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Evaluating the relationship between biotic and sediment metrics using mesocosms and field studies

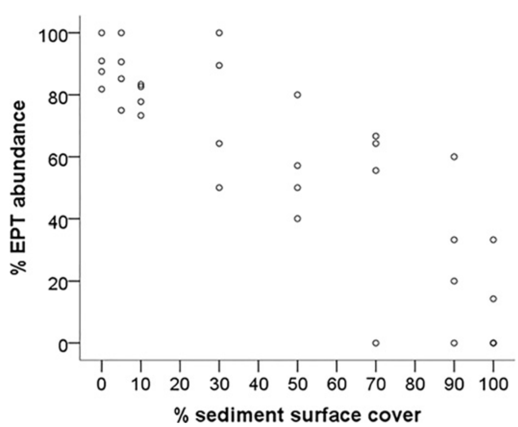
E. Conroy [a,*], J.N. Turner [b], A. Rymszewicz [c], M. Bruen [c], J.J. O'Sullivan [c], D.M. Lawler [d], H. Lally [a,e], M. Kelly-Quinn [a]

[a] School of Biology and Environmental Science, University College Dublin, Belfield, Dublin 4, Ireland
[b] School of Geography and UCD Earth Institute, University College Dublin, Belfield, Dublin 4, Ireland
[c] UCD Dooge Centre for Water Resources Research, School of Civil Engineering, UCD Earth Institute, University College Dublin, Belfield, Dublin 4, Ireland
[d] Centre for Agroecology, Water and Resilience, Coventry University, UK
[e] Marine and Freshwater Research Centre, Galway-Mayo Institute of Technology, Dublin Rd, Galway City, Ireland

HIGHLIGHTS

- Sediment impact detection using appropriate bioassessment metrics is challenging.
- Mesocosm and field observations were used to assess the relationship between metrics.
- % EPT abundance and richness metrics were negatively correlated with surface cover.
- Inclusion of biotic and sediment metrics in fluvial monitoring would be beneficial

GRAPHICAL ABSTRACT



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ABSTRACT

An ongoing research challenge is the detection of biological responses to elevated sediment and the identification of sediment-specific bioassessment metrics to evaluate these biological responses. Laboratory mesocosms and field observations in rivers in Ireland were used to evaluate the relationship between a range of biological and sediment metrics and to assess which biological metrics were best at discerning the effects of excess sediment on macroinvertebrates. Results from the mesocosm study indicated a marked decrease in the abundance of sensitive taxa with increasing sediment surface cover. % EPT (Ephemeroptera, Plecoptera, Trichoptera) and % E abundances exhibited the strongest negative correlation with sediment surface cover in the mesocosm study. The field study revealed that % EPT abundance was most closely correlated with % sediment surface cover, explaining 13% of the variance in the biological metric. Both studies revealed weaker relationships with a number of other taxonomy-based metrics including total taxon abundance, total taxon richness and moderate relationships with the Proportion of Sediment-sensitive Invertebrates metric (PSI). All trait-based metrics were poorly correlated with sediment surface cover in the field study. In terms of sediment metrics, % surface cover was

* Corresponding author.
E-mail address: elizabeth.conroy@ucdconnect.ie (E. Conroy).

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more closely related to biological metrics than either re-suspendable sediment or turbidity. These results indicate that % sediment surface cover and % EPT abundance may be useful metrics for assessing the effect of excessive sediment on macroinvertebrates. However, EPT metrics may not be specific to sediment impact and therefore when applied to rivers with multiple pressures should be combined with observations on sediment cover.

1. Introduction

Fine sediment is a vital element in freshwater systems and important to nutrient cycling, substrate composition and heterogeneity, all of which play a part in regulating the micro-environmental conditions in which macroinvertebrates reside (Rabeni and Minshall, 1977; Minshall, 1988; Richards et al., 1997; Wood and Armitage, 1997; Owens et al., 2005). Excessive fine sedimentation, however, may alter substrate composition, increase habitat homogeneity and is considered to be a major ecosystem stressor leading to ecological impairment (Rabení et al., 2005; Wood et al., 2005; Niyogi et al., 2007; Bryce et al., 2010). A suite of impacts including the clogging of substrate interstices, smothering of habitats and reduction in habitat stability (Wood and Armitage, 1997; Bilotta and Brazier, 2008; Jones et al., 2012) may cause significant environmental degradation and, in extreme cases, lead to a significant deviation from 'reference conditions' (Bilotta and Brazier, 2008).

A number of research strategies, such as field surveys, stream-side experiments and/or laboratory experiments (mesocosms) may help to detect ecological responses to, and differentiate between natural and anthropogenic stressors. (Robinson and Uehlinger, 2008; Townsend et al., 2008; Wagenhoff et al., 2012). Combining different research strategies, each with their own specific strengths and weaknesses, may also help to tease out the effects of confounding factors, e.g. multiple stressors and flow variations (Robinson and Uehlinger, 2008; Townsend et al., 2008; Wagenhoff et al., 2012). Using mesocosms allows for the isolation and direct manipulation of stressors while minimising confounding factors (Suren and Jowett, 2001; Connolly and Pearson, 2007; Wagenhoff et al., 2012; O'Callaghan et al., 2015; Piggott et al., 2015). However, the extrapolation of findings from mesocosm studies to whole river systems should be undertaken with caution due, in part, to differences in spatial and temporal scales (Townsend et al., 2008; Sandin and Solimini, 2009). While field surveys best represent natural conditions, they may be influenced by a range of co-varying drivers which may mask or exacerbate biological responses (Robinson and Minshall, 1986; Larsen et al., 2009; Matthaei et al., 2010; Robinson et al., 2011; Wagenhoff et al., 2011; Burdon et al., 2013; Glendell et al., 2014; Turley et al., 2014).

