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Discussions

Identifying the controls of soil loss in agricultural catchments using ex situ turbidity-based suspended sediment monitoring

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Abstract

Soil erosion and suspended sediment (SS) pose risks to chemical and ecological water quality. Agricultural activities may accelerate erosional fluxes from bare, poached or compacted soils, and enhance connectivity through modified channels and artificial drainage networks. Storm-event fluxes dominate SS transport in agricultural catchments; therefore, high temporal-resolution monitoring approaches are required but can be expensive and technically challenging. Here, the performance of in situ turbidity-sensors, conventionally installed submerged at the river bankside, is compared with installations where river water is delivered to sensors ex situ, i.e. within instrument kiosks on the riverbank, at two experimental catchments (Grassland B and Arable B). Calibrated against storm-period depth-integrated SS data, both systems gave comparable results; using the ex situ and in situ methods respectively, total load at Grassland B was estimated at 128 ± 28 and 154 ± 35, and 225 ± 54 and 248 ± 52 t at Arable B. The absence of spurious turbidity peaks relating to bankside debris around the in situ sensor and its greater security, make the ex situ sensor more robust. The ex situ approach was then used to characterise SS dynamics and fluxes in five intensively managed agricultural catchments in Ireland which feature a range of landscape characteristics and land use pressures. Average annual suspended sediment concentration (SSC) was below the Freshwater Fish Directive (FFD) guideline of 25 mg L^{-1}, and the continuous hourly record demonstrated that exceedance occurred less than 12 % of the observation year. Soil drainage class and proportion of arable land were key controls determining flux rates, but all catchments reported a high degree of inter-annual variability associated with variable precipitation patterns compared to the long-term average. Poorly-drained soils had greater sensitivity to runoff and soil erosion, particularly in catchments with periods of bare soils. Well drained soils were less sensitive to erosion even on arable land; however, under extreme rainfall conditions, all bare soils remain a high sediment loss risk. Analysis of storm-period and seasonal dynam-

ics (over the long term) using high resolution monitoring would be beneficial to further explore the impact of landscape, climate and land use characteristics on SS export.

1 Introduction

Excessive supply of fine sediments (< 125 µm) and sediment-associated pollutants are detrimental to aquatic ecosystems (Wood and Armitage, 1997; Collins et al., 2011; Kemp et al., 2011). Elevated suspended sediment (SS) concentrations decrease light penetration and can reduce primary productivity. Deposition of sediments onto river channel beds also degrades habitat quality for benthic species and spawning fish (Bilotta and Brazier, 2008). In the European Union, the Water Framework Directive (WFD – OJEU, 2000) requires that water quality meet a "good" standard, but no binding environmental standards yet exist for SS across Member States (Brils, 2008; Collins and Anthony, 2008). In rivers, the EU Freshwater Fish Directive (FFD – OJEU, 2006) introduced a mean annual threshold of 25 $mg\,L^{-1}$, but this was subsequently repealed. Phosphorus (P) targets are, however, binding and because of its strong affinity for particulate transport, catchment sediment fluxes are an essential area of research.

Agriculture is commonly linked with elevated rates of soil erosion (Foster et al., 2011; Glendell and Brazier, 2014), but the degree to which sediment exports from catchments can be attributed to specific land-management practices is challenging to measure (Rowan et al., 2012). Catchments exhibit complex responses to different land uses, (e.g. arable or grazing practices) which are further influenced by climate, landscape setting and topographic controls (Wass and Leeks, 1999). A full evaluation of the extent of erosion and elevated sediment supply, therefore, requires a robust determination of the fluxes (amount and timing of sediment delivery) (Navratil et al., 2011); greater knowledge of the sources and fate of fine sediments within the system (Walling, 2005); and a better appreciation of the risks that elevated concentrations present to aquatic ecosystems (Bilotta and Brazier, 2008). This evidence base can be used to better inform land, water and sediment management strategies.

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Sediment losses from agricultural areas are commonly attributed to arable practices (Walling et al., 1999; Wass and Leeks, 1999; Freebairn et al., 2009; Van Oost et al., 2009; Duvert et al., 2010), especially where bare or freshly tilled soils are exposed to rainfall–runoff processes (Regan et al., 2012). Arable farming typically involves the mechanical redistribution of soil through ploughing and seed bed preparation and via erosion from compacted and/or bare fields and down-slope tramlines (Chambers and Garwood, 2000; Withers et al., 2006; Boardman et al., 2009; Silgram et al., 2010; Regan et al., 2012; Soane et al., 2012). Over-grazed grassland soils are also an increasingly acknowledged sediment source (Bilotta et al., 2010) and critical to the transport of particle-bound pollutants, such as P (Haygarth et al., 2006). Poaching of soils by livestock, particularly cattle wintered outside, results in loss of soil structure and compaction around gates, drinking troughs and, where access is not restricted, channel banks (Trimble and Mendel, 1995; Evans et al., 2006).

Erosion risk is conditioned by physical catchment characteristics (soil type and hydrology), and erodibility determined by topography (slope length, steepness and shape), ground cover and soil management. Soil drainage class, for example, is dictated by landscape position such that well-drained soils, such as Brown Earths and Podzols commonly located on hillslopes, contribute sediment predominantly through sub-surface pathways. Conversely, poorly-drained soils, such as Gleys (surface and groundwater) and alluvium, are at greater risk of overland-flow generation and surface soil erosion due to reduced infiltration capacity. The installation of surface and subsurface drains is also suggested to alter natural flow pathways (Ibrahim et al., 2013). Drainage installation and maintenance, for example, can result in faster quick-flow, resulting in an increased likelihood of more frequent, higher magnitude and short duration sediment transfers associated with storm runoff (Wiskow and van der Ploeg, 2003; Deasy et al., 2009; Florsheim et al., 2011).

