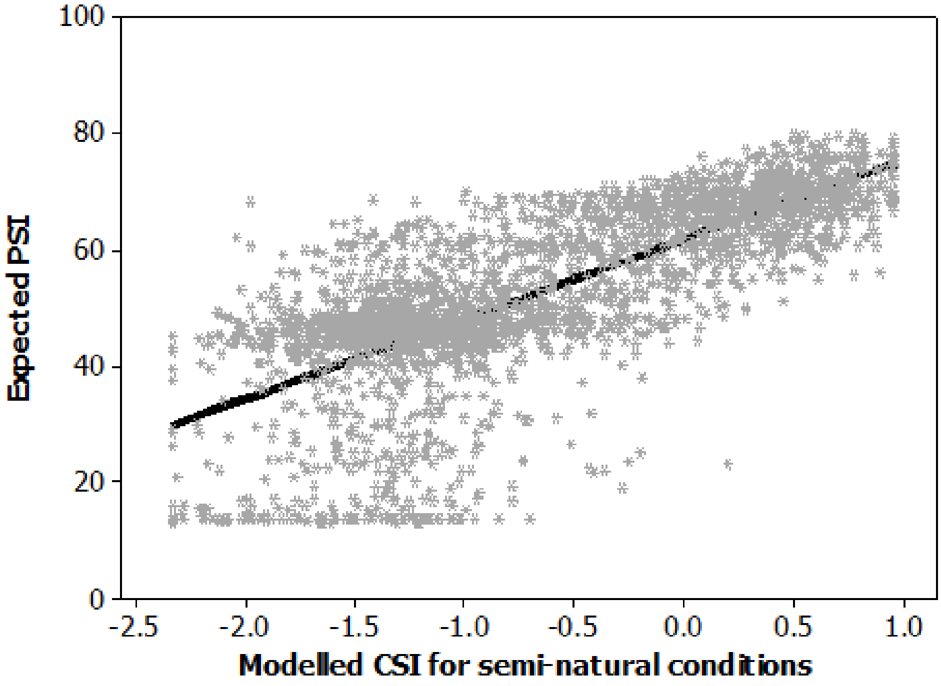

Figure 2: Predicted PSI values compared with Modelled CSI for semi-natural conditions (PSIexp = 61.8 + 13.7 CSIexp R-Sq = 57.9%, n = 2963).