An array of biological metrics have been developed to detect the impact of specific environmental stressors such as nutrients, acidification, flow and habitat loss (Hilsenhoff, 1987; Hawkes, 1998; Extence et al., 1999; Davy-Bowker et al., 2005; Dunbar et al., 2010). In contrast, relatively few metrics have been specifically developed to detect the effects of sedimentation on macroinvertebrates (Relyea et al., 2000; Zweig and Rabení, 2001; Bryce et al., 2010), and the lack of a standardised bioassessment method to detect the impacts of fine sediment deposition makes inter-study comparisons problematic (Clews and Ormerod, 2009). Recently, Extence et al. (2013) developed a sediment-sensitive macroinvertebrate metric, Proportion of Sediment-sensitive Invertebrtes (PSI), based on expert review of existing literature and an assessment of biological traits to assign taxa to one of four sensitivity groups while Murphy et al. (2015) developed a combined fine sediment metric (CoFSIsp) based on two sub-indices which captures macroinvertebrate responses to organic sediment in erosional zones (oFSIsp) and to total fine sediment in depositional zones (ToFSIsp).

Conserving and protecting aquatic systems is of huge importance for environmental sustainability, but also politically and in terms of public perception (Strayer, 2006). Macroinvertebrates are key water quality indicators in many bioassessment programmes (e.g. Bonada et al., 2006) and their sensitivity to pollutants, including fine sediment, make them ideal organisms for assessing water quality (Rosenberg and Resh, 1993; Bonada et al., 2006). It is clear from previous studies that EPT taxa are sensitive to elevated sediment, but the mechanisms causing the responses are not well elucidated and thus a wide range of mechanisms have been proposed to explain observed changes in community structure (Zweig and Rabení, 2001; Niyogi et al., 2007; Bryce et al., 2010; Wagenhoff et al., 2011; Sutherland et al., 2012; Burdon et al., 2013). For example, ephemeropteran taxa can be impacted by sedimentation in a number of ways. Smothering of the periphyton by sediment can lead to impaired scraper feeding (Larsen and Ormerod, 2010). The grazer/clinger *Ecdyonurus* sp., requires clean interstices so as to maintain position in the substrate and the grazer *Baetis rhodani* have been shown to generally avoid fine substrates (Rabení et al., 2005; Wood et al., 2005; Larsen and Ormerod, 2010; Pollard and Yuan, 2010). Fine particles can also impair the gill respiring mechanisms of these two taxa (Lemly, 1982). In contrast, others have found positive relationships between sedimentation and baetid mayflies (Angradi, 1999; Sutherland et al., 2012). Taxa from some trichopteran families have also been shown to be negatively impacted by sediment (Larsen et al., 2009). The preferred habitat of hydropsychids is fast flowing, sediment-free habitats as sediment can interfere with the feeding nets of this taxon (Strand and Merritt, 1997). In contrast, a number of Limnephilidae (Trichoptera) and Caenidae (Ephemeroptera) taxa are known to be less sensitive to fine sediment (Turley et al., 2014). Clearly, a current research challenge is the detection of biological responses to elevated fine sediment and identification of sediment-specific bioassessment metrics. Furthermore, while many of the biological impacts of sedimentation are linked to sediment deposition, current guidelines, based on suspended sediment concentration as set out in the recently repealed Freshwater Fisheries Directive (78/659/EEC), may not be appropriate to protect ecological status (Cooper et al., 2008: Kefford et al., 2010; Bilotta et al., 2012; Jones et al., 2012).

The present study explores how a number of commonly used macroinvertebrate taxonomy- and trait-based metrics respond to measures of deposited sediment using both mesocosm laboratory channels and a field study, with the latter representing more realistic conditions. Three sediment metrics, % sediment surface cover, re-suspendable sediment and turbidity, which gave accurate estimates of deposited sediment levels (Conroy et al., 2016a, 2016b) were assessed together with a range of biological metrics to establish which were the most appropriate in detecting sediment effects. In addition, the mesocosms provided evidence of responses to sediment addition through analysis of macroinvertebrate drift and of the taxa remaining in the channels at the end of the experiment. In this regard it was hypothesised that the channels with high sediment loads would (i) show increased rates of macroinvertebrate drift during the first 24 h and throughout the experiment and (ii) have decreased abundance of taxa remaining in channel at the end of the experiment. The field study also assessed temporal variability in the strength of the associations between biological metrics and sediment metrics and whether taxonomic resolution, i.e. family versus species, influenced the strength of the associations. The hypothesis to be tested was that taxa richness and abundance metrics would be negatively correlated with % fine sediment surface cover.

2. Materials and methods

2.1. Mesocosm study

The experimental design consisted of eight levels of deposited fine sediment amounts with four replicates of each treatment giving a total of 32 experimental channels. A 3-cm bed of washed, sieved, gravel and pebble substrate (4–20 mm diameter), 9 L of river water (approximate water depth 120 mm) and four flat cobbles with attached algae sourced from a good status (WFD) river (Rathmore Stream, Co. Kildare, Ireland) were added to each channel (1500 mm × 150 mm). A 63-μm mesh 'drift net' was secured within each channel to capture drifting macroinvertebrates and flow was maintained using an aquarium pump (Fig. 1). Each channel was seeded with macroinvertebrates (from two Surber samples) collected from the aforementioned stream on the same day, reducing the potential for natural temporal variability in biological communities (Rosenberg and Resh, 1993). Macroinvertebrates were allowed to acclimatise for two days prior to sediment treatment. Sediment was sourced from an exposed river bank of a stream draining a catchment (Glencullen River, Co Wicklow, Ireland) with minimal human influence and no historic nutrient inputs thus reducing potential responses to multiple stressors (Ormerod et al., 2010). The sediment sampling site is within a woodland that is a designated nature reserve and the river that drains the site is at high status with low nutrient content. Thus, the sediments would not be expected to have anthropogenically enhanced nutrient levels so a detailed analysis of the sediment composition was not considered necessary.