To accurately quantify sediment fluxes from complex catchments, field monitoring programmes require three considerations. Firstly, that flow and sediment concentration data are sufficiently robust; therefore, capable of accurately describing short-term

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fluxes (Navratil et al., 2011). Secondly, the duration of the measurements must be sufficiently long to be "representative" of either stationary long-term averages (inclusive of natural variability), or to reveal temporal trends of increasing or decreasing loads or concentrations. Capturing crucial high magnitude, low recurrence interval events is, therefore, vital to generating meaningful flux determinations (Walling and Webb, 1988; Wass and Leeks, 1999). Thirdly, monitoring programmes need to be operationally cost-effective.

In-stream sampling of sediment concentrations using manual depth-integrating samplers during selected flow events to establish concentration-discharge relationships, has been widely superseded by catchment outlet, near-continuous turbidity monitoring (Lewis, 2003; Jarstram et al., 2010; Melland et al., 2012a). The latter requires turbidity sensors, loggers and infrastructure that copes with issues such as debris interference, bio-fouling, power outages and equipment/data security (Wass and Leeks, 1999; Jordan et al., 2007; Owen et al., 2012). Assessment of new monitoring strategies, compared to traditional in situ turbidity-SSC monitoring programmes, is essential to assess improvements, limitations, and validate their implementation.

There have been relatively few sediment flux investigations in Ireland (Harrington and Harrington, 2013; Melland et al., 2012a; Thompson et al., 2014). Initially regulated and managed through the Nitrates Directive (OJEU, 1991, 2007), the transfer of diffuse agricultural pollutants across the EU is now primarily integrated into obligations under the WFD. In Ireland, soil conservation issues also fall under the Nitrate Directive regulations, but the impact of SS in rivers is commonly compared to the repealed FFD target due to the absence of explicit sediment targets within the WFD. As part of an experiment to evaluate the Nitrates Directive in Ireland, a common experimental design across six agricultural catchments included high temporal-resolution measurements of river nutrient and sediment exports (Wall et al., 2011). Using these catchments and data, the aims of this study were, (1) to assess the efficacy of a novel ex situ SS monitoring technique in two catchments, and (2) to investigate annual average sediment concentrations and loads in relation to soil drainage class and land use in five monitored catchments. One catchment, situated in low-relief karst terrain was omitted from this study due to intermittent runoff combined with very low SS concentrations (cf. Mellander et al., 2012).

Study location

Suspended sediment monitoring was conducted in five catchments across Ireland (Fig. 1) which are summarised as follows:

Grassland A catchment (7.9 km^2) is located in south-west Ireland (51°38′ N 8°47′ W). Catchment soils are predominantly shallow well-drained brown earths and podzols with loam dominating the texture of A- and B-horizons, and smaller areas of surface-water gleys at the base of hillslopes. Soils are underlain by Devonian old red sandstone and mudstone from the Toe Head and Castlehaven formations (Sleeman and Pracht, 1995), which form an unconfined productive aquifer (Mellander et al., 2014). Sub-surface water pathways are therefore dominant. Land management is predominantly intensive dairy, with some beef production and minor areas of arable (Table 1).

Grassland B catchment (11.0 km^2) is located in south-east Ireland (52°36′ N, 6°20′ W). Soil type is predominantly poorly-drained groundwater gleys with a clay loam texture in A- and B-horizons, and smaller areas of well-drained brown earths confined to the upper catchment. The underlying geology is permeable, dominated by Ordovician volcanics and metasediments of the Campile formation (Tietzsch-Tyler et al., 1994), which form a productive aquifer with faults (Mellander et al., 2012). Artificial drainage is a key feature including open drains, defined here as ditches, and closed, sub-surface piped drains (predominantly 80 mm diameter). Grassland B is considered to be dominated by overland flow pathways (Shore et al., 2013; Mellander et al., 2012) except for areas of well-drained soils featuring sub-surface transport pathways. Land management is predominantly grass-based with dairy, beef and sheep enterprises (Shore et al., 2013). Arable crops such as spring barley are common on the well-drained soils.

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Grassland C catchment (3.3 km^2) is located in north-central Ireland (54°10′ N, 6°51′ W). Soils are mainly deep and moderate- to poorly-drained characterised by a loam A-horizon texture and clay loam B-horizon, and areas of shallow well-drained soils in the upper catchment areas. The geology is Silurian metasediments and volcanics of the Shercock Formation (Geraghty et al., 1997), which create an unproductive aquifer. Overland flow and near-surface pathways are, therefore, dominant here. Land use is principally grass-based for dairy, sheep and beef grazing.

Arable A catchment (11.2 km^2) is located in south-east Ireland (52°34′ N, 6°36′ W). Soils are predominantly shallow well-drained brown earths with loam texture dominating the A- and B-horizons, and limited areas of poorly-drained groundwater gleys around the stream corridor to the east of the catchment (Melland et al., 2012a). Geology comprises slate and silt stones of the Oaklands Formation (Tietzsch-Tyler et al., 1994), which produces a poorly-productive aquifer. The well-drained soils result in below-ground hydrological transfers, particularly bedrock fissure-flow (Mellander et al., 2012). Artificial drainage is limited to the poorly-drained soil areas and comprises of open ditches and sub-surface piped drainage. Land-use is dominated by spring barley with areas of permanent grassland for beef and sheep in more poorly-drained areas (Melland et al., 2012a).