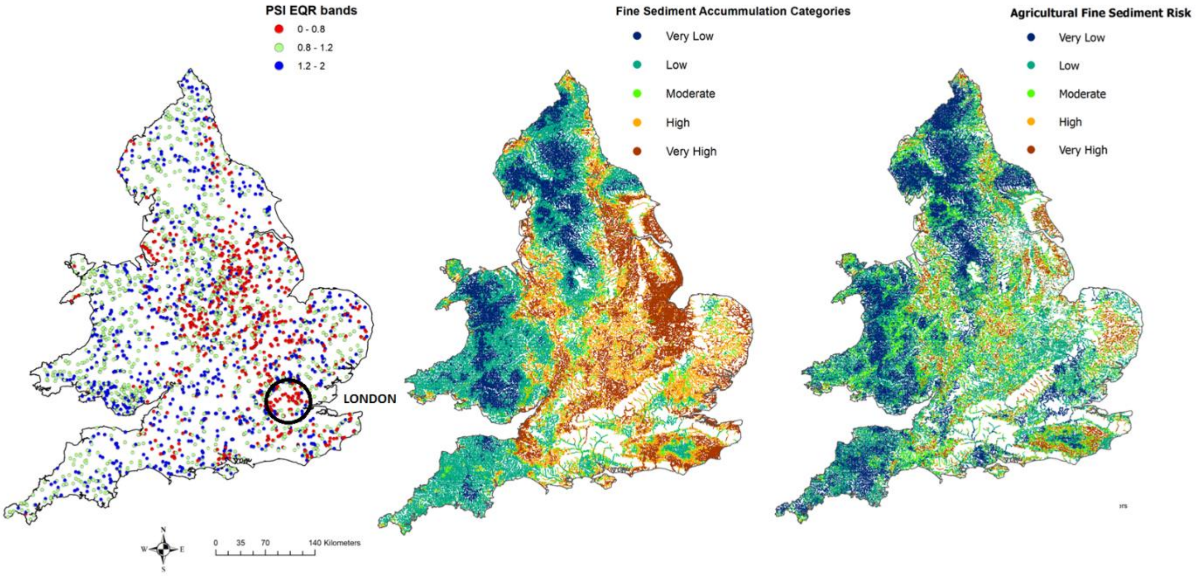

Figure 3. The distribution of PSI EQR (EQI) scores, the ratio of observed scores to expected scores predicted using reference condition models (left), modelled Fine Sediment Accumulation FSA (middle) and Agricultural Sediment Risk ASR (right) across England and Wales. Maps of FSA and ASR reprinted from Naura *et al*. (2016b), with permission from Elsevier.

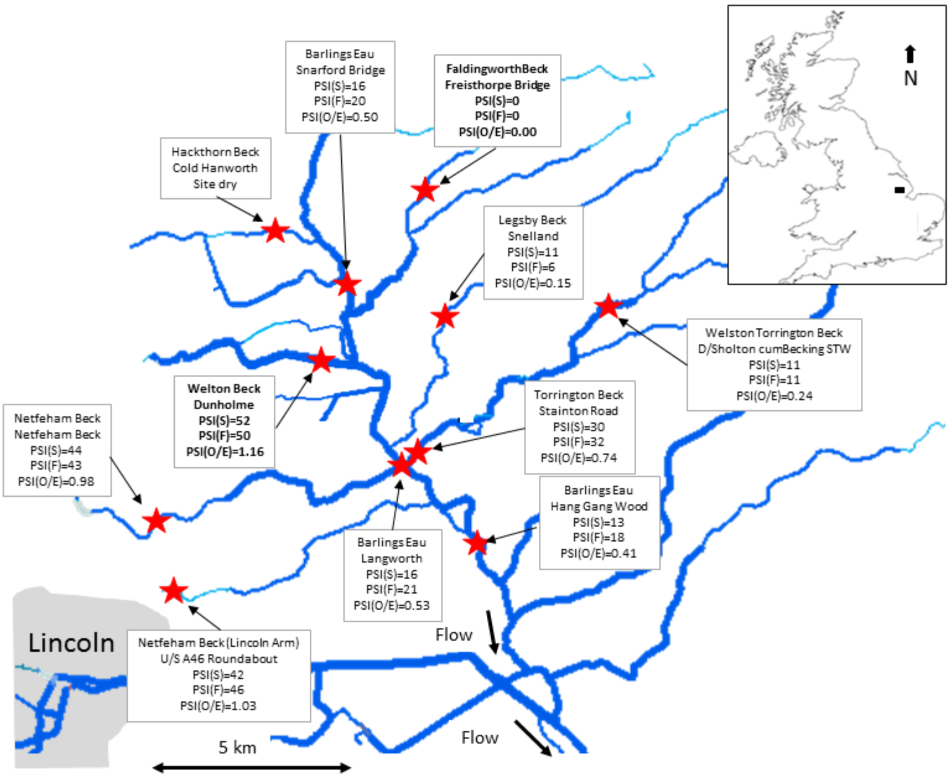

Figure 4: Macro-invertebrate survey sites on Barlings Eau, Lincolnshire, UK (Autumn 2013) showing raw PSI scores and PSI Environmental Quality Index (EQI) scores