The sediment was oven-dried, sieved and fine sediment (<1 mm) was retained. Sediment disposition was facilitated by turning off pumps prior to addition. A predetermined weight of fine sediment was evenly spread over the gravel substrate as undertaken by Wagenhoff et al. (2012) at 17.00 h on day one so as to achieve the required sediment surface cover of: 100%, 90%, 70%, 50%, 30%, 10%, 5% and 0% (control). These sediment levels were estimated by the same observer, thereby reducing possible observer bias (Wang et al., 1996). Macroinvertebrates were exposed to 1 of 8 fine sediment treatments in a randomised-block design with four replicates per treatment. Drifting macroinvertebrates were collected at midnight and 0600 h, and combined to give daily drift, on each of six consecutive days. While the experiment covered a short time period, previous studies have shown that macroinvertebrate responses to sediment generally occur within 24 to 48 h following sediment addition (Suren and Jowett, 2001; Larsen and Ormerod, 2010; Larsen et al., 2011; O'Callaghan et al., 2015). Macroinvertebrates remaining in the channels on day seven were retrieved by elutriating the substrate through a 250 μm-mesh sieve and preserved in 70% Industrial Methylated Spirits (IMS). In the laboratory, the macroinvertebrates were identified to the lowest practicable taxonomic level (species where possible) using Freshwater Biological Association (FBA) identification keys (Hynes, 1977; Macan and Cooper, 1977; Elliott and Mann, 1979; Elliott et al., 1988; Wallace et al., 1990; Edington and Hildrew, 1995; Nilsson, 1996, 1997). To ensure comparable conditions between treatments, daily measurements were taken of water pH, temperature, conductivity and dissolved oxygen (DO) using a WTW automatic field probe, velocity using a FLO-MATE flow meter and turbidity, using a HACH 2100NIS turbidity meter.

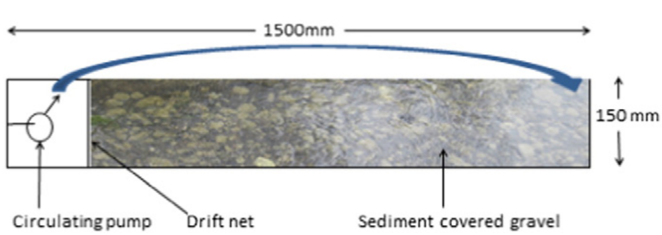

Fig. 1. Schematic illustrating experimental mesocosm (not to scale).

2.2. Field study

The field study was conducted during two seasons, spring (April/May) and autumn (Sept./Oct.) in 2013, across eight rivers located in the North East and midlands of Ireland (Fig. 2) where cattle access represented a potential point source of sediment. Dominant land use at all sampling locations was intensive agriculture (mainly dairy) while river typology was calcareous with low slope (Dodkins et al., 2005). In each study catchment, sampling was conducted at two locations, upstream and downstream of each cattle access drinking point. Six replicate Surber samples (1 mm-mesh) were taken within the mid-channel and margins at each sampling location, which included the first run/riffle area in each direction. Surber samples capture smaller-scale variations where the number of taxa collected can be related to a well-defined sampling area providing an absolute measure of taxon density per unit area (Carter and Resh, 2001). Surber samples also allowed for the collection of macroinvertebrates and sediment measurements at the same scale and location. Extence et al. (2013) suggested that any suitable sampling method can be used to collect macroinvertebrate sampling for PSI calculations. Macroinvertebrate samples were preserved in 70% IMS and processed as described for the mesocosm experiments.

Visual estimations of % deposited fine sediment (<2 mm), which gives an approximation of surface sediment levels, were made within each Surber sampler frame prior to macroinvertebrate sampling (Zweig and Rabení, 2001; Rabení et al., 2005; Matthaei et al., 2006; Larsen et al., 2009). Two additional sediment metrics were included in this study, re-suspendable sediment (RSS) and turbidity, both of which give an approximation of surface and subsurface sedimentation levels (Conroy et al., 2016a, 2016b). A stilling well (215 × 400 mm) was pressed into the stream bed within the frame of the Surber sampler. Water depth within the stilling well was recorded, the water and top 5 cm of the bed substratum was agitated manually for 30 s and a manual grab sample containing re-suspendable sediment was taken (Lambert and Walling, 1988; Wagenhoff et al., 2011; Conroy et al., 2016a, 2016b). Turbidity (NTU) of grab samples was also recorded using the HACH 2100nIS turbidity meter. Water samples were filtered, dried, weighed and calculated as re-suspendable sediments (g m^{-2}) (Lambert and Walling, 1988; Conroy et al., 2016a, 2016b).

3. Statistical analysis

The Asterics 3.3 programme (http://www.aqem.de/) was used to calculate a number of taxonomy-based metrics for the mesocosm and field studies including total taxon richness and abundance, Ephemeroptera (E) abundance, % Ephemeroptera, % Plecoptera and Trichoptera (% EPT) abundance, Biological Monitoring Working Party (BMWP) score and Average Score Per Taxon (ASPT), and habitat, feeding and locomotion trait metrics. PSI species scores (PSI_S) and PSI family (PSI_F) were also calculated (Extence et al., 2013). Habitat, feeding and locomotion trait metrics (e.g. grazers/scrapers, % gatherers/collectors and % sprawlers/walkers) and total taxon richness were only calculated for the field data because few taxa remained in the mesocosm channels at the end of the experiment. Summary statistics showing means and standard deviations for biological metrics and environmental variables are included in Appendix A while Appendix B shows summary statistics for biological and sediment metrics in the field study.

Friedman's ANOVA (non-parametric, repeated measures ANOVA, see Clews and Ormerod, 2010: Dytham, 2011; O'Callaghan et al., 2015) was used to compare numbers of drifting taxa (total taxon abundance and Heptageniidae abundance) between sediment treatments throughout the whole experiment. Post-hoc analysis using Wilcoxon signed-rank tests, with a significance level of $\alpha < 0.01$, were used to indicate where significant differences lay. One-way ANOVA followed by Tukey post-hoc tests were used to detect differences in macroinvertebrate drift within the first 24 h and total numbers of macroinvertebrates remaining in the channels at the end of the experiment between