Arable B catchment (9.5 km^2) is located in east-central Ireland (53°49′ N, 6°27′ W). The soil type is a complex pattern of poor- to moderately-drained soils (Melland et al., 2012a). Loam soil texture dominates the A-horizon and clay loams are dominant in the B-horizon. Soils are underlain by calcareous greywacke and banded mudstone geology (McConnell et al., 2001) and produce a poorly productive aquifer (Mellander et al., 2012). Hydrologically, surface pathways dominate; however, below-ground pathways may also be important especially during winter (Melland et al., 2012a; Mellander et al., 2012). Artificial drainage is dominant, particularly in the poorly-drained catchment areas. Arable land is dominated by winter-sown cereals, but also comprises maize and potatoes. Additional areas of permanent grassland are utilised for dairy, beef and sheep.

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2 Materials and methods

2.1 Suspended sediment monitoring

Monitoring for SS at catchment outlets was initiated in 2009 for Grassland B, Arable A and Arable B catchments and 2010 for Grassland A and Grassland C catchments. All catchments had identical instrumentation deployed for temporally high-resolution nutrient, conductivity, temperature and turbidity data capture using bankside analysers (Wall et al., 2011; Jordan et al., 2012). Turbidity (T) data were collected using a turbidity sensor (Solitax, Hach-Lange, Germany; range 0–4000 NTU; factory calibrated to 1000 NTU) and SC1000 controller at 10 min intervals. The sensors were located out-of-stream (ex situ) in a rapidly and continuously circulating header tank ($30\,m^3\,h^{-1}$) with river water delivered from the channel by an in-stream pump. Synchronised discharge data (Q – $m^3\,s^{-1}$) were calculated from converted vented pressure-transducer stage measurements (OTT Orpheus-mini; OTT Germany) rated over non-standard flat-v weirs (custom made, Corbett Concrete, Ireland).

Turbidity units (NTU) were field-calibrated to SSC ($mg\,L^{-1}$) using a combination of regular low-flow samples and intensive, discrete, high magnitude flow events with elevated SSCs. In all cases, water samples were collected from the instrument tank either manually, or using a programmable automatic water sampler (ISCO 6712; ISCO Inc. USA) with 1 m pumping tube (pump capacity $\sim 0.9\,m\,s^{-1}$) at predefined intervals of 30 or 60 min according to the specific storm characteristics. High SSC data capture was further targeted in Grassland B and Arable B using a turbidity-stratified sampling programme, thus circumventing the need to pre-set water samplers according to forecasted event characteristics. Water samples were stored at 4 °C on return to the laboratory before a sub-sample (minimum 100 mL) was processed for SSC. Whatman GF/C glass-fibre filter papers (1.2 µm) were pre-dried at 105 °C for 1 h, cooled in a desiccator and weighed before being used for vacuum filtration. Sediment concentrations were calculated from the weight of residue retained on the filter post-filtration once dried $>$ 12 h at 105 °C and cooled in a desiccator.

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2.2 Method comparison

In order to compare the ex situ sampling methodology described above with the conventional in situ monitoring approach, additional instrumentation to measure T was installed in Grassland B and Arable B from September to December 2012, and December 2012 to March 2013 respectively. A turbidimeter (T_{IN}) (Analite, McVan, Australia, range 0–1000 NTU) and automatic pumping sampler ($ISCO_{IN}$) intake were positioned in situ, adjacent to the channel edge, in proximity to the bankside analyser pump intake, but sufficiently distant not to affect, or to be affected by the ex situ instrumentation. T_{IN} and $ISCO_{IN}$ sample collection was synchronised to replicate the ex situ turbidity sensor (T_{OUT}) and pumping sampler ($ISCO_{OUT}$) programme as described above; T-SSC rating curves were developed for each sensor using water samples collected at the respective positions ($ISCO_{OUT}$ and $ISCO_{IN}$). Five storm-flow events were captured in Grassland B and two in Arable B for T-SSC calibration. Due to the location settings, the in situ automatic water sampler was fitted with a 7 m long intake tube in both catchments.

Depth integrated water samples were manually collected ($n = 225$) from a bridge over each investigated channel during flood events, using a depth-integrating SS sampler (US DH-48, Rickly Hydrological; USA). These samples were used firstly to investigate the cross-sectional variability in sediment transportation, and secondly to provide a validation dataset to assess and compare the efficacy of estimated SSC using at in situ and ex situ T sensors. Samples were collected using two strategies, (1) depth-integrated samples taken at 20 cm intervals across the channel width in rapid succession, and (2) samples taken at coarser widths and multiple depth positions. All samples were processed for SSC as described above. Due to the sampling approach used, consecutive depth-integrated samples reflected the event trend (either the rising or falling sedigraph limb) plus the cross-sectional trend. The event effect was de-trended using SSC estimated from the ex situ turbidimeter.

Where sufficient sample volume and sediment concentration existed, samples were analysed for particle size distribution using laser diffraction (Malvern Mastersizer

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2000G, Malvern, UK). Samples were circulated for 2 min (pump speed 2000 rpm, stirrer speed 800 rpm) before analysis with no pre-treatment, i.e. physical or chemical dispersant, to broadly replicate the "effective particle size" measured by the turbidity sensor. To assess the effect of automatic sampler tube length, laboratory prepared SSC samples were collected using the two intake pump lengths (1 and 7 m) used in-field. Ten 500 mL sub-samples (at 5, 10, 25, 50, 100, 250, 500, 750 and 1000 $mg L^{-1}$) were collected from homogenised 10 L mixtures using each pump length and processed for SSC.

A non-parametric Mann–Whitney *U* test was conducted to compare SSC values collected at $ISCO_{IN}$ (SSC $ISCO_{IN}$) and $ISCO_{OUT}$ (SSC $ISCO_{OUT}$), and particle size characteristics at the two study sites.

