

1 Investigating suspended sediment dynamics in contrasting agricultural catchments using ex situ turbidity-based  
2 suspended sediment monitoring

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14 Abstract

15 Soil erosion and suspended sediment (SS) pose risks to chemical and ecological water quality. Agricultural  
16 activities may accelerate erosional fluxes from bare, poached or compacted soils, and enhance connectivity  
17 through modified channels and artificial drainage networks. Storm-event fluxes dominate SS transport in  
18 agricultural catchments; therefore, high temporal-resolution monitoring approaches are required but can be  
19 expensive and technically challenging. Here, the performance of in situ turbidity-sensors, conventionally  
20 installed submerged at the river bankside, is compared with installations where river water is delivered to  
21 sensors ex situ, i.e. within instrument kiosks on the riverbank, at two experimental catchments (Grassland B and  
22 Arable B). The in-situ and ex-situ installations gave comparable results when calibrated against storm-period,  
23 depth-integrated SS data, with total loads at Grassland B estimated at 12828 t and 15435 t, and 22554 t and  
24 24852 t at Arable B, respectively. The absence of spurious turbidity **readings** relating to bankside debris around  
25 the in situ sensor and its greater security, make the ex situ sensor more robust. The ex situ approach was then  
26 used to characterise SS dynamics and fluxes in five intensively managed agricultural catchments in Ireland  
27 which feature a range of landscape characteristics and land use pressures. Average annual suspended sediment  
28 concentration (SSC) was below the Freshwater Fish Directive (78/659/EEC) guideline of 25 mg L<sup>-1</sup>, and the  
29 continuous hourly record demonstrated that exceedance occurred less than 12% of the observation year. Soil  
30 drainage class and proportion of arable land were key controls determining flux rates, but all catchments  
31 reported a high degree of inter-annual variability associated with variable precipitation patterns compared to the  
32 long-term average. Poorly-drained soils had greater sensitivity to runoff and soil erosion, particularly in  
33 catchments with periods of bare soils. Well drained soils were less sensitive to erosion even on arable land;  
34 however, under extreme rainfall conditions, all bare soils remain a high sediment loss risk. Analysis of storm-  
35 period and seasonal dynamics (over the long term) using high resolution monitoring would be beneficial to  
36 further explore the impact of landscape, climate and land use characteristics on SS export.

37

38 Keywords: Suspended sediment; stage-discharge; turbidity calibration; flux determinations; agricultural land use

39 1 Introduction

40 Excessive supply of fine sediments (<125 µm) and sediment-associated pollutants are detrimental to aquatic  
41 ecosystems (Wood and Armitage, 1997; Collins et al., 2011; Kemp et al., 2011). Elevated suspended sediment  
42 (SS) concentrations decrease light penetration and can reduce primary productivity. Deposition of sediments  
43 onto river channel beds also degrades habitat quality for benthic species and spawning fish (Bilotta and Brazier,  
44 2008). In the European Union, the Water Framework Directive (WFD – OJEU, 2000) requires that water quality  
45 meet a “good” standard, but no binding environmental standards yet exist for SS across Member States (Brils,  
46 2008; Collins and Anthony, 2008). In rivers, the EU Freshwater Fish Directive (FFD – OJEU, 2006) introduced  
47 a mean annual threshold of 25 mg L<sup>-1</sup>, but this was subsequently repealed. Phosphorus (P) targets are, however,  
48 binding and because of its strong affinity for particulate transport, catchment sediment fluxes are an essential  
49 area of research.

50

51 Agriculture is commonly linked with elevated rates of soil erosion (Foster et al., 2011; Glendell and Brazier,  
52 2014), but the degree to which sediment exports from catchments can be attributed to specific land-management  
53 practices is challenging to measure (Rowan et al., 2012). Catchments exhibit complex responses to different  
54 land uses, (e.g. arable or grazing practices) which are further influenced by climate, landscape setting and  
55 topographic controls (Wass and Leeks, 1999). A comprehensive evaluation of the extent of erosion and elevated  
56 sediment supply, therefore, requires a robust determination of sediment flux (Navratil et al., 2011), knowledge  
57 of the sources and fate of fine sediments within the system (Walling, 2005), and an appreciation of the risks that  
58 elevated concentrations present to aquatic ecosystems (Bilotta and Brazier, 2008). This evidence base can be  
59 used to better inform integrated land, water and sediment management strategies.

60

61 Sediment losses from agricultural areas are commonly attributed to arable practices (Walling et al., 1999; Wass  
62 and Leeks, 1999; Freebairn et al., 2009; Van Oost et al., 2009; Duvert et al., 2010), especially where bare or  
63 freshly tilled soils are exposed to rainfall-runoff processes (Regan et al., 2012). Arable farming typically  
64 involves the mechanical redistribution of soil through ploughing and seed bed preparation, and via erosion from  
65 compacted and/or bare fields and down-slope tramlines (Chambers and Garwood, 2000; Withers et al., 2006;  
66 Boardman et al., 2009; Silgram et al., 2010; Regan et al., 2012; Soane et al., 2012). Over-grazed grassland soils  
67 are also an important sediment source (Bilotta et al., 2010) and critical to the transport of particle-bound  
68 pollutants, such as P (Haygarth et al., 2006). Poaching of soils by livestock, particularly cattle wintered outside,

69 results in loss of soil structure and compaction around gates, drinking troughs and, where access is not  
70 restricted, channel banks (Trimble and Mendel, 1995; Evans et al., 2006).

71

72 Erosion risk is conditioned by physical catchment characteristics (soil type and hydrology), and erodibility  
73 determined by physiography (slope length, steepness and shape, ground cover and soil management). Soil  
74 drainage class, for example, is dictated by landscape position whereby well-drained soils, such as Brown Earths  
75 and Podzols commonly located on hillslopes, contribute sediment predominantly through sub-surface pathways  
76 such as relocation of fine surface sediments vertically and/or horizontally through the soil profile, preferential  
77 flow through macropores (Chapman et al., 2001; Deasy et al., 2009). Conversely, poorly-drained soils, such as  
78 Gleys (surface and groundwater) and silt and clay dominated alluvial soils in proximity to watercourses, are at  
79 greater risk of overland-flow generation and surface soil erosion due to reduced infiltration capacity. The  
80 installation of surface and sub-surface drains can also alter natural flow pathways (Ibrahim et al., 2013).  
81 Drainage installation and maintenance, for example, can result in faster quick-flow, resulting in an increased  
82 likelihood of more frequent, higher magnitude and short duration sediment transfers associated with storm  
83 runoff (Wiskow and van der Ploeg, 2003; Deasy et al., 2009; Florsheim et al., 2011).

84

85 To accurately quantify sediment fluxes from complex catchments, field monitoring programmes require three  
86 considerations. Firstly, robust flow and sediment concentration data capable of accurately describing short-term  
87 fluxes (Navratil et al., 2011). Secondly, the duration of the measurements must be sufficiently long to be  
88 'representative' of either stationary long-term averages (inclusive of natural variability), or to reveal temporal  
89 trends of increasing or decreasing loads or concentrations. Capturing crucial high magnitude, low frequency  
90 events is, therefore, vital to generating meaningful flux determinations (Walling and Webb, 1988; Wass and  
91 Leeks, 1999). Thirdly, monitoring programmes need to be operationally cost-effective.

92

93 Sediment load estimation based on SSC -discharge rating curves has been widely superseded by catchment  
94 outlet, near-continuous turbidity monitoring (Lewis, 2003; Jarstram et al., 2010; Melland et al., 2012a). The  
95 latter requires turbidity sensors, loggers and infrastructure that cope with issues such as debris interference, bio-  
96 fouling, power outages and equipment/data security (Wass and Leeks, 1999; Jordan et al., 2007; Owen et al.,  
97 2012). Assessment of new monitoring strategies, compared to traditional in situ turbidity-SSC monitoring  
98 programmes, is essential to assess improvements, limitations, and validate their implementation.