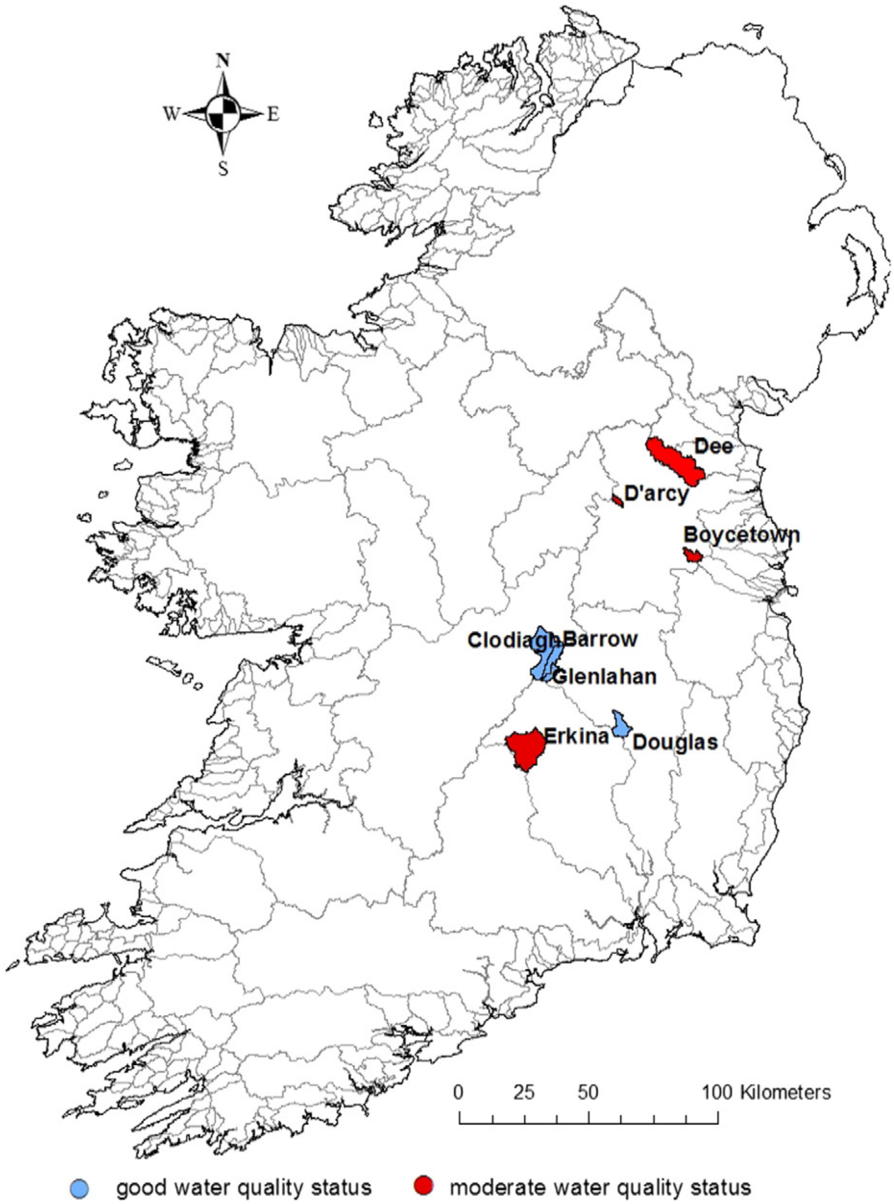

Fig. 2. Map showing the locations of the eight field study catchments in Ireland.

treatments. The association between biological metrics, derived from taxa remaining in mesocosm channels, and % sediment surface cover was measured using Kendall's tau rank correlation, τ.

In the field study, Spearman's rank correlations were used to analyse the association between biological metrics and three sediment metrics (% sediment surface cover, RSS and turbidity) for spring and autumn datasets separately using the software package PASW Statistics 18. Generalised linear mixed-effects models were used to establish how much of the variance in the biological metrics could be explained by the sediment metrics. The mixed model approach deals explicitly with the spatial and temporal non-independence and are appropriate for use on data which has a hierarchical structure (repeated sampling) (Gelman and Hill, 2006) and they take account of the differences in species composition and biological responses across rivers (Pinheiro and Bates, 2000; Pinheiro et al., 2007) and of multiple sites on the same river. The models were fitted by restricted maximum likelihood (REML) using the Imer function in the Ime4 library in R (Pinheiro et al., 2007). The function AIC was used to extract Akaike's information criterion and the R-function p norm was used to estimate the *P*-values. Site and location were treated as random effects while sediment variables (% sediment surface cover, resuspended sediment and turbidity) were fixed effects. Bonferroni corrections were not applied as a priori hypotheses were generated in relation to marcoinvertebrate metrics and sediment metrics (Moran, 2003). The R package r.squared GLMM was used to estimate the variance contribution from both the fixed factors and the combined fixed and random effect factors. The methods used are described by Nakagawa and Schielzeth (2013). The package and methods used are described at http://www.inside-r.org/packages/cran/MuMIn/docs/r.squaredGLMM

4. Results

4.1. Mesocosm study

Water temperature in the experimental channels varied between 5.6 and 7 °C (6.03 mean $\pm$ 0.03 SE). DO always remained >13 mgL-1 (12.36 mean $\pm$ 0.03 SE) while pH was between 7.8 and 8.3 (8.08 mean $\pm$ 0.01 SE). Velocity and turbidity ranged from 0.28 to 0.45 m/s (0.36 mean $\pm$ 0.01 SE) and 1.94 to 10.6 NTU (7.15 mean $\pm$ 0.02 SE), respectively. No significant differences were detected between treatments

($P > 0.05$). Visual observation and low turbidity readings indicated that sedimentation was maintained throughout the experiment.

The total mean abundance in the channels (calculated as total drifting plus total remaining in the channel at the end of the experiment) did not differ significantly between treatments ($F_{(7,24)} = 0.310$, $P = 0.942$). At the beginning of the experiment Diversity (Simpson-Index) and Eveness were not significantly different between treatments ($F_{(7,24)} = 0.310$, $P = 0.942$ and $F_{(7,24)} = 0.310$, $P = 0.942$, respectively) indicating that seeding of channels was relatively uniform. Total macroinvertebrate abundance remaining in channels at the end of the experiment was significantly different between the treatments ($F_{(7,24)} = 7.384$, $P < 0.05$). Abundances in the 5% treatment were significantly higher than all other sediment treatments ($P < 0.025$) bar the 10% sediment cover treatment.