2.3 Suspended sediment rating curve construction

Data pairs for *T*-SSC calibration for each individual site (each catchment outlet over complete time series) and method comparison investigations were statistically assessed using SAS 9.3 (SAS Institute Inc., USA). Two regression equations; power (Eq. 1) and split linear (Eq. 2), were assessed using the mean square error (MSE) of the SSC predictions.

$$\text{Power} \qquad \text{SSC} = aT^{b} \tag{1}$$

$$\text{Split linear} \qquad \text{Where } T < n,\ \text{SSC} = aT$$

$$\text{Where } T > n,\ \text{SSC} = c(b_1 - b_2) + b_2 T \tag{2}$$

The intercept was set at zero for all regressions and was considered not to compromise fit at the upper end of the dataset (cf. Thompson et al., 2014). Using the selected curves, continuous turbidity measurements were computed to SSC and, using discharge data, were converted to instantaneous sediment load (SSL – $t s^{-1}$) and yield (SSY – $t km^{-2} yr^{-1}$).

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3 Results and discussion

3.1 Method comparison

Dataset completeness was similar in both T records (98–99 %); however, the timing and nature of spurious and/or missing T data were dissimilar. Spurious data at T_{IN} coincided with random peaks possibly relating to local debris interference around the sensor which is a frequent problem in T analysis (Lewis and Eads, 2001). This trend was not recorded at T_{OUT}, suggesting that the ex situ approach was less vulnerable to local in-stream debris interference (Jansson et al., 2002). Missing data at T_{IN} during periods of high sediment concentration was attributed to sensor saturation. Sporadically, pump blockages occurred in T_{OUT} due to extreme debris transport in the channel (Melland et al., 2012b). The ex situ turbidity monitoring may be at greater risk of delivery system blockages, especially during key periods of elevated turbidity and sediment transfer. These short periods are critical for sediment transport as they are responsible for the majority of the annual sediment load (Walling and Webb, 1988; Lawler et al., 2006; Estrany et al., 2009; Navratil et al., 2011). Other key issues such as bio-fouling trends were not found in either dataset, reflecting the sub-weekly frequency of maintenance at these sites.

Estimated sediment metrics (Table 2) during both monitoring periods showed discrepancies between the two measurement locations. Suspended sediment load estimated by ex situ equipment was 83 and 91 % of in situ at Grassland B and Arable B, respectively, and mean SSC at SSC_{OUT} was 85 % of SSC_{IN} at both locations. Differences in raw T output between the sensors were negated by calibration with SSC; however, the SSC of water samples from in situ (SSC $ISCO_{IN}$) and ex situ (SSC $ISCO_{OUT}$) measurement locations showed consistent differences. Samples at SSC $ISCO_{OUT}$ were 90 and 94 % of SSC $ISCO_{IN}$ at Grassland B and Arable B catchments respectively. The differences in SSC and loads between the two approaches was not statistically significant, as confirmed by the non-parametric Mann–Whitney test between SSC $ISCO_{OUT}$ and SSC $ISCO_{IN}$ ($p > 0.05$).

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Particle size analysis of event samples showed that the proportion of silt and sand particles changed through the events, whereas clay remained consistent. The greater density of sand particles compared to silts and clays, are suggested to greatly impact SSC, and are also suggested to be oversampled by pumped samples such as the $ISCO_{IN}$ approach (Horowitz, 2008). The percentage of sand (or sand-sized aggregates) between SSC $ISCO_{IN}$ and SSC $ISCO_{OUT}$ did not differ significantly ($p > 0.05$). Additionally, the ratio of the sand-sized fraction between simultaneous samples at $ISCO_{IN}$ and $ISCO_{OUT}$ showed no consistent evidence of over- or under-collection by either collection method. The hypothesis that inadequate sample collection using either method could affect SSC is unlikely, as contrasts between the sand-sized fractions seemed to be event specific. The proportion of sand-sized material collected at both $ISCO_{IN}$ and $ISCO_{OUT}$ was negatively related to Q which differs from the positive relationship found elsewhere (Grangeon et al., 2012).

Differences between SSC $ISCO_{IN}$ and SSC $ISCO_{OUT}$ could not be directly attributed to diverging particle-size of the collected samples ($p > 0.05$), or the pump length of the water sample collection ($p > 0.05$). It is possible that the proximity of the $ISCO_{IN}$ pump intake to the channel bank could influence the relationship; however, differences could additionally result from methodological dissimilarities which could not be tested in isolation, i.e. the piped-delivery of river water to the ex situ instrument tank. The impact of elevated SSCs from $ISCO_{IN}$, compared to $ISCO_{OUT}$ on the calibration of turbidity sensors T_{IN} and T_{OUT}, and the consequential prediction of high-resolution turbidity-based SSC record is discussed below.

3.2 Method validation

Samples collected from the channel cross-section were used to test the accuracy of predicted SSC using calibrated turbidity sensors at in situ and ex situ locations. The average SSC from each cross-sectional, depth-integrated set of measurements was plotted onto the rating curve over the method comparison monitoring period (Fig. 2). At Grassland B, measured SSCs plot within the 95 % confidence intervals of predicted

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SSC using both methodologies using the simultaneous T values. This trend is repeated for the majority of samples at Arable B; however, some data points plot outside of the 95 % confidence intervals for both in situ and ex situ method datasets. In the case that these out of range values were consistently higher or lower than the predicted values, this may suggest a systematic error due to sampling strategy; however, both upper and lower confidence limits were exceeded by the SSC values (Fig. 2c and d). Therefore, the error associated with the measurement method was generally less than that encapsulated within the 95 % prediction intervals of the T to SSC calibration curve and consequently, both measurement approaches can be accepted as accurate for the estimation of SS metrics in these catchments.