100 There have been relatively few sediment flux investigations in Ireland (Melland et al., 2012a; Harrington and  
101 Harrington, 2013; Thompson et al., 2014). Initially regulated and managed through the Nitrates Directive  
102 (OJEU, 1991; 2007), the transfer of diffuse agricultural pollutants across the EU is now primarily integrated into  
103 obligations under the WFD. In Ireland, soil conservation issues also fall under the Nitrates Directive regulations,  
104 but the impact of SS in rivers is commonly compared to the repealed FFD target due to the absence of explicit  
105 sediment targets within the WFD.

106

107 As part of an experiment to evaluate the Nitrates Directive in Ireland, a common experimental design across six  
108 agricultural catchments included high temporal-resolution measurements of river nutrient and sediment exports  
109 (Wall et al., 2011). Using these catchments and data, the aims of this study were, (1) to assess the efficacy of a  
110 novel ex situ SS monitoring technique in two catchments, and (2) to investigate annual average sediment  
111 concentrations and loads in relation to soil drainage class and land use in five monitored catchments. One  
112 catchment, situated in low-relief karst terrain was omitted from this study due to intermittent runoff combined  
113 with very low SS concentrations (cf. Mellander et al., 2012).

## 114 2 Study location

115 Suspended sediment monitoring was conducted in five catchments (Table 1) across Ireland (Fig. 1). Catchments  
116 were selected to represent the main intensive agricultural land use types in Ireland, dominant hydrological  
117 pathways (surface or sub-surface) at a scale where headwater to channel hydrological process were detectable  
118 (Fealy et al., 2010). The characteristics of individual catchments are summarised as follows:

119

120 Grassland A catchment (7.9 km<sup>2</sup>) is located in south-west Ireland (51°38'N 8°47'W). Catchment soils are  
121 predominantly shallow well-drained Brown Earths and Podzols with loam dominating the texture of A- and B-  
122 horizons, and smaller areas of surface-water Gleys at the base of hillslopes. A coarse loamy drift with siliceous  
123 stone subsoil is underlain by Devonian old red sandstones and mudstones from the Toe Head and Castlehaven  
124 formations (Sleeman and Pracht, 1995), which form an unconfined productive aquifer (Mellander et al., 2014).  
125 Sub-surface water pathways are therefore dominant. Land is predominantly grazed by cattle for intensive dairy  
126 production and smaller areas of beef production with an average catchment stocking rate of 1.98 livestock units  
127 (LU) ha<sup>-1</sup> additionally minor areas of arable land use are present (Table 1).

128

129 Grassland B catchment (11.0 km<sup>2</sup>) is located in south-east Ireland (52°36'N, 6°20'W). Soil type is  
130 predominantly poorly-drained Groundwater Gleys in the catchment lowlands with a clay loam texture in A- and  
131 B-horizons resulting from a clayey calcareous Irish Sea till subsoil. The uplands contain smaller areas of well-  
132 drained Brown Earths, these soils are underlain by drift deposits with siliceous stones. The underlying geology  
133 is permeable, dominated by Ordovician volcanics and metasediments of the Campile formation (Tietzsch-Tyler  
134 et al., 1994), which form a productive aquifer with faults (Mellander et al., 2012). Artificial drainage is a key  
135 feature including open drains, defined here as ditches, and closed, sub-surface piped drains (predominantly 80  
136 mm diameter). Grassland B is considered to be dominated by overland flow pathways (Mellander et al., 2012;  
137 Shore et al., 2013) except for areas of well-drained soils featuring sub-surface transport pathways. Land is  
138 predominantly grass-based for dairy and beef cattle grazing, and also sheep enterprises (Shore et al., 2013) with  
139 stocking rate of 1.04 LU ha<sup>-1</sup>. Arable crops such as spring barley are common on the well-drained soils which  
140 are unmanaged between harvest and ploughing for following crop.

141

142 Grassland C catchment (3.3 km<sup>2</sup>) is located in north-east Ireland (54°01'N, 6°51'W). Soils are mainly deep and  
143 moderate- to poorly-drained characterised by a loam A-horizon texture and clay loam B-horizon, and areas of

144 shallow well-drained soils in the upper catchment areas underlain predominately by Lower Palaeozoic shale  
145 tills. The geology is Silurian metasediments and volcanics of the Shercock Formation (Geraghty et al., 1997),  
146 which create an unproductive aquifer. Overland flow and near-surface pathways are, therefore, dominant here.  
147 Land use is principally grass-based for dairy cattle, sheep and beef cattle grazing (stocking rate 1.00 LU ha<sup>-1</sup>).

148  
149 Arable A catchment (11.2 km<sup>2</sup>) is located in south-east Ireland (52°34'N, 6°36'W). Soils are predominantly  
150 shallow well-drained Brown Earths with loam texture dominating the A- and B-horizons, and limited areas of  
151 poorly-drained Groundwater Gleys around the stream corridor to the east of the catchment (Melland et al.,  
152 2012a). Subsoils predominantly comprise fine loamy drift with siliceous stones over slate and silt stones of the  
153 Oaklands Formation (Tietzsch-Tyler et al., 1994), which produces a poorly-productive aquifer. The well-drained  
154 soils result in below-ground hydrological transfers, particularly bedrock fissure-flow (Mellander et al., 2012).  
155 Artificial drainage is limited to the poorly-drained soil areas and comprises of open ditches and sub-surface  
156 piped drainage. Land use is dominated by spring barley (land is unmanaged between cropping cycles and crop  
157 rotation is limited) with areas of permanent grassland for beef cattle and sheep grazing in more poorly-drained  
158 areas (Melland et al., 2012a) at 0.40 LU/ha.

159  
160 Arable B catchment (9.5 km<sup>2</sup>) is located in north-east Ireland (53°49'N, 6°27'W). The soil type is a complex  
161 pattern of poor- to moderately-drained soils (Melland et al., 2012a). Loam soil texture dominates the A-horizon  
162 and clay loams are dominant in the B-horizon. Subsoil is dominated by fine till containing siliceous stones with  
163 fluvioglacial sediments located near-channel. Soils are underlain by calcareous greywacke and banded mudstone  
164 geology (McConnell et al., 2001) and produce a poorly productive aquifer (Mellander et al., 2012).  
165 Hydrologically, surface pathways dominate; however, below-ground pathways may also be important especially  
166 during winter (Melland et al., 2012a; Mellander et al., 2012). Artificial drainage is dominant, particularly in the  
167 poorly-drained catchment areas. Arable land is dominated by winter-sown cereals, but also comprises maize and  
168 potatoes. These areas are unmanaged between cropping cycles; however, crop rotation is more common than at  
169 Arable A due to the wider range of crop types. Additional areas of permanent grassland are utilised for dairy  
170 cattle, beef cattle, and sheep grazing (0.77 LU ha<sup>-1</sup>).

171 3 Materials and methods

172 3.1 Suspended sediment monitoring

173 Monitoring for SS at catchment outlets was initiated in 2009 for Grassland B, Arable A and Arable B  
174 catchments and 2010 for Grassland A and Grassland C catchments. All catchments had identical instrumentation  
175 deployed for temporally high-resolution nutrient, conductivity, temperature and turbidity data capture using  
176 bankside analysers mains powered at 230V (**Fig. 2** - Wall et al., 2011; Jordan et al., 2012; Melland et al.,  
177 2012b). Turbidity (T) data were collected using a turbidity sensor (Solitax, Hach-Lange, Germany; range 0-4000  
178 NTU; factory calibrated to 1000 NTU) and SC1000 controller at 10 min intervals. The sensors were located out-  
179 of-stream (ex situ) in a rapidly and continuously circulating header tank with river water delivered from the  
180 channel by an in-stream pump ( $30 \text{ m}^3 \text{ hr}^{-1}$ ) located on the channel bed. The instrument tank was assumed well-  
181 mixed as no particulate deposition occurred. Turbidity probes were fitted with wipers to prevent biological  
182 fouling, and checked monthly against deionised water (0 NTU) and a 20 NTU Formazin turbidity standard.  
183 Synchronised discharge data ( $Q - \text{m}^3 \text{ s}^{-1}$ ) were calculated from vented pressure-transducer stage measurements  
184 (OTT Orpheus-mini; OTT Germany). Stage height was converted to Q using velocity-area measurements (OTT  
185 Acoustic Doppler Current meter; OTT Germany) collected over non-standard flat-v weirs (custom made, Corbett  
186 Concrete, Ireland) and WISKI-SKED software (Grassland A,  $R^2=0.96$ ,  $n=272$ ; Grassland B,  $R^2=1$ ,  $n=166$   
187 (Mellander et al., 2015); Grassland C,  $R^2=0.95$  and  $0.97$ ,  $n=316$ ; Arable A,  $R^2=1$ ,  $n=376$  (Mellander et al.,  
188 2015); Arable B,  $R^2=0.94$  and  $1$ ,  $n=493$ ). Both Grassland C and Arable B had changing controls at higher  
189 discharges and WISKI-SKED provided two parts to the curves with two  $R^2$  coefficients.