Drift rates followed a diurnal pattern with peaks in drift observed during the hours of darkness. While there were no differences in total abundance drifting in the first 24 h between the treatments ($F_{(7,24)} = 1.37$, $P = 0.264$), there were significant differences in macroinvertebrates drifting over the duration of the experiment ($\chi^2(5) = 98.83$, $P < 0.05$). Abundance of Heptageniidae drifting was higher in the first 24 h than during the other time periods ($\chi^2(5) = 42.62$, $P < 0.05$). While post-hoc tests could not establish which sediment treatments were significantly different, the numbers of drifting Heptageniidae were generally higher in channels with higher sediment cover (>30% coverage) (Fig. 3).

All biological metrics (based on taxa remaining at the end of the experiment) were negatively correlated with % sediment surface cover (Fig. 4). Percentage EPT abundance (Fig. 4a) had the strongest relationship with % sediment surface cover ($\tau = -0.68$) followed by % E abundance (Fig. 4b) and E abundance (Fig. 4c) ($\tau = -0.67$ and $\tau = -0.64$, respectively at $P < 0.001$). The PSI_S metric (Fig 4e, $\tau = -0.52$, $P < 0.001$) and total abundances (Fig. 4d, $\tau = -0.48$, $P < 0.001$) had slightly weaker, moderate correlations, with % sediment surface cover). The weakest relationships were between BMWP (Fig. 4f) and ASPT and % sediment surface cover ($\tau = -0.25$ and $\tau = -0.29$, $P < 0.05$).

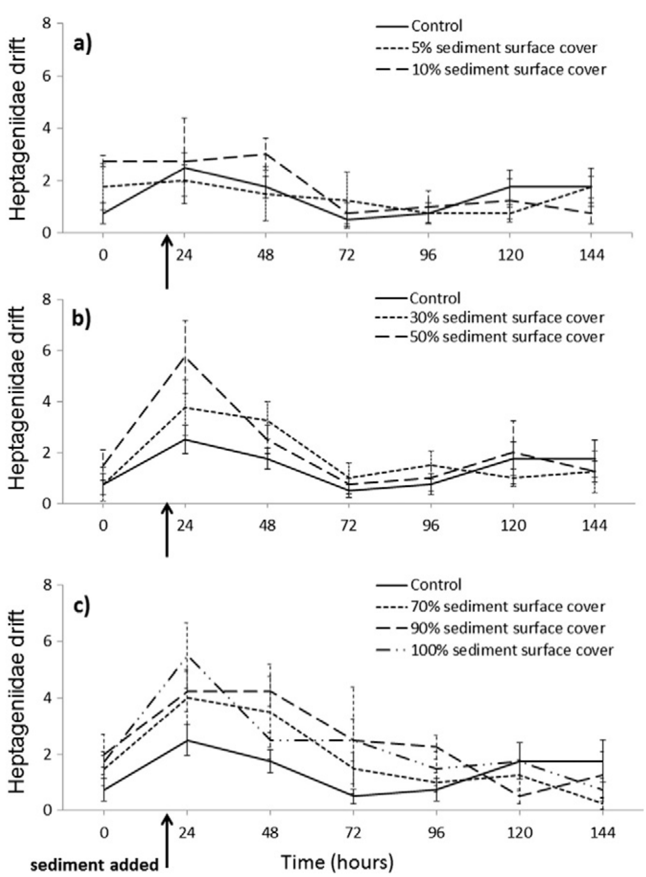

Fig. 3. Mean (± standard error) abundance of Heptageniidae drifting in a) control, 5% and 10% sediment surface cover, b) control, 30% and 50% sediment surface cover and c) control, 70%, 90% and 100% sediment surface cover in mesocosm study.

4.2. Field survey

A total of 384 Surber samples (patch scale) across two seasons were obtained in the field study. Sediment cover at patch scale ranged from 1 to 100% which is wider coverage than reported in other studies (Larsen et al., 2009; Sutherland et al., 2012). However, the sediment gradient was not evenly distributed in the current study, due to natural variability, as almost three quarters of the 384 observations had <50% sediment surface cover (mean 32% ± 1.4 SE). Re-suspendable sediment (RSS) varied between 1 and 3788 g m^{-2} (mean183 g m^{-2} ± 15.3 SE) while turbidity ranged from 2 to 2299 NTU's (mean183 g m^{-2} ± 15.3 SE).

A number of metrics (e.g. E abundance, % E abundance and PSI) showed evidence of seasonal variability in their relationship with sediment measures. The spring dataset showed stronger relationships with surface cover for all but ASPT (Table 1). These seasonal differences were particularly evident between E abundance and surface cover (spring $r_s = -0.43$, $P < 0.01$ and autumn $r_s = -0.16$, $P < 0.05$). The PSI metric also showed some seasonal variability at both species and family level. Spring PSI_S scores (Fig. 5a) were strongly correlated with surface cover ($r_s = 0.47$, $P < 0.01$) and correlations with autumn PSI_S scores were much weaker ($r_s = 0.26$, $P < 0.01$) (Fig. 5b). Similar results was detected for the PSI_F metric (spring data $r_s = -0.40$; autumn $r_s = 0.29$ $P < 0.01$) (Table 1).

Sediment cover (%) had a higher correlation with the biological metrics than either RRS or turbidity (Table 1) and explained a higher proportion of the variation in the models (Table 2).

The strongest associations were with % EPT abundance (spring) and sediment cover ($r_s = -0.57$, $P < 0.01$) and explained 13% of the variance in the models (Table 2). Both total taxon richness and abundance (spring and autumn) were weakly correlated with % surface cover (Table 1). The spring correlation coefficients for the other metrics ranged from −0.33 (BMWP) to −0.47 (PSI_S) while those for the autumn dataset ranged from −0.16 (E abundance) to −0.44 (% EPT richness). The species traits (e.g. % grazers/scrapers, % gatherers/collectors, % swimming/diving, % sprawlers/walkers and % coarse gravel taxa) explained less variance than taxonomy-based metrics and accounted for 1–8% of the variance, compared to 10–19% for the taxonomy-based metrics (Table 2). It should be borne in mind that there has been considerable debate in the scientific community about the usefulness of *P*-values in general and the position is further complicated in the case of mixed effects models. Thus the values reported here should only be used as initial rough guides.