3.3 Suspended sediment metrics in five agricultural catchments

High magnitude SSCs were of short duration in all five catchments (Fig. 3), but such periods are typically critical to cumulative annual SSY (Walling and Webb, 1988; Navratil et al., 2011). Grassland B and Arable B had a large proportion (80 % of the monitoring period) of sediment transported at SSCs between 1 and 10 $mg\,L^{-1}$, and shorter periods of concentrations ≥ 10 $mg\,L^{-1}$ for 15 and 20 % of the monitoring period respectively (Fig. 3). In the remaining catchments, low concentrations of < 1 $mg\,L^{-1}$ were more common and occurred over 50 % of the time. High concentrations (≥ 10 $mg\,L^{-1}$) were limited to only 10 % of the monitoring period. Overall, however, the FFD average annual SSC guideline was not exceeded in any monitoring year in any of the catchments (Table 3). The highest mean SSC of up to 17 $mg\,L^{-1}$ was recorded at Grassland B and Arable B and the remaining catchments reported very low values of < 6 $mg\,L^{-1}$. Accordingly, the instantaneous exceedance of the FFD guideline (Table 3) occurred during extremely short time periods (1–11 % of sampled time per year). The values here are similar to those reported by Thompson et al. (2014) in two other predominantly improved grassland catchments in Ireland; 8 % exceedance was reported in a moderately-drained catchment in Co. Down and 18 % exceedance in a poorly-drained catchment in Co. Louth. Although instantaneous FFD exceedance metrics have been

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reported in other sediment studies (Glendell et al., 2014; Peukert et al., 2014; Thompson et al., 2014), the transferability of such coarse thresholds (compliance to which requires an undefined annual sample number) to high-resolution SS data is questionable.

Average SSYs in the five catchments were 8.5, 24.7, 11.6, 12.0 and 24.4 $t\,km^{-2}\,yr^{-1}$ at Grassland A, Grassland B, Grassland C, Arable A and Arable B respectively. Figure 4 illustrates average annual SSYs from Ireland, the UK and the wider Atlantic climatic region of Europe (Vanmaercke et al., 2011). These values align with existing data on SSY in Ireland (cf. Huang and O'Connell, 2000; Jordan et al., 2002; Harrington and Harrington, 2013; Thompson et al., 2014), and are consistently low compared with the UK and Europe. Considering the agricultural intensity of these catchments, (for example, Grassland A is within the highest region of milk yield in Ireland (Läppe and Hennessy, 2012), and crop yields across Ireland are internationally high, Melland et al., 2012a), these values are particularly low. Catchment observations suggest that high landscape complexity comprising small (low runoff length) and irregularly shaped fields, separated by hedgerows and vegetated ditches, contribute to lower water and sediment connectivity between hillslopes and the channel network.

In the UK, Cooper et al. (2008) suggested annual "target" and threshold "investigation" SSY values be based upon drainage class and catchment terrain characteristics. Grassland A and Arable A qualify as lowland well-drained catchments and, on average, fall well below target and investigation SSY of 20 and 50 $t\,km^{-2}\,yr^{-1}$, respectively. Grassland B, Grassland C and Arable B, categorised as lowland predominantly poorly-drained catchments, on average, fall below target and investigation thresholds of 40 and 70 $t\,km^{-2}\,yr^{-1}$, respectively. Total SSY data for individual years (Table 3), however, indicate variability and exceeded respective SSY target values; Grassland B in 2009 and 2012, Arable A 2012 and Arable B in 2011 and 2012.

Higher average SSC, intra-annual period of FFD exceedance, and average SSY in catchments Grassland B and Arable B are suggested to result from poorer soil drainage. During rainfall events, soils are rapidly saturated and critical overland flow pathways established; consequently, eroded particles within these connected areas are transported through the catchment (Mellander et al., 2012; Shore et al., 2013). The SSC responses here suggest, as in other catchments with impeded drainage, that high overland-flow potential is also associated with a notable proportion of sediment delivered at lower concentrations over a longer period, through surface and sub-surface flow pathways (Deasy et al., 2009; Melland et al., 2012a) resulting in increased average SSCs. In catchments Grassland A and Arable A, sub-surface flow pathways dominate, due to well-drained soils reducing the likelihood of overland flow and consequently surface soil losses. Consequently, SSCs, intra-annual period of FFD exceedance, and SSYs were low. Conversely, Grassland C more accurately reflects the sediment characteristics of the well-drained catchments despite the moderate- to poorly-drained soils. Near complete cover of permanent pasture here was considered to sufficiently reduce sediment source availability and transport of sediment to the watercourse.

Generalisations can be made in relation to the overriding controls on SSY across the monitored catchments (Fig. 5). Inter-catchment comparisons used data from hydrological years 2010 to 2013, where data were available for all five catchments. Sediment delivery was enhanced by the combined effect of an overland-flow dominated transport system (poorly-drained soils) and, to a lesser extent, source availability (arable soils with potentially lengthy periods of bare ground cover, Regan et al., 2012 or seasonally thinly vegetated grassland soils cf. Bilotta et al., 2010). Catchments that possess better drainage characteristics and/or permanent crop cover have greater resilience to extreme sediment losses. In catchments such as Arable A, where good-drainage is combined with high source availability, the risk associated with sediment transport during extreme rainfall events and years was, however, high. Similarly, poorly-drained soils stabilised by permanent pasture should be maintained and periods of bare cover should be avoided.

High inter-annual variability was evident, particularly with regard to SSY (Table 3). The annual SSY CV% were 67, 76, 79, 83 and 50 % in Grassland A, Grassland B, Grassland C, Arable A and Arable B, respectively. Notably, in Grassland B and Arable

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B catchments, the inter-annual SSY ranges of 41.7, and 26.2 $\text{t km}^{-2}\,\text{yr}^{-1}$, respectively, were greater than average annual inter-catchment SSY of 24.0 $\text{t km}^{-2}\,\text{yr}^{-1}$. The variability found within each of the five monitoring catchment was comparable to the results of Vanmaercke et al. (2012) who reported CV% ranging from 6–313 % (median 75 %) in 726 catchments worldwide. The catchment with the lowest inter-annual SSY (11.0 $\text{t km}^{-2}\,\text{yr}^{-1}$), Grassland A, received the least variable rainfall input and total discharge.