190

191 Turbidity units (NTU) were field-calibrated to SSC ( $\text{mg L}^{-1}$ ) using a combination of regular low-flow samples  
192 (at least fortnightly since programme initiation) and intensive sampling during high magnitude flow events with  
193 elevated SSCs. In all cases, water samples were collected from the instrument tank either manually, or using a  
194 programmable automatic water sampler (ISCO 6712; ISCO Inc. USA) with 1 m pumping tube (pump capacity  
195  $\sim 0.9 \text{ m}^3 \text{ s}^{-1}$ ) at predefined intervals of 30- or 60-mins according to the specific storm characteristics. High SSC  
196 data capture was further targeted in Grassland B and Arable B using a turbidity-stratified sampling programme,  
197 whereby collection of 1000 ml samples were triggered when T measurements were within threshold turbidity  
198 bands of 140 to 160 NTU, 240 to 260 NTU, 480 to 530 NTU and 700 to 800 NTU. This circumvented the need  
199 to pre-set water samplers according to forecasted event characteristics. Water samples were stored at  $4^\circ\text{C}$  on  
200 return to the laboratory before a sub-sample (minimum 100 ml) was processed for SSC. Whatman GF/C glass-



201 fibre filter papers (1.2  $\mu\text{m}$ ) were pre-dried at 105°C for 1 hr, cooled in a desiccator and weighed before being  
202 used for vacuum filtration. Sediment concentrations were calculated from the weight of residue retained on the  
203 filter post-filtration once dried >12 hr at 105°C and cooled in a desiccator.

204

### 205 3.2 Method comparison

206 In order to compare the ex situ sampling methodology described above with the conventional in situ monitoring  
207 approach, additional instrumentation to measure T was installed in Grassland B and Arable B from September to  
208 December 2012, and December 2012 to March 2013 respectively. A turbidimeter ( $T_{\text{IN}}$ ) (Analite, McVan,  
209 Australia, range 0-1000 NTU) fitted with a wiper blade to prevent biological fouling and automatic pumping  
210 sampler ( $\text{ISCO}_{\text{IN}}$ ) intake were positioned in situ, adjacent to the channel edge, in proximity to the bankside  
211 analyser pump intake (1 m and 4 m upstream, respectively in both catchments), but sufficiently distant not to  
212 affect, or to be affected by the ex situ instrumentation. The turbidity sensor  $T_{\text{IN}}$  and the  $\text{ISCO}_{\text{IN}}$  intake at  
213 Grassland B were approximately 20 cm above the channel bed and 15 cm from the bank edge. At Arable B,  $T_{\text{IN}}$   
214 and the  $\text{ISCO}_{\text{IN}}$  intake were positioned approximately 10 cm from the bank edge and 10 cm above the channel  
215 bed.  $T_{\text{IN}}$  and  $\text{ISCO}_{\text{IN}}$  sample collection was synchronised to replicate the ex situ turbidity sensor ( $T_{\text{OUT}}$ ) and  
216 pumping sampler ( $\text{ISCO}_{\text{OUT}}$ ) programme as described above. T-SSC rating curves were developed for each  
217 sensor using water samples collected at the respective positions ( $\text{ISCO}_{\text{OUT}}$  and  $\text{ISCO}_{\text{IN}}$ ) and applied to the raw  
218 turbidity set. Low quality data capture attributed to spurious readings (**a short-term increase in T output not**  
219 **associated with a known environmental process such as accompanying rise in Q or equipment**  
220 **maintenance**), saturation of the  $T_{\text{IN}}$  sensor or missing data at  $T_{\text{OUT}}$  due to delivery system blockages did not  
221 undergo correction such that comparisons between methodologies could be made. Five storm-flow events were  
222 captured in Grassland B and two in Arable B for T-SSC calibration. Due to the location settings, the in situ  
223 automatic water sampler was fitted with a 7 m long intake tube in both catchments.

224

225 Depth integrated water samples were manually collected (n=171) from a bridge over each investigated channel  
226 during flood events, using a depth-integrating SS sampler (US DH-48, Rickly Hydrological; USA). These  
227 samples were used firstly to investigate the cross-sectional variability in sediment transportation, and secondly  
228 to provide a validation dataset to assess and compare the efficacy of estimated SSC using at in situ and ex situ T  
229 sensors. Samples were collected using two strategies; 1) depth-integrated samples taken at 20 cm intervals  
230 across the channel width in rapid succession, and 2) samples taken at coarser widths roughly 1 m intervals. All

231 samples were processed for SSC as described above. Due to the sampling approach used, consecutive depth-  
 232 integrated samples reflected the event trend (either the rising or falling sedigraph limb) plus the cross-sectional  
 233 trend. The event effect was de-trended using SSC estimated from the ex situ turbidimeter. The average change  
 234 in SSC during transect sampling at  $T_{OUT}$  was 16.3 mg/l (range 0.1 – 81.94 mg/l), average transect time was 22  
 235 mins.

236

237 Where sufficient sample volume and sediment concentration existed, samples were analysed for particle size  
 238 distribution using laser diffraction (Malvern Mastersizer 2000G, Malvern, UK). Samples were circulated for 2  
 239 min (pump speed 2000 rpm, stirrer speed 800 rpm) before analysis with no pre-treatment, i.e., physical or  
 240 chemical dispersant, to broadly replicate the ‘effective particle size’ measured by the turbidity sensor. To assess  
 241 the effect of automatic sampler tube length, laboratory prepared SSC samples were collected using the two  
 242 intake pump lengths (1 m and 7 m) used in-field. Ten 500 ml sub-samples (at 5-, 10-, 25-, 50-, 100-, 250-, 500-,  
 243 750- and 1000-mg L<sup>-1</sup>) were collected from homogenised 10 litre mixtures using each pump length and  
 244 processed for SSC. A non-parametric Mann-Whitney U-test was conducted to compare SSC values collected at  
 245  $ISCO_{IN}$  (SSC  $ISCO_{IN}$ ) and  $ISCO_{OUT}$  (SSC  $ISCO_{OUT}$ ), and particle size characteristics at the two study sites.

246

### 247 2.3 Suspended sediment rating curve construction

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

252 Power  $SSC = aT^b$  Eq. 1

253 Split linear  $Where T < n \quad SSC = aT$  Eq. 2

$$Where T > n \quad SSC = c(b_1 - b_2) + b_2T$$

254 The intercept was set at zero for all regressions and was considered not to compromise fit at the upper end of the  
 255 dataset (cf. Thompson et al., 2014). Power relationships provided the best fit in Grassland A, Grassland B,  
 256 Grassland C and Arable A, whereas the split linear relationship considerably improved fit at Arable B (Table 2).  
 257 Using the selected curves, continuous turbidity measurements were computed to SSC and, using discharge data,  
 258 were converted to instantaneous sediment load ( $SSL - t \text{ s}^{-1}$ ) and yield ( $SSY - t \text{ km}^{-2} \text{ yr}^{-1}$ ).