5. Discussion

Elevated inputs of anthropogenic fine sediment is widely recognised as a significant threat to the ecological integrity of rivers (USEPA, 2002; Molinos and Donohue, 2009) resulting in changes in community structure and increased macroinvertebrate drift (Molinos and Donohue, 2009; Larsen and Ormerod, 2010; O'Callaghan et al., 2015). However, very few sediment-sensitive metrics have been developed to detect impacts due to this pervasive stressor. Furthermore, there is currently no generally accepted, standardised method for measuring deposited sediment and any method used in the field must be able to accurately estimate deposited sediment levels and be related to biological metrics.

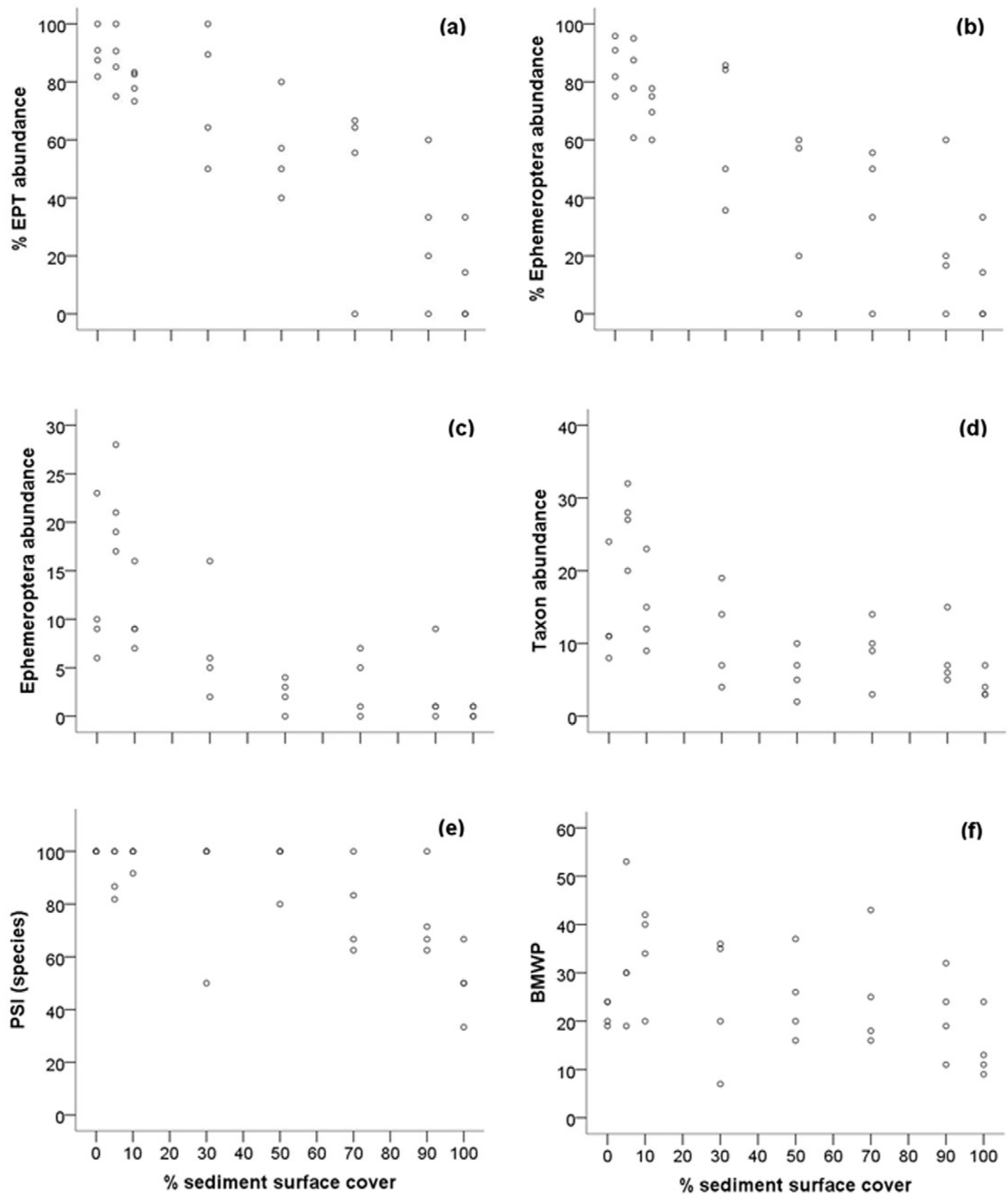

Fig. 4. Relationship between selected biotic metrics including a) % EPT abundance, b) % Ephemeroptera abundance, c) Ephemeroptera abundance, d) total taxon abundance, e) PSI species score and f) BMWP and % sediment surface cover in the mesocosm study. EPT: Ephemeroptera, Plecoptera, and Trichoptera; BMWP: Biological Monitoring Working Party; PSI: Proportion of Sediment-sensitive Invertebrates.

5.1. Responses to sediment in the mesocosm experiments and field study

As expected, macroinvertebrate drift was delayed until the hours of darkness after sediment addition: this is consistent with diurnal patterns as published in a number of other studies (e.g. Matthaei et al., 2006; Larsen and Ormerod, 2010). These responses were probably due to avoidance of impacted habitats rather than immediate behavioural displacement (Larsen and Ormerod, 2010). No significant differences were observed in drift rates between controls and high sediment treatments in the first 24 h. This is in contrast to the results obtained from a mesocosm study in Honduras (O'Callaghan et al., 2015) where taxa abundance doubled and taxa richness increased from 21% in control and low-sediment treatments to 37% in high-sediment treatments during the first 24 h following sediment addition. In the present study, there were significant differences in total abundance drifting and Heptageniidae abundance drifting over the time period of the experiment. Although the post-hoc tests could not link each of these to specific treatments, the data indicated that total drift abundance in the control channels were similar to those in low and moderate sediment treatments (5, 10 and 30% sediment surface cover). There was a 30 to 40% increase in drift rates in channels with higher sediment surface cover (50 to 100%) compared to the control. However, this was not statistically significant indicating that the different levels of sediment addition did not significantly affect the temporal pattern of macro-invertebrate drift they did affect the total abundance remaining in treatments at the end of the experiment. Suren and Jowett (2001) also reported substantial increases in drift rates for a number of species in response to sediment deposition in mesocosm channels in New Zealand while in Australia, Connolly and Pearson (2007) reported no overall differences in drift at channels ends although taxa did move a short distance downstream

Table 1
Correlations between biological metrics with sediment metrics for seasonal field data.