Inter-annual SSY variability results from strong seasonality combining the timing and character of rainfall events in relation to soil moisture deficit and land management; this in turn conditions sediment availability in critical source areas. Analysis of shorter term sediment losses i.e. at seasonal, monthly and event scales would also provide empirical evidence to inform both high level policy considerations and local decision making. Additionally, assessment of seasonal transfers are likely to have greater ecological significance as mean annual thresholds such as SSC (through the FFD), and SSY may underestimate the seasonality of risk of sediments to aquatic ecosystems (Thompson et al., 2014). Sensitivity to sediment is species-specific and dependent upon life stage (Collins et al., 2011); therefore, shorter-term metrics such as the timing, magnitude, duration and frequency of sediment transfers are important concepts to consider. Existing static thresholds may, therefore, be considered ecologically irrelevant, particularly when utilised as an instantaneous threshold for high-resolution data. Future discussion regarding sediment targets requires an assessment of multiple species and habitat quality. This task is particularly complicated where ecological condition is subject to multiple-stressors such as nutrients (Bilotta and Brazier, 2008), bed substrate quality (Kemp et al., 2011) and time lag (Fenton et al., 2011; Vero et al., 2014).

Overall, the annual average sediment metrics reported here are internationally low. Considering the spatial dominance and intensity of agricultural land use and high effective rainfall in the study catchments, this is perhaps unexpected. As previously discussed, the complexity of landscape features (e.g. fields, hedgerows, ditches) can be expected to decrease the likelihood of field-scale soil erosion, and/or increase the opportunity for interception and deposition of mobile particles, i.e. reducing the sediment delivery ratio by retaining sediment on the land or within the hydrological network (Borselli et al., 2008). The Irish landscape may, therefore, improve the resilience of agricultural soils to soil loss. However, even from modest SSY, the potential for other specific risks to ecologically sensitive habitats, from SS deposition in rivers for example, will need a cautionary approach. Therefore, identification of the specific mechanisms promoting soil conservation or sediment retention in multiple catchments with contrasting physical and land use characteristics will be important. This is particularly relevant for water and agricultural policy, as the prevention of environmental degradation and maintenance and/or sustainable intensification of agricultural production are simultaneously considered. Furthermore, other sediment sources, for example, from channel banks and road networks may contribute significant proportions of the annual load (Collins et al., 2013; Rowan et al., 2012; Sherriff et al., 2014). Assessment of such sources could be a useful insight to prioritise sediment management strategies (Wilson et al., 2008).

4 Conclusions

This study assessed the accuracy and reliability of an ex situ, turbidity-based methodology to estimate suspended sediment fluxes in multiple monitored catchments. Applying the method, annual SSC, FFD exceedance and SSY data in five catchments were further investigated in relation to physical catchment characteristics and land management. The key findings were:

- Suspended sediment metrics between in situ and ex situ methodologies were not significantly different from in-stream cross-sectional, depth-integrated samples in two monitoring catchments.

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- The ex situ methodology reported less sensitivity to spurious data peaks; however, periods of extreme large debris transport increased the sensitivity of the ex situ instrumentation to short-term blockages.
- All catchments reported mean annual SSCs of less than the FFD threshold of $25\,\mathrm{mg\,L^{-1}}$ and short-term exceedance of 1–11 % of sampled time.
- Inter-annual variability of SSY was strong due to the seasonality of timing and character of rainfall events in relation to land management.
- Average annual SSYs in all five Irish catchments reported here were low in comparison to equivalent catchments and landscape settlings elsewhere in Europe. Farming practices favouring relatively small fields, a high density of field boundaries including ditches, with low consequent connectivity are likely to explain this.
- Within the study catchments, SSY was higher in catchments dominated by poorly-drained soils than those with well-drained soils. Furthermore, on poorly-drained soils, catchments coincident with a greater proportion of arable land use reported the highest annual average SSY.
- The sediment loss risk on well drained soils did, however, show the potential to supply significant quantities of sediment when extreme climatic conditions coincided with bare soils.
- Complexity of the landscape may provide resilience to soil erosion and/or sediment transport despite spatial dominance and intensity of agriculture and these will be important considerations for future management (such as sustainable intensification) and/or SS mitigation in Ireland and elsewhere.

These findings illustrate that interactions between climate, landscape and land use regulate the supply of sediments from Irish agricultural catchments. Whilst the current SSYs are low by international standards, key questions still remain regarding the magnitude and frequency characteristics of sediment transfers at shorter timescales. This includes both seasonal and storm-event scale, which are important to inform erosion risk and sediment pulses moving into the channel network within ecologically sensitive periods. Further to this, seasonal sediment provenance and field-scale soil loss assessments within this land management and landscape framework are crucial to quantify the contributions made from specific agricultural and other sediment sources.

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Table 1. Summary of study catchments.

Catchment	Size (km^2)	30 year average rainfall ($mm\,yr^{-1}$)	Median slope (°)	Dominant soil drainage class/ **flow pathway**	Land-use
Grassland A	7.9	1228	4	Well-drained **Sub-surface**	89 % grassland predominantly for dairy; 5 % arable
Grassland B	11.5	906	3	Poorly-drained **Surface**	77 % grassland for dairy, beef and sheep; 12 % spring crops 2 % winter crops
Grassland C	3.3	960	6	Poorly-drained **Surface**	94 % grassland for beef, dairy and sheep
Arable A	11.2	906	3	Well-drained **Sub-surface**	54 % arable predominantly spring crops; 39 % grass mainly beef and sheep
Arable B	9.4	758	3	Moderately- to poorly-drained **Surface**	24 % winter crops; 29 % grazing for beef and sheep; 19 % dairy grazing

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Table 2. Suspended sediment metrics estimated using in-situ and ex-situ methods.