## 259 4 Results and discussion

### 260 4.1 Method comparison

261 Dataset completeness was similar in both T records (98-99%); however, the timing and nature of spurious  
262 and/or missing T data were dissimilar (**Fig 3**). Spurious data at T<sub>IN</sub> coincided with random peaks possibly  
263 relating to local debris interference around the sensor which is a frequent problem in T analysis (Lewis and  
264 Eads, 2001). This trend was not recorded at T<sub>OUT</sub>, suggesting that the ex situ approach was less vulnerable to  
265 local in-stream debris interference (Jansson, 2002). Missing data at T<sub>IN</sub> during periods of high sediment  
266 concentration was attributed to sensor saturation at Arable B. The T<sub>OUT</sub> probe estimated 5% of the total sediment  
267 load was delivered whilst T<sub>IN</sub> was saturated. Sporadically, pump blockages occurred in T<sub>OUT</sub> at Arable B due to  
268 extreme debris transport in the channel (Melland et al., 2012b), data collection was ordinarily restored in less  
269 than 2 hr. At T<sub>IN</sub> 6% of the total load was delivered during this period. The ex situ turbidity monitoring may be  
270 at greater risk of delivery system blockages, especially during key periods of elevated turbidity and sediment  
271 transfer. These short periods are critical for sediment transport as they are responsible for the majority of the  
272 annual sediment load (Walling and Webb, 1988; Lawler et al., 2006; Estrany et al., 2009; Navratil et al., 2011).  
273 Other key issues such as bio-fouling trends were not found in either dataset, reflecting the sub-weekly frequency  
274 of maintenance at these sites.

275

276 Estimated sediment metrics (Table 3) during both monitoring periods showed discrepancies between the two  
277 measurement locations. Suspended sediment load estimated by ex situ equipment was 83% and 91% of in situ at  
278 Grassland B and Arable B, respectively, and mean SSC at SSC<sub>OUT</sub> was 85% of SSC<sub>IN</sub> at both locations.  
279 Differences in raw T output between the sensors were negated by calibration with SSC; however, the SSC of  
280 water samples from in situ (SSC ISCO<sub>IN</sub>) and ex situ (SSC ISCO<sub>OUT</sub>) measurement locations showed consistent  
281 differences. Samples at SSC ISCO<sub>OUT</sub> were 90% and 94% of SSC ISCO<sub>IN</sub> at Grassland B and Arable B  
282 catchments respectively. The differences in SSC and loads between the two approaches was not statistically  
283 significant, as confirmed by the non-parametric Mann-Whitney between SSC ISCO<sub>OUT</sub> and SSC ISCO<sub>IN</sub>  
284 ( $p > 0.05$ ).

285

286 Particle size analysis of event samples showed that the proportion of silt and sand particles changed through the  
287 events, whereas clay remained consistent. The greater density of sand particles compared to silts and clays can  
288 impact SSC and be oversampled by pumped samples such as the ISCO<sub>IN</sub> approach (Horowitz, 2008). The

289 percentage of sand (or sand-sized aggregates) between SSC ISCO<sub>IN</sub> and SSC ISCO<sub>OUT</sub> did not differ  
290 significantly ( $p>0.05$ ). Additionally, the ratio of the sand-sized fraction between simultaneous samples at  
291 ISCO<sub>IN</sub> and ISCO<sub>OUT</sub> showed no consistent evidence of over- or under-collection by either collection method.  
292 The hypothesis that inadequate sample collection could affect the differences between SSCs at ISCO<sub>IN</sub> and  
293 ISCO<sub>OUT</sub> is unlikely, as contrasts between the sand-sized fractions seemed to be event specific.

294

295 Differences between SSC ISCO<sub>IN</sub> and SSC ISCO<sub>OUT</sub> could not be directly attributed to diverging particle-size of  
296 the collected samples ( $p>0.05$ ), the pump length of the water sample collection ( $p>0.05$ - **Fig 4**), **or the position**  
297 **of the sample intake within the cross section (Fig 5)**. It is possible that the proximity of the ISCO<sub>IN</sub> pump  
298 intake to the channel bank could influence the relationship; however, differences could additionally result from  
299 methodological dissimilarities which could not be tested in isolation, i.e. the piped-delivery of river water to the  
300 ex situ instrument tank. The impact of elevated SSCs from ISCO<sub>IN</sub>, compared to ISCO<sub>OUT</sub> on the calibration of  
301 turbidity sensors T<sub>IN</sub> and T<sub>OUT</sub>, and the consequential prediction of high-resolution turbidity-based SSC record is  
302 discussed below.

303

### 304 3.2 Method validation

305 Samples collected from the channel cross-section were used to test the accuracy of predicted SSC using  
306 calibrated turbidity sensors at in situ and ex situ locations. The average SSC from each cross-sectional, depth-  
307 integrated set of measurements was plotted onto the rating curve over the method comparison monitoring period  
308 (**Fig. 6**). At Grassland B, measured SSCs plot within the 95% confidence intervals of predicted SSC using both  
309 methodologies using the simultaneous T values. This trend is repeated for the majority of samples at Arable B;  
310 however, some data points plot outside of the 95% confidence intervals for both in situ and ex situ method  
311 datasets. In the case that these out of range values were consistently higher or lower than the predicted values,  
312 this may suggest a systematic error due to sampling strategy; however, both upper and lower confidence limits  
313 were exceeded by the SSC values (**Fig. 7c and 7d**). Therefore, the error associated with the measurement  
314 method was generally less than that encapsulated within the 95% prediction intervals of the T to SSC calibration  
315 curve and consequently, both measurement approaches can be accepted as accurate for the estimation of SS  
316 metrics in these catchments. The suitability of ex situ water monitoring equipment installation must consider  
317 programme specific research objectives. Melland et al. (2012b) stated that for policy evaluation studies  
318 including multiple water quality parameters in addition to SSC, the improved resolution, accuracy and precision,

319 in particular for hydrologically dynamic catchments, justified the increased financial costs of initial installation  
320 of ex situ instrumentation.

321

### 322 3.3 Suspended sediment metrics in five agricultural catchments

323 High magnitude SSCs were of short duration in all five catchments (e.g. **Fig 3** for Grassland B and Arable B),  
324 but such periods are typically critical to cumulative annual SSY (**Fig. 7b** - Walling and Webb, 1988; Navratil et  
325 al., 2011). Grassland B and Arable B had a large proportion (80% of the monitoring period) of sediment  
326 transported at SSCs between 1 and 10 mg L<sup>-1</sup>, and shorter periods of concentrations  $\geq 10$  mg L<sup>-1</sup> for 15% and  
327 20% of the monitoring period respectively (**Fig. 7**). In the remaining catchments, low concentrations of <1 mg  
328 L<sup>-1</sup> were more common and occurred between 25 and 40% of the time. High concentrations ( $\geq 10$  mg L<sup>-1</sup>) were  
329 limited to less than 10% of the monitoring period. Overall, however, the FFD average annual SSC guideline was  
330 not exceeded in any monitoring year in any of the catchments (Table 4). The highest mean SSC of up to 17 mg  
331 L<sup>-1</sup> was recorded at Grassland B and Arable B and the remaining catchments reported very low values of <6 mg  
332 L<sup>-1</sup>. Accordingly, the instantaneous exceedance of the FFD guideline (Table 4) occurred during extremely short  
333 time periods (1-11% of sampled time per year). The values here are similar to those reported by Thompson et al.  
334 (2014) in two other intensively managed grassland catchments in Ireland; 8% exceedance was reported in a  
335 moderately-drained catchment in Co. Down and 18% exceedance in a poorly-drained catchment in Co. Louth.

336 Although the instantaneous exceedance of the FFD metric have been reported in other sediment studies  
337 (Glendell et al., 2014; Peukert et al., 2014; Thompson et al., 2014), the transferability of this coarse threshold  
338 (compliance to which requires an undefined annual sample number) to high-resolution SS data is questionable.