Metrics	% Sediment surface cover		Re-suspendable sediment (RSS)		Turbidity	
	Spring	Autumn	Spring	Autumn	Spring	Autumn
Total taxon richness	−0.25**	−0.20**	ns	ns	ns	−0.16*
Total taxon abundance	−0.24**	0.13*	0.24**	0.16*	ns	ns
% EPT richness	−0.45**	−0.44**	−0.32**	−0.21**	ns	−0.27**
% EPT abundance	−0.57**	−0.41**	−0.25**	−0.20**	ns	−0.29**
% E abundance	−0.41**	−0.23**	−0.37**	−0.20**	−0.23**	−0.17**
E abundance	−0.43**	−0.16*	−0.12*	−0.14*	−0.17*	−0.14*
ASPT	−0.40*	−0.42**	−0.27**	−0.31**	ns	−0.27**
BMWP	−0.33**	−0.33**	ns	−0.16*	ns	−0.24**
PSI_S	−0.47**	−0.26**	−0.41**	−0.25**	−0.14*	−0.15*
PSI_F	−0.40**	−0.29**	−0.32**	−0.20**	ns	−0.17*

EPT: Ephemeroptera, Plecoptera, and Trichoptera; ASPT: Average Score Per Taxon; BMWP: Biological Monitoring Working Party; PSI: Proportion of Sediment-sensitive Invertebrtes; RSS: resuspendable sediment, * $P < 0.05$, ** $P < 0.01$(one tailed), ns = not significant.

Table 2
Fraction of the variance explained by % sediment surface cover, re-suspendable sediment (g m^{-2}) and turbidity (NTU's) at patch-scale in the field study, indicated by generalised linear mixed-effects models.

Metrics	% sediment surface cover	Re-suspendable sediment (RSS)	Turbidity
Taxonomy-based metrics			
Total taxon richness	0.13***	0.04***	0.02**
Total taxon abundance	0.01*	ns	ns
% EPT richness	0.18***	0.04***	0.03***
% EPT abundance	0.13***	0.02***	0.04***
% E abundance	0.12***	0.04***	0.06***
E abundance	0.19***	0.05***	0.08***
ASPT	0.16***	0.04***	ns
BMWP	0.19***	0.06***	0.05***
PSI_S	0.17***	0.01*	ns
Trait-based metrics			
% coarse gravel taxa	0.08***	0.02**	0.02**
% grazers/scrapers	0.08***	0.02**	ns
% gatherers/collectors	0.04***	0.02**	0.03***
% shredders	0.01*	ns	0.01*
% swimming/diving	0.08***	0.03***	0.06***
% sprawlers/walkers	0.06***	0.03***	0.02**

EPT: Ephemeroptera, Plecoptera, and Trichoptera; BMWP: Biological Monitoring Working Party; ASPT: Average Score Per Taxon; PSI: Proportion of Sediment-sensitive Invertebrtes *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$; Random factors season and site, fixed effects sediment measurement.

within the experimental channels. The second hypothesis was supported as total abundance remaining in treatments at the end of experiment reduced with increasing sediment addition. Interestingly, the numbers remaining in the control (with no added fine sediment) was lower than the 5% sediment treatment, perhaps supporting the assertion that some fine sediment is required in healthy fluvial systems (Yarnell et al., 2006; Kemp et al., 2011; Jones et al., 2012).

Results from the mesocosm study showed that all metrics tested were significantly correlated with sediment surface cover but the strongest correlations were with metrics derived from the abundance of sensitive taxa (% EPT abundance, % E abundance and E abundance) and sediment surface cover. Marked decreases in % EPT abundance, % E abundance and, to a lesser degree, E abundance occurred with increasing sediment cover. These findings concur with a number of other mesocosm studies (e.g. Wagenhoff et al., 2012; Piggott et al., 2015). In the field experiments, which are representative of more realistic conditions, both total taxon abundance and richness were poorly correlated with surface cover. These findings are in general agreement with Piggott et al. (2015), who found that total taxon richness decreased as sediment cover increased whereas total taxon abundance was largely unaffected owing to increases in sediment tolerant taxa (e.g. chironomids) offsetting decreases in sensitive EPT taxa. Overall, despite the potential for confounding factors, the field observations also returned the highest association between sediment surface cover and % EPT abundance. However, the percentage of the variance in any one metric explained by sediment cover was relatively low (maximum 19%). Larsen et al. (2009) found weaker sediment effects on macroinvertebrate community in lowland catchments compared to upland areas. It was suggested that, as lowland catchments have lower diversity compared to upland catchments, observed sediment effects may be site-specific and depend on the diversity and sensitivity of species present at each site (Larsen et al., 2009). These factors may also explain the relatively low percentage variances explained for the lowland rivers in the present study.

The PSI_S, a sediment-specific metric, showed a weaker relationship with sediment cover than % EPT abundance in both the mesocosm experiment and field study. In contrast, Turley et al. (2014) found a marginally stronger relationship between the PSI metric and sediment surface cover compared to the relationships for EPT abundance and richness metrics. There was also evidence of increased variability in PSI_S scores at higher sediment loadings which concurred with Turley et al. (2014) who suggested that this increased variability may have been due to natural variability within biological communities, responses to multiple stressors and/or the quality of biological data and sediment metrics. However, as the experimental design of our mesocosm study controlled for most, if not all, of these factors it is likely that the PSI_S metric is not sufficiently specific to register sediment impact. Furthermore, the very high sediment loadings (e.g. 100%) returned substantially higher than expected PSI scores (*c.* 63) than the expected PSI score of between 0 and 20 (Extence et al., 2013). It is worth noting that, in the