Catchment	Total load (t)[a]		Mean concentration (mg L^{-1})		Max concentration (mg L^{-1})	
	SSL_{OUT}	SSL_{IN}	SSC_{OUT}	SSC_{IN}	SSC_{OUT}	SSC_{IN}
Grassland B	128 ± 28	154 ± 35	13.7	16.2	1010	1188
Arable B	225 ± 54	248 ± 52	29.1	34.1	2043	899[b]

[a] Confidence intervals are the coefficient of variance of the mean prediction.
[b] T_{IN} sensor saturated at 1000 NTU.

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Table 3. Annual rainfall, discharge and suspended sediment flux summary for five catchments. Monitoring years correspond to hydrologic years (October to September).

	Grassland A			Grassland B				Grassland C			Arable A				Arable B			
Year	2010	2011	2012	2009	2010	2011	2012	2010	2011	2012	2009	2010	2011	2012	2009	2010	2011	2012
Rainfall ($mm\,yr^{-1}$)	1045	1139	1097	1278	800	1155	920	965	1234	969	1240	763	1102	827	896	742	1049	844
Runoff ($mm\,yr^{-1}$)	443	633	608	643	330	504	382	424	727	575	750	366	517	473	383	319	521	542
Mean SSC ($mg\,L^{-1}$)	4.60	3.88	5.15	14.32	5.48	7.64	11.65	4.42	4.09	3.48	5.95	2.60	4.07	5.58	9.36	9.60	10.42	17.42
Max SSC ($mg\,L^{-1}$)	707	467	966	1020	426	882	707	419	813	462	773	224	737	2141	494	707	688	1120
$> 25\,mg\,L^{-1}$ (% of ST*)	3.22	2.25	3.08	11.29	4.88	5.64	7.78	2.38	2.39	2.11	3.84	1.14	1.91	2.77	6.34	6.18	6.12	11.30
SSY ($t\,km^{-2}\,yr^{-1}$)	3.95	6.61	14.92	48.39	6.65	13.46	30.08	6.07	22.28	6.52	17.44	2.11	5.22	23.10	15.59	15.97	24.20	41.81

ST* = sampled time.

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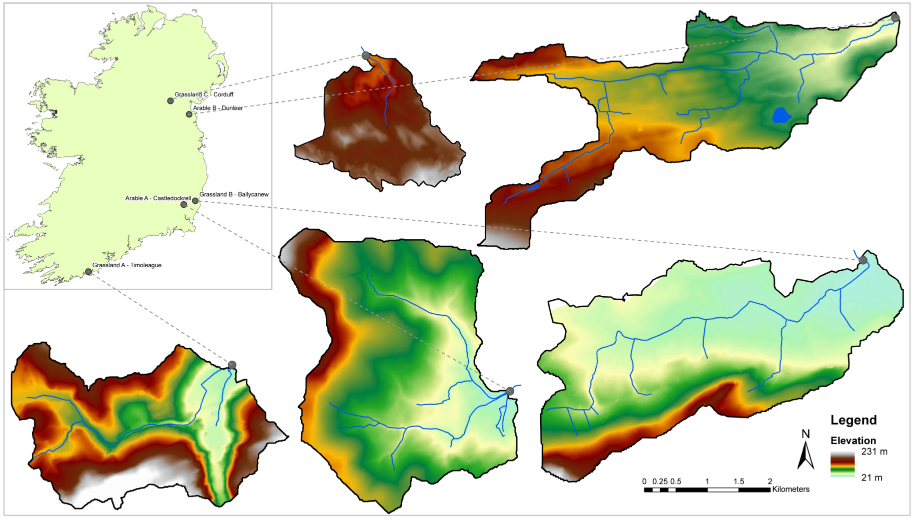

Figure 1. Map of catchment monitoring locations and study catchments with topographic information.

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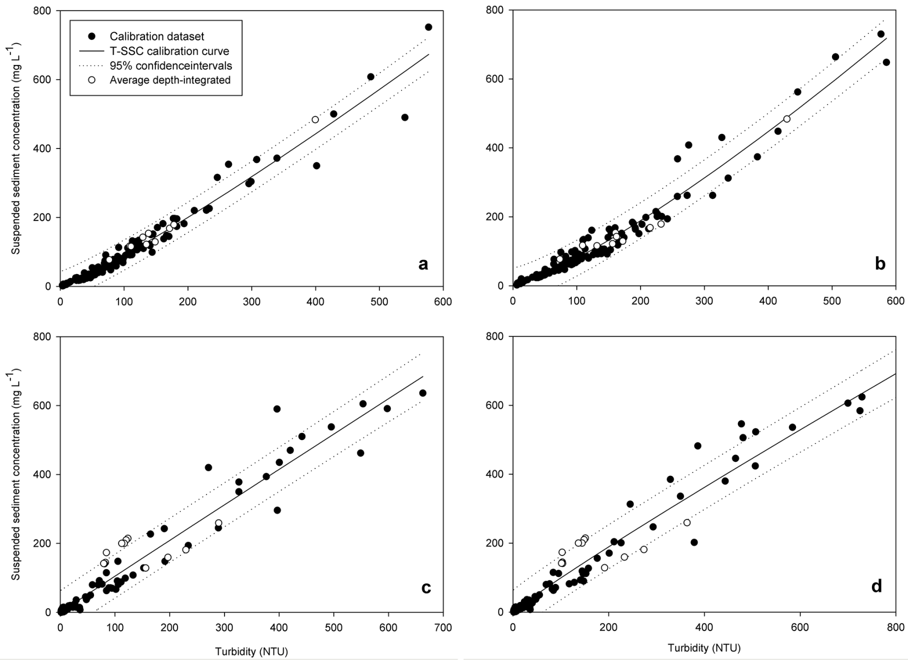

Figure 2. Turbidity-suspended sediment concentration rating curves, confidence intervals, calibration data and cross-section averaged depth-integrated suspended sediment concentration samples for, **(a)** Grassland B T_{OUT}, **(b)** Grassland B T_{IN}, **(c)** Arable B T_{OUT}, **(d)** Arable B T_{IN}.