339

340 Average SSYs in the five catchments were 8.5, 24.7, 11.6, 12.0 and 24.4 t km<sup>-2</sup> yr<sup>-1</sup> at Grassland A, Grassland B,  
341 Grassland C, Arable A and Arable B respectively. **Figure 8** illustrates average annual SSYs from Ireland, the  
342 United Kingdom (UK) and the wider Atlantic climatic region of Europe (Vanmaercke et al. 2011). **The**  
343 **variability of average SSYs may be partly described by catchment size (x axis) but furthermore according**  
344 **to physical attributes such as soil type which controls soil erodibility. Values from catchments assessed in**  
345 **this study** align with existing data on SSY in Ireland (cf. Huang and O'Connell, 2000; Jordan et al., 2002;  
346 Harrington and Harrington, 2013; Thompson et al., 2014), and are consistently low compared with the UK and  
347 Europe. Considering the agricultural intensity of these catchment, (for example, Grassland A is within the

348 highest region of milk yield in Ireland (Läppe and Hennessy, 2012), and crop yields across Ireland are  
349 internationally high (Melland et al., 2012a)), these values are particularly low.

350

351 Catchment observations suggest high landscape complexity, comprising small and irregularly shaped fields,  
352 separated by a dense network of hedgerows and vegetated ditches (Table 1) reduced water and sediment  
353 connectivity potential between hillslopes and the channel network. Efficient drainage can be considered to  
354 reduce the spatial extent and temporal stability of connected areas and, considering the over-engineered nature  
355 of these ditch networks, encouraged sediment deposition (Shore et al., 2014). Furthermore, lower slope-lengths  
356 reduce the hillslope erosion potential (Lal, 1988), and sediment trapping and soil erosion prevention by root  
357 binding of hedgerows was observed.

358

359 In the UK, Cooper et al. (2008) suggested annual average 'target' and threshold 'investigation' SSY values be  
360 based upon drainage class and catchment terrain characteristics. Grassland A and Arable A qualify as lowland  
361 well-drained catchments and, on average, fall well below target and investigation SSY of 20 and 50 t km<sup>-2</sup> yr<sup>-1</sup>  
362 respectively. Grassland B, Grassland C and Arable B, categorised as lowland predominantly poorly-drained  
363 catchments, on average, fall below target and investigation thresholds of 40 and 70 t km<sup>-2</sup> yr<sup>-1</sup>, respectively.  
364 Total SSY data for individual years (Table 4), however, indicate variability and exceeded respective SSY target  
365 values at Grassland B in 2009 and 2012, Arable A in 2012 and Arable B in 2011 and 2012.

366

367 Higher average SSC, intra-annual period of FFD exceedance, and average SSY in catchments Grassland B and  
368 Arable B are suggested to result from poorer soil drainage. During rainfall events, soils are rapidly saturated and  
369 critical overland flow pathways established, and consequently, eroded particles within these connected areas are  
370 transported through the catchment (Mellander et al., 2012; Shore et al., 2013). The SSC responses here suggest,  
371 as in other catchments with impeded drainage, that high overland-flow potential is also associated with a notable  
372 proportion of sediment delivered at lower concentrations over a longer period, through surface and sub-surface  
373 flow pathways such as through macropores and tile drains (e.g., Deasy et al., 2009; Melland et al., 2012a;  
374 Ibrahim et al., 2013 Mellander et al., 2015) resulting in increased average SSCs. In catchments Grassland A  
375 and Arable A, sub-surface flow pathways dominate, due to well-drained soils reducing the likelihood of  
376 overland flow and consequently surface soil losses. Furthermore, at Arable A, Mellander et al., (2015) found  
377 weather bedrock formed groundwater pathways further decreasing surface pathway initiation. Consequently,

378 SSCs, intra-annual period of FFD exceedance, and SSYs were low. Conversely, Grassland C more accurately  
379 reflects the sediment characteristics of the well-drained catchments despite the moderate- to poorly-drained  
380 soils. Near complete cover of permanent pasture here was considered to sufficiently reduce sediment source  
381 availability and transport of sediment to the watercourse.

382

383 Generalisations can be made in relation to the overriding controls on SSY across the monitored catchments (**Fig.**  
384 **9**). Inter-catchment comparisons here used data from hydrological years 2010 to 2013, where data were  
385 available for all five catchments. Sediment delivery was enhanced by the combined effect of an overland-flow  
386 dominated transport system (poorly-drained soils) and, to a lesser extent, source availability (arable soils with  
387 potentially lengthy periods of bare ground cover (Regan et al., 2012) or seasonally thinly vegetated grassland  
388 soils (cf. Bilotta et al., 2010)). Catchments that possess better drainage characteristics and/or permanent crop  
389 cover have greater resilience to extreme sediment losses. In catchments such as Arable A, where good-drainage  
390 is combined with high source availability, the risk associated with sediment transport during extreme rainfall  
391 events and years was, nevertheless, high. Similarly, poorly-drained soils stabilised by permanent pasture should  
392 be maintained and periods of bare cover should be avoided.

393

394 High inter-annual variability was evident, particularly with regard to SSY (Table 4). The annual SSY coefficient  
395 of variation (CV%) were 67%, 76%, 79%, 83% and 50% in Grassland A, Grassland B, Grassland C, Arable A  
396 and Arable B, respectively. Notably, in Grassland B and Arable B catchments, the inter-annual SSY ranges of  
397 41.7, and 26.2 t km<sup>-2</sup> yr<sup>-1</sup>, respectively, were greater than average annual inter-catchment SSY of 24.0 t km<sup>-2</sup> yr<sup>-1</sup>.  
398 The variability found within each of the five monitoring catchment was comparable to the results of  
399 Vanmaercke et al., (2012) who reported CV% ranging from 6-313% (median 75%) in 726 catchments  
400 worldwide. The catchment with the lowest inter-annual SSY (11.0 t km<sup>-2</sup> yr<sup>-1</sup>), Grassland A, received the least  
401 variable rainfall input and total discharge.

402

403 Inter-annual SSY variability results from strong seasonality due to the timing and character of rainfall events,  
404 soil moisture deficit and land management which conditions sediment availability in critical source areas.  
405 Analysis of shorter term sediment losses i.e., at seasonal, monthly and event scales would also provide empirical  
406 evidence to inform both high level policy considerations and local decision making. Additionally, assessment of  
407 seasonal transfers are likely to have greater ecological significance as mean annual thresholds such as SSC

408 (through the FFD), and SSY may underestimate the seasonal fluctuations of risk of sediments to aquatic  
409 ecosystems (Thompson et al., 2014). Sensitivity to sediment is species-specific and dependent upon life stage  
410 (Collins et al., 2011); therefore, shorter-term metrics such as the timing, magnitude, duration and frequency of  
411 sediment transfers are important concepts to consider. Existing static thresholds may, therefore, be considered  
412 ecologically irrelevant, particularly when utilised as an instantaneous threshold for high-resolution data. Future  
413 discussion regarding sediment targets requires an assessment of multiple species and habitat quality. This task is  
414 particularly complicated where ecological condition is subject to multiple-stressors such as nutrients (Bilotta and  
415 Brazier, 2008), bed substrate quality (Kemp et al., 2011) and time lag of water quality response to pollutant  
416 mitigation measures (Fenton et al., 2011; Vero et al., 2014).