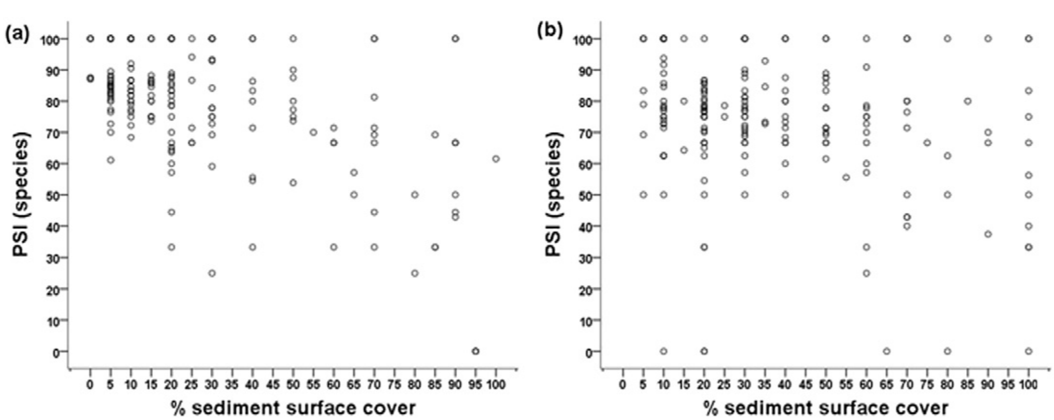

Fig. 5. Relationship between a) spring and b) autumn PSI (species) data with % sediment surface cover for field survey.

Turley et al. (2014) study, all sites were close to reference conditions and largely unimpacted by anthropogenic alterations, whereas sites in this study were lowland sites in agricultural catchments encompassing a wider diversity of environmental conditions. Despite this, surface cover explained 17% of the variance in the PSI model in the present study compared to 10.7% in another study which reviewed the relationship between the PSI metric and surface cover (Glendell et al., 2014). This higher variance in the current field study is probably explained by inclusion of all particles <2 mm as fine sediment. In contrast, while Glendell et al. (2014) assessed a number of sediment metrics and found that only % fine bed sediment cover, defined as particles <0.06 mm (and noted by the authors as difficult to accurately quantify), had a significant relationship with the PSI metric. Ongoing work in relation to the development of a new metric, E-PSI which incorporates species-specific sensitivity weightings, may help to further optimise the performance of the PSI metric (Turley et al., 2015). However, the E-PSI metric was not evaluated in the current study as it specifies a different sampling strategy to that used in the current study.

Other studies have linked changes in trait-based metrics to elevated sediment deposition (Rabení et al., 2005; Sutherland et al., 2012). Sutherland et al. (2012) found that only one functional feeding group (scrapers) was weakly related to deposited fine sediment while two habitat groups (sprawlers, swimmers) showed stronger responses. This contrasted with the results in this study, where most trait-based metrics were more weakly related to deposited sediment compared to taxonomy-based metrics. A better understanding of the mechanisms of biological response to deposited sediment would help in selecting appropriate trait-based metrics.

Visual estimates of surface cover have been described as subjective in nature and offering only a crude measure of levels of deposited fine sediment (Sutherland et al., 2012). Despite this, Sutherland et al. (2012) reported that visual sediment estimates were well correlated with, and were good predictors of seven macroinvertebrate metrics and were strongly related to riparian and catchment land cover. Furthermore, Conroy et al. (2016a, 2016b) found that surface cover estimates were strongly related to, and able to distinguish between known levels of added sediment. Zweig and Rabení (2001) found similarly strong relationships and suggested that this method was not alone more efficient in term of time and effort, but as good as, if not superior to embeddedness measurements. Results from the present study support this as most taxonomy based metrics showed moderate to strong correlations with % sediment surface cover while relationships with the two other sediment metrics (re-suspendable sediment and turbidity) were considerably weaker. These findings concur with Glendell et al. (2014) who found that PSI was not related to total suspendable bed sediment concentration. Differences in sampling resolution were cited by those authors as the most likely cause of this difference because macroinvertebrates were sampled at reach-scale while total suspendable bed sediment concentration was assessed at patch-scale at three points across the channel (Glendell et al., 2014). A number of other studies have also implied that the ability to detect impacts may be dependent on the choice of sampling scale (Townsend et al., 1997; Smiley and Dibble, 2008; Larsen et al., 2009). However, differences in sampling resolution was not a factor in the present field study as macroinvertebrates and sediment measures were taken sequentially within the frame of each Surber sampler. In effect the response to sedimentation by taxa such as EPT may be most pronounced to sediment draped on the surface of the river bed (captured in % surface cover estimates) and thus a more meaningful ecological measurement than RSS and turbidity which give a measure of sediment both draped on and deposited within the river bed.

5.2. Effects of season and taxonomic resolution on performance of biological metrics

The spring dataset showed stronger relationships with % surface cover than the autumn dataset for all but ASPT. The seasonal scatterplots indicate increased variability for autumn PSI and sediment relationships which may be due in part at least to biological responses to multiple stressors e.g. sediment and nutrients (Ormerod et al., 2010; Wagenhoff et al., 2011) in these lowland, agricultural catchments which are grazed from late spring to late autumn. Wood et al. (2011) also found seasonal variability for PSI scores in their study although no seasonal differences were detected in a separate study which examined two agricultural catchments in the UK (Glendell et al., 2014).

With regard to the effects of taxonomic resolution, the results in this study are in agreement with those of Turley et al. (2014) and Murphy et al. (2015) in that species-level identification is preferable to family/genus/order levels. The U.K. Environment Agency has also recognised the benefits of increased taxonomic resolution and its biologists are identifying taxa to species or genus level where feasible (Davy-Bowker et al., 2010). While increased taxonomic resolution may have additional time and cost implications and necessitates higher identification expertise, these considerations can be offset against the improved ability of the higher resolution metrics to highlight impacts, aiding regulatory authorities in the detection of sites which fail to meet environmental standards (Jones, 2008).

The present study explores how a number of commonly used macroinvertebrate taxonomy- and trait-based metrics respond to measures of deposited sediment using both mesocosm laboratory channels and a field study. Overall, this study has demonstrated that a range of biotic metrics respond clearly and negatively to increasing levels of deposited sediment. These results indicate that % sediment surface cover and % EPT abundance may be useful metrics for assessing the negative effect of excessive sediment on macroinvertebrates. However, variability in taxa-specific response to sedimentation indicates that refinement of biotic metrics needs to include those taxa with specific responses to sediment. This will require additional research on the mechanisms linking elevated deposited sediment levels and sediment composition to useful metrics of ecological response.

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