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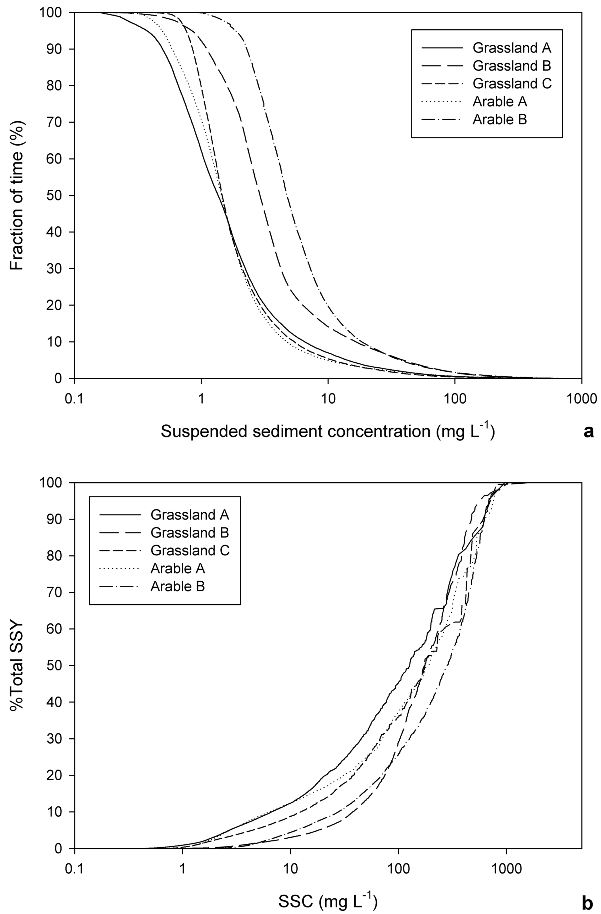

Figure 3. Frequency-duration graphs of, **(a)** suspended sediment concentration with time and, **(b)** percentage of suspended sediment yield with suspended sediment concentration.

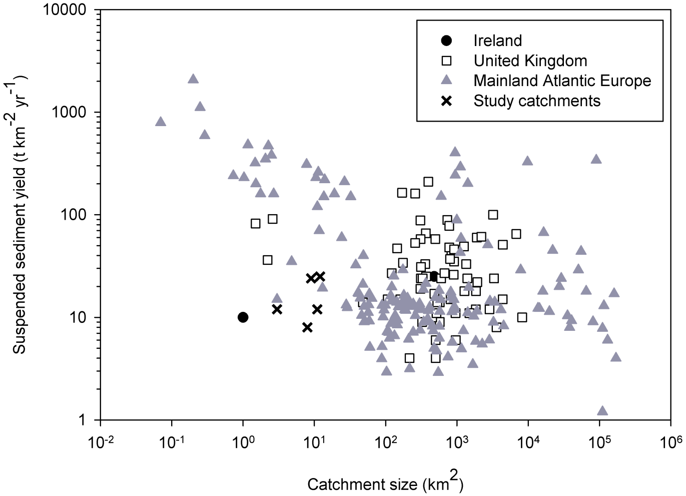

Figure 4. Catchment size and suspended sediment yield of European river catchments. Sources: Foster et al. (1986); Milliman and Syvitski (1992); McManus and Duck (1996); Wass and Leeks (1999); Huang and O'Connell (2000); Verstraeten and Poesen (2001); Jordan et al. (2002); Walling et al. (2002); Harlow et al. (2006); Oeurng et al. (2010); Zabaleta et al. (2007); Gay et al. (2014).

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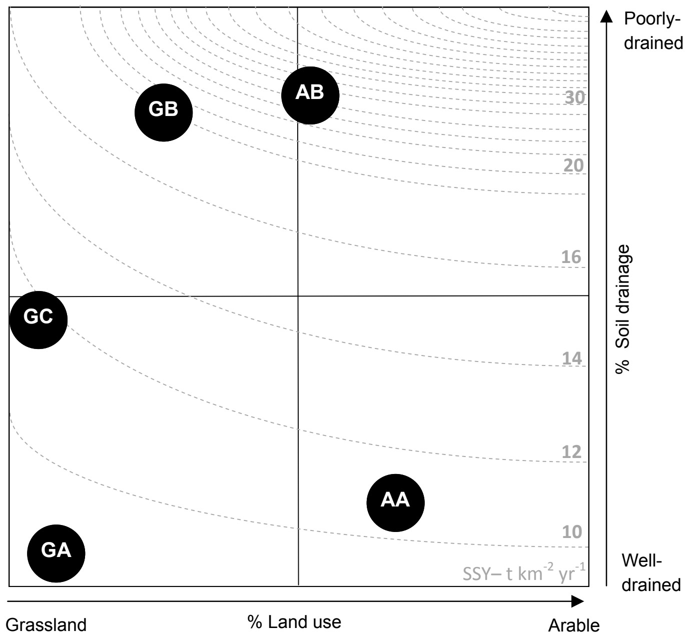

Figure 5. Conceptual diagram of suspended sediment yield as represented by iso-lines according to land use and dominant soil drainage class. Catchment abbreviations: GA – Grassland A, GB – Grassland B, GC – Grassland C, AA – Arable A, AB – Arable B.

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