417

418 Overall, annual average sediment metrics from small catchments ( $\sim 10 \text{ km}^2$ ) with dominant land uses  
419 representative of main land use types in Ireland reported here are internationally low. Considering the spatial  
420 dominance and intensity of agricultural land use and high effective rainfall in the study catchments, this is  
421 perhaps unexpected particularly considering the small scale of study. As previously discussed the complexity of  
422 landscape features (e.g., fields, hedgerows, ditches) which are representative of the wider Irish agricultural  
423 landscape (Deverell et al., 2009) can be expected to decrease the likelihood of field-scale soil erosion, and/or  
424 increase the opportunity for interception and deposition of mobile particles on land or within the hydrological  
425 network. The Irish landscape may, therefore, improve the resilience of agricultural soils to soil loss. However,  
426 even with modest SSY, the potential for other specific risks to ecologically sensitive habitats, from SS  
427 deposition in rivers for example, will need a cautionary approach. Therefore, identification of the specific  
428 mechanisms promoting soil conservation or sediment retention in multiple catchments with contrasting physical  
429 and land use characteristics is important. This is particularly relevant for water and agricultural policy, as the  
430 prevention of environmental degradation and maintenance and/or sustainable intensification of agricultural  
431 production are simultaneously considered. Furthermore, other sediment sources, for example, from channel  
432 banks and road networks may contribute significant proportions of the annual load (Rowan et al., 2012; Collins  
433 et al., 2013; Sherriff et al., 2014). Assessment of such sources could be a useful insight to prioritise sediment  
434 management strategies (Wilson et al., 2008).



435 5 Conclusions

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

- 440 • Suspended sediment metrics between in situ and ex situ methodologies were not significantly different  
441 from in-stream cross-sectional, depth-integrated samples in two monitoring catchments.
- 442 • The ex situ methodology reported less sensitivity to spurious data peaks; however, periods of extreme  
443 large debris transport increased the sensitivity of the ex situ instrumentation to short-term blockages.
- 444 • All catchments reported mean annual SSCs of less than the FFD threshold of 25 mg L<sup>-1</sup> and short-term  
445 exceedance of 1-11% of sampled time.
- 446 • Inter-annual variability of SSY was strong due to the timing and character of rainfall events in relation  
447 to land management.
- 448 • Average annual SSYs in all five Irish catchments reported here were low in comparison to similar  
449 catchment and landscape settings elsewhere in Europe. Farming practices favouring relatively small  
450 fields, a high density of field boundaries including ditches, with low consequent connectivity are likely  
451 to explain this.
- 452 • Within the study catchments, SSY was higher in catchments dominated by poorly-drained soils than  
453 those with well-drained soils. Furthermore, on poorly-drained soils, catchments with a greater  
454 proportion of arable land use reported the highest annual average SSY.
- 455 • Well drained soils dominated by arable crops did, however, show the potential to supply significant  
456 quantities of sediment.
- 457 • Complexity of landscape features (hedgerows, drainage ditches and irregular field sizes) may provide  
458 resilience to soil erosion and/or sediment transport despite spatial dominance and intensity of  
459 agriculture and these will be important considerations for future management (such as sustainable  
460 intensification) and/or SS mitigation in Ireland and elsewhere.

461

462 These findings illustrate that interactions between climate, landscape and land use regulate the supply of  
463 sediments from Irish agricultural catchments. Whilst the current SSYs are low by international standards, key  
464 questions still remain regarding the impact of land use on the magnitude and frequency characteristics of

465 sediment transfers at shorter timescales. Seasonal and storm-event scale sediment transfers may better inform  
466 erosion risk due to better detection of sediment pulses moving into the channel network particularly within  
467 ecologically sensitive periods. Further to this, seasonal sediment provenance and field-scale soil loss  
468 assessments within this land management and landscape framework are crucial to quantify the contributions  
469 made from specific agricultural and other sediment sources.

470

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478

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690 Tables

691 Table 1. Summary of study catchments.

Catchment	Size (km <sup>2</sup> )	30-year average rainfall <sup>a</sup> (mm yr <sup>-1</sup> )	Median slope (°)	Dominant soil drainage class/ flow pathway	Land-use	Landscape complexity features			
						Field size (ha)	Maximum down-slope length (m)	Hedgerow density (km/km <sup>2</sup> )	Ditch density (km/km <sup>2</sup> )
Grassland A	7.9	1228	4	Well-drained <i>Sub-surface</i>	89% grassland predominantly for dairy cattle; 5% arable	2.00	170	0.061	1.7
Grassland B	11.5	906	3	Poorly-drained <i>Surface</i>	77% grassland for dairy cattle, beef cattle and sheep; 12% spring crops 2% winter crops	3.04	189	0.011	5.7 <sup>b</sup>
Grassland C	3.3	960	6	Moderately- to poorly- drained <i>Surface</i>	94% grassland for beef cattle, dairy cattle and sheep	1.12	114	0.044	2.6
Arable A	11.2	906	3	Well-drained <i>Sub-surface</i>	54% arable predominantly spring crops; 39% grassland mainly for beef cattle and sheep	3.32	194	0.011	1.3 <sup>b</sup>
Arable B	9.4	758	3	Poorly- drained <i>Surface</i>	24% winter crops; 29% grazing for beef cattle and sheep; 19% dairy cattle grazing	2.70	200	0.011	2.3

692 <sup>a</sup>1981-2010 mean annual rainfall

693 <sup>b</sup>from Shore et al., (2013)

694

695 Table 2: Turbidity- Suspended sediment calibration dataset summary and rating curve equations and fit  
 696 parameters

Catchment	Data points	Calibrated turbidity range (NTU)	Maximum measured turbidity in NTU (number of data points outside calibrated range) <sup>a</sup>	Calibration equation	MSE
Grassland A	247	0.044-725	1074 (n=7)	$SSC=0.6636T^{1.1045}$	495
Grassland B	443	0.506-577	1179 (n=37)	$SSC=0.5657T^{1.1109}$	580
Grassland C	339	1.170-154	1225 (n=207)	$SSC=0.4341T^{1.2148}$	38
Arable A	231	1.177-767	2730 (n=30)	$SSC=0.4119T^{1.1456}$	891
Arable B	242	0.75-1853	1853 (n=0)	Where $T < 432.2$ $SSC=1.1320T$ Where $T > 432.2$ $SSC=0.5288+0.6032T$	1335

697 <sup>a</sup>Number of data points at 10 min resolution  
 698

699 Table 3. Suspended sediment metrics estimated using in situ and ex situ turbidity based SSC estimation  
 700 methods.

Catchment	Total load (t) <sup>a</sup>		Mean concentration (mg L <sup>-1</sup> )		Max concentration (mg L <sup>-1</sup> )	
	SSL <sub>OUT</sub>	SSL <sub>IN</sub>	SSC <sub>OUT</sub>	SSC <sub>IN</sub>	SSC <sub>OUT</sub>	SSC <sub>IN</sub>
Grassland B	128±28	154±35	13.7	16.2	1010	1188
Arable B	225±54	248±52	29.1	34.1	2043	823 <sup>b</sup>

701 Note: <sup>a</sup> confidence intervals are the coefficient of variance of the mean prediction, <sup>b</sup> T<sub>IN</sub> sensor saturated at 1000  
 702 NTU