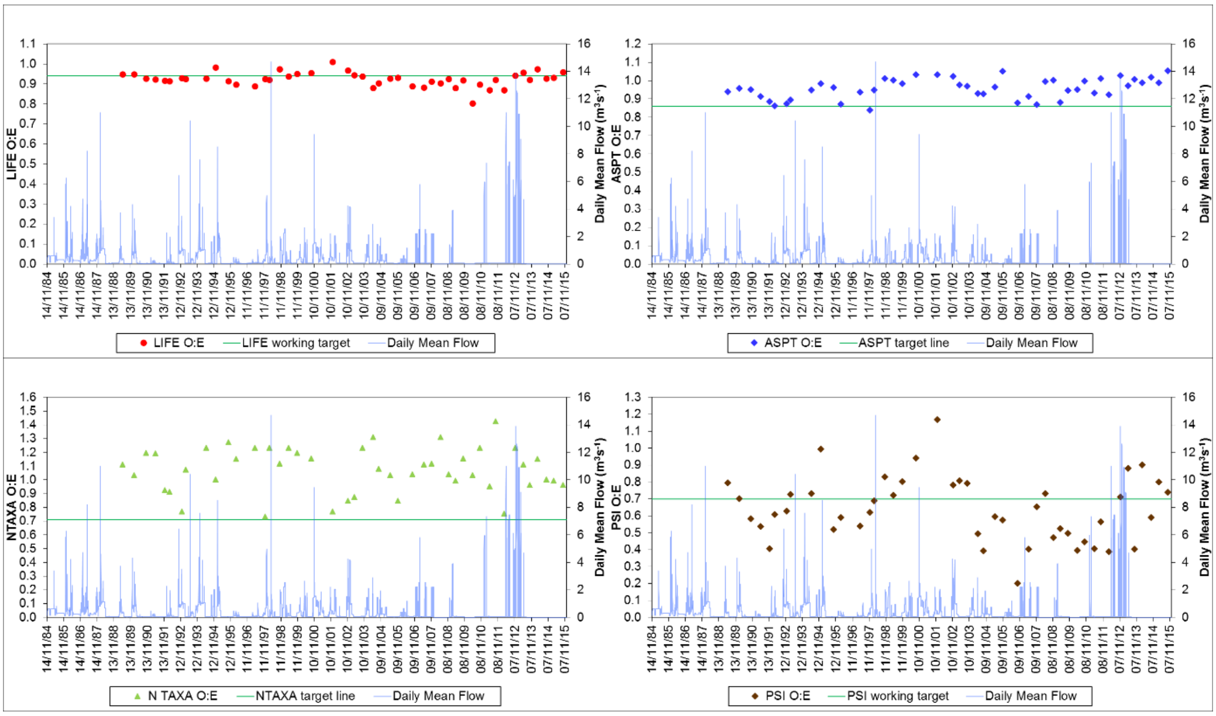

Figure 5: Environmental Quality Index (EQI) time-series plots for BMWP ASPT, BMWP Ntaxa, LIFE and PSI from Eye Brook, Caldecott, Leicestershire, UK.