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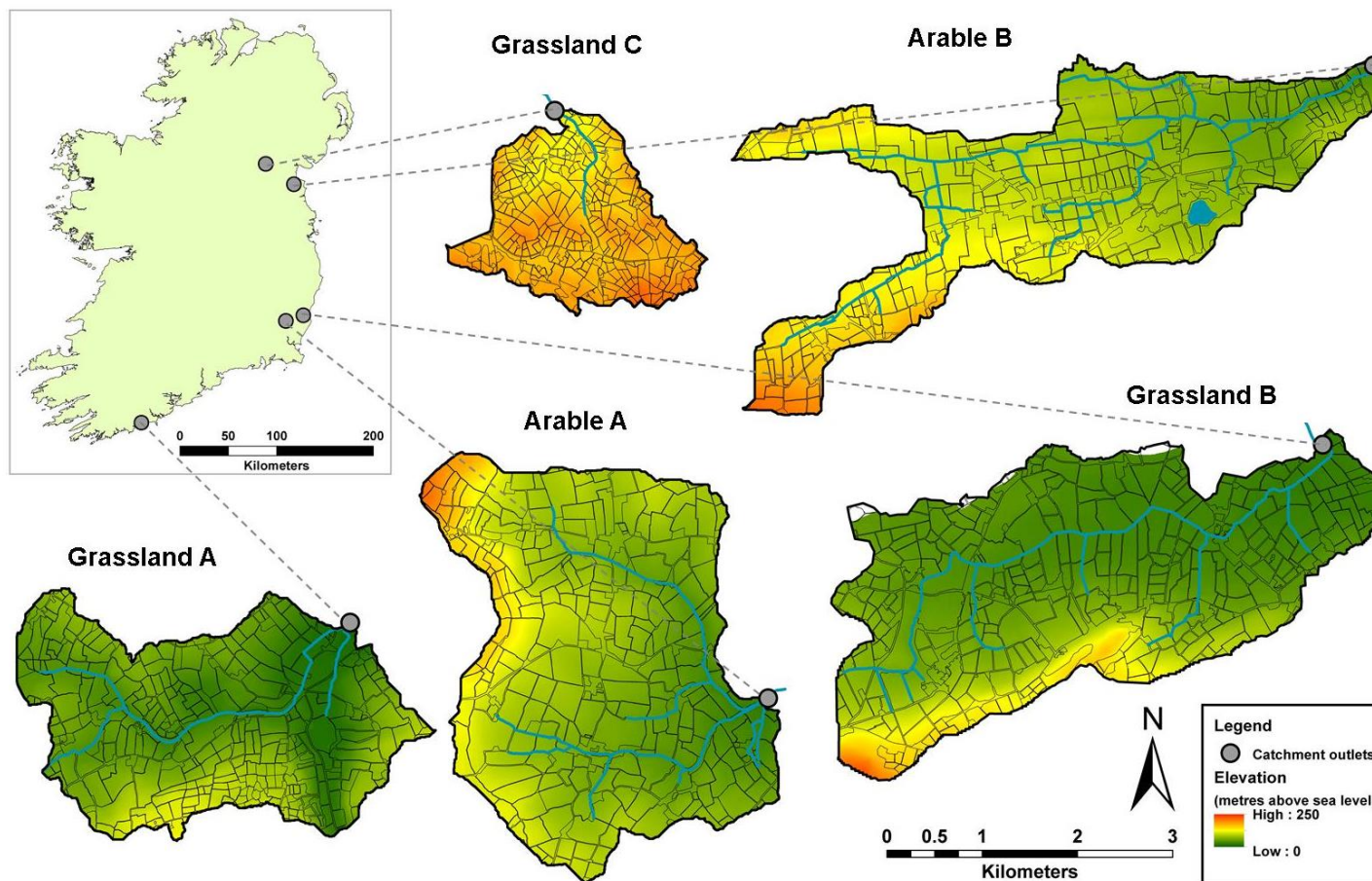
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705 Table 4. Annual rainfall, discharge and suspended sediment flux summary for five catchments. Monitoring years correspond to hydrologic years (October to September).

Year	Grassland A				Grassland B				Grassland C				Arable A				Arable B		
	2010	2011	2012	2009	2010	2011	2012	2010	2011	2012	2009	2010	2011	2012	2009	2010	2011	2012	
Rainfall (mm yr <sup>-1</sup> )	1045	1139	1097	1278	800	1155	920	965	1234	969	1240	763	1102	827	896	742	1049	844	
Runoff (mm yr <sup>-1</sup> )	443	633	608	643	330	504	382	424	727	575	750	366	517	473	383	319	521	542	
Mean SSC (mg L <sup>-1</sup> )	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 <sup>-1</sup> )	707	467	966	1020	426	882	707	419	813	462	773	224	737	2141	494	707	688	1120	
>25 mg L <sup>-1</sup> (% 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 (tonnes km <sup>-2</sup> yr <sup>-1</sup> )	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	

706 \*ST=sampled time

707



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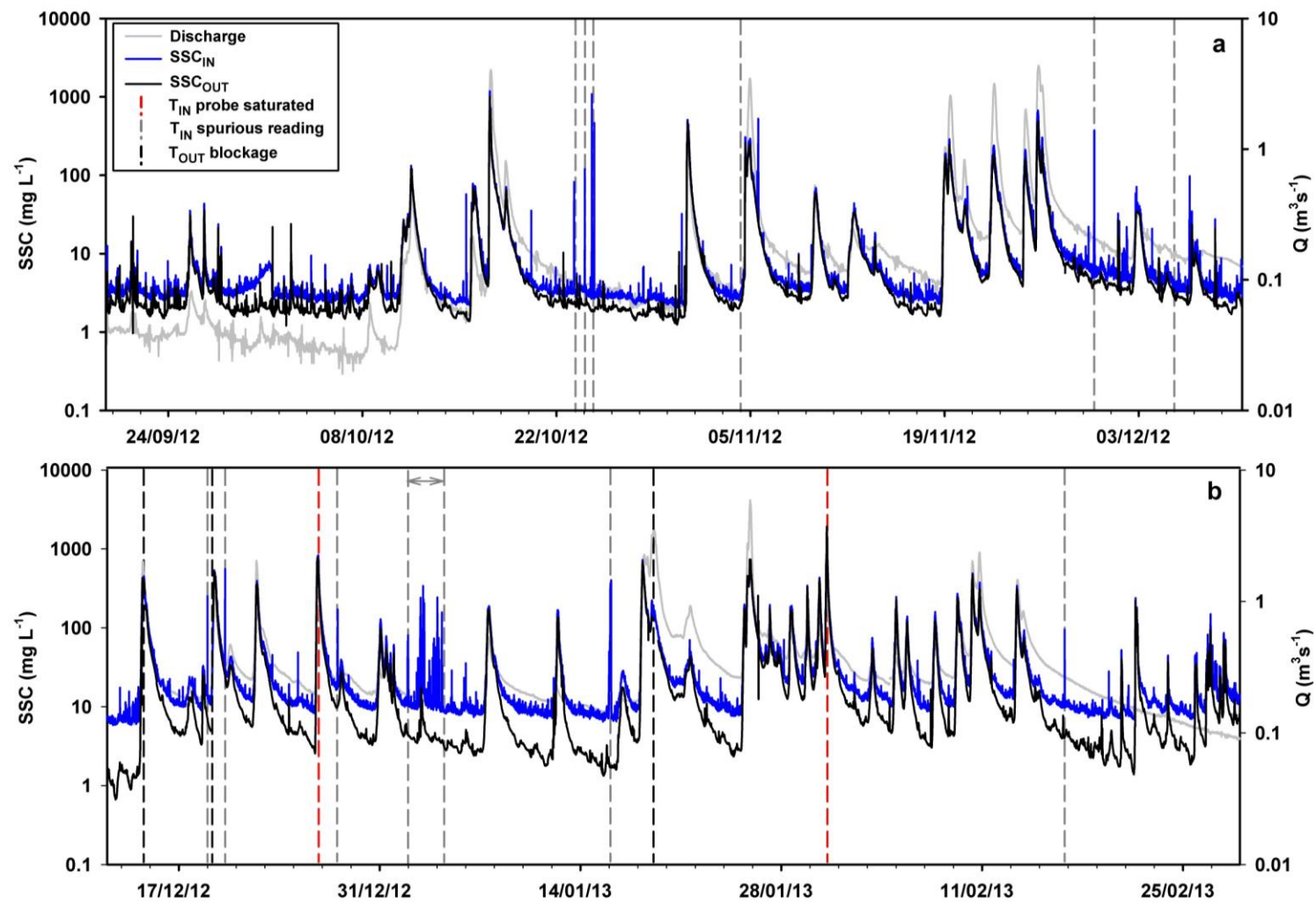
710 **Figure 1.** Map of catchment monitoring locations and study catchments with topographic and field size information.



711

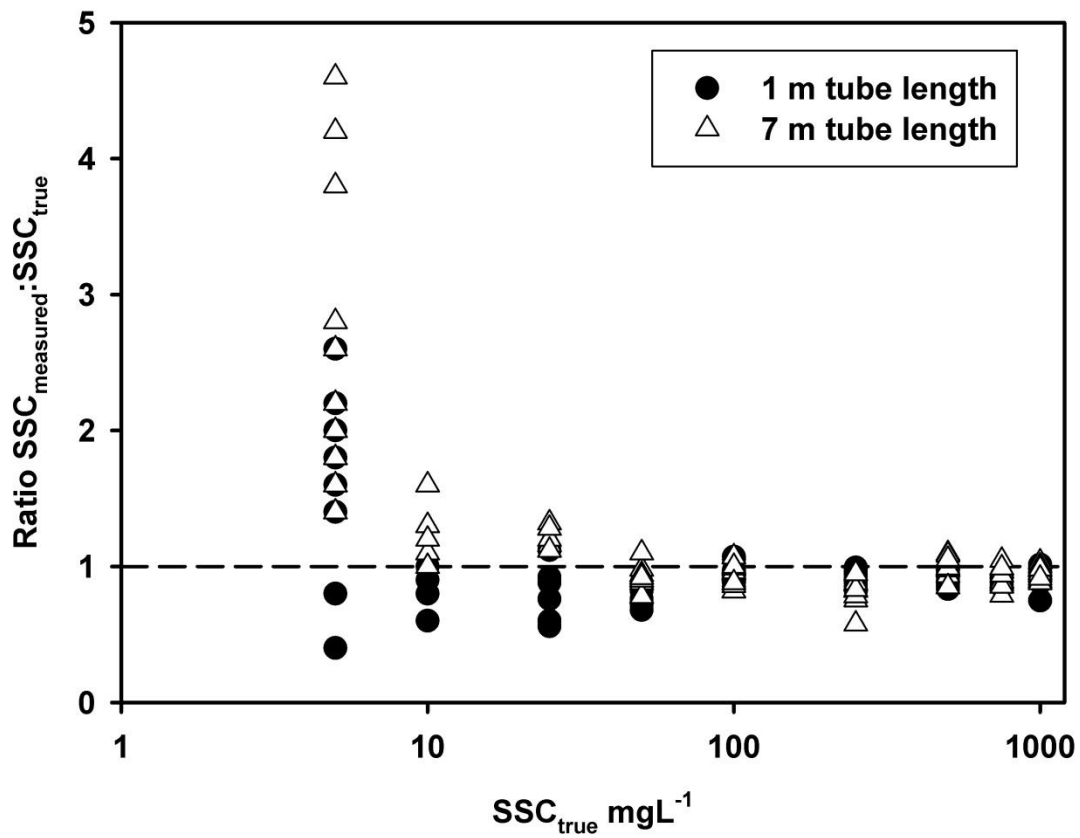
712 **Figure 2:** Picture of in situ and ex situ suspended sediment and discharge instrumentation at Grassland B.





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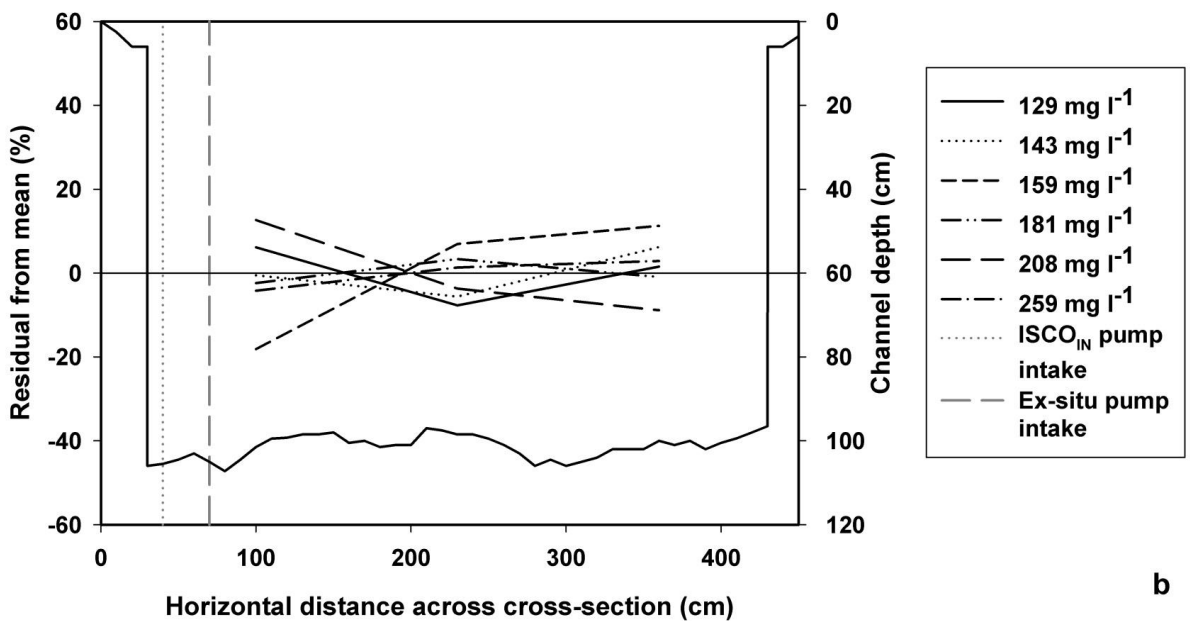
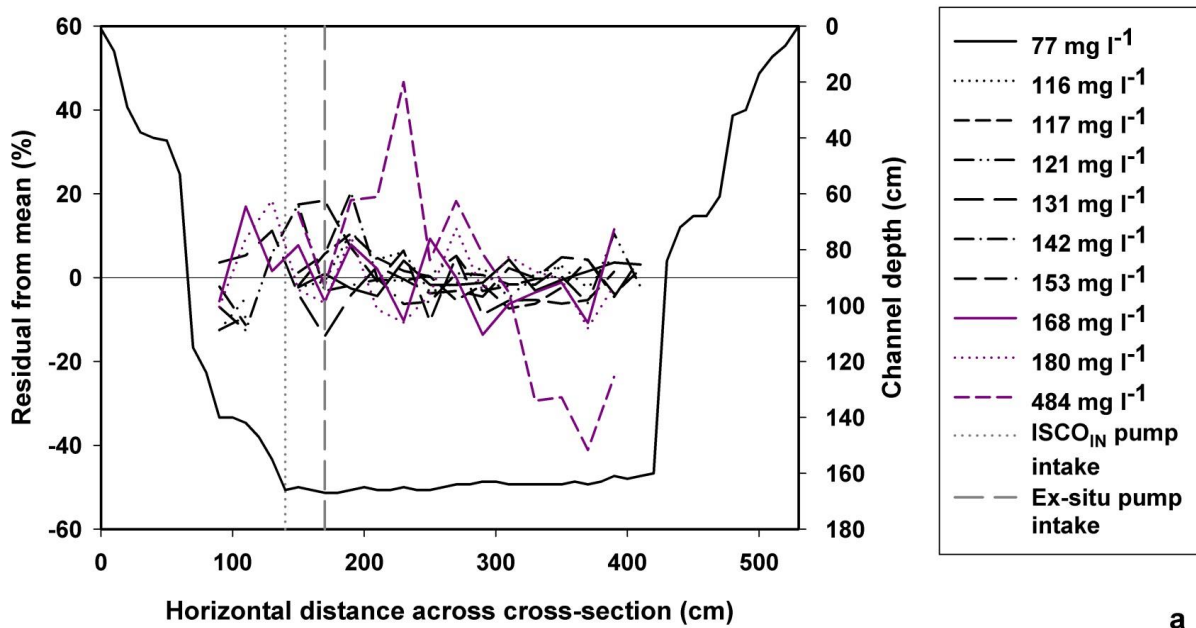
714 **Figure 3:** Raw turbidity output of T<sub>IN</sub> and T<sub>OUT</sub> sensors (converted to SSC) and discharge at a) Grassland B and, b) Arable B. Periods of missing data are annotated by  
 715 **dashed lines.**



716

717 **Figure 4:** Suspended sediment concentration of samples collected from known concentration mixtures ( $SSC_{true}$ )

718 using ISCO water samplers with 1m and 7m tube lengths.