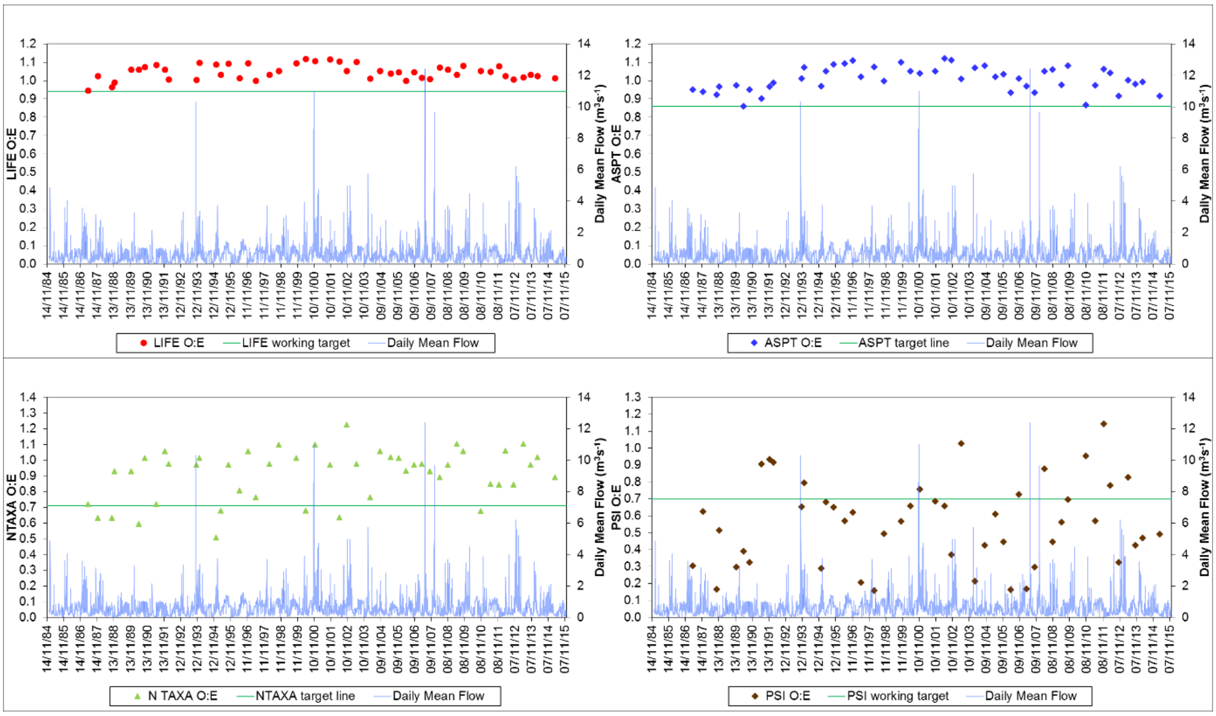

Figure 6: Environmental Quality Index (EQI) time-series plots for BMWP ASPT, BMWP Ntaxa, LIFE and PSI from the River Ancholme, Cadney, Lincolnshire, UK.

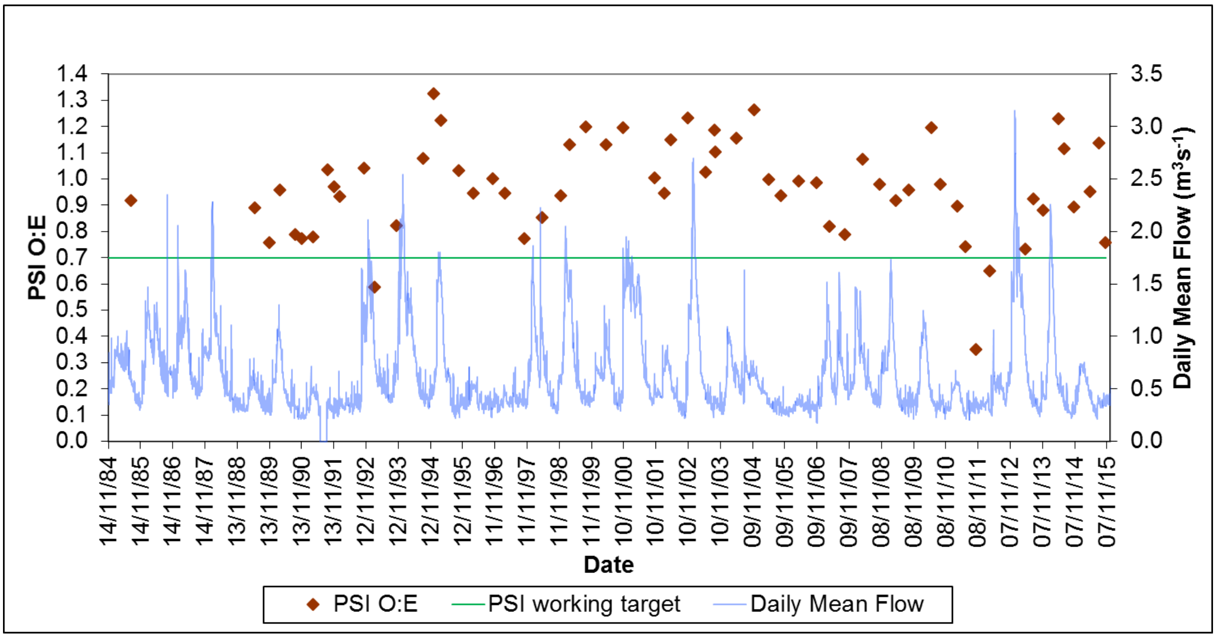

Figure 7: PSI Environmental Quality Index (EQI) time-series for the River Gwash at Belmesthorpe, Rutland, UK, showing associated hydrograph and Environment Agency working target - horizontal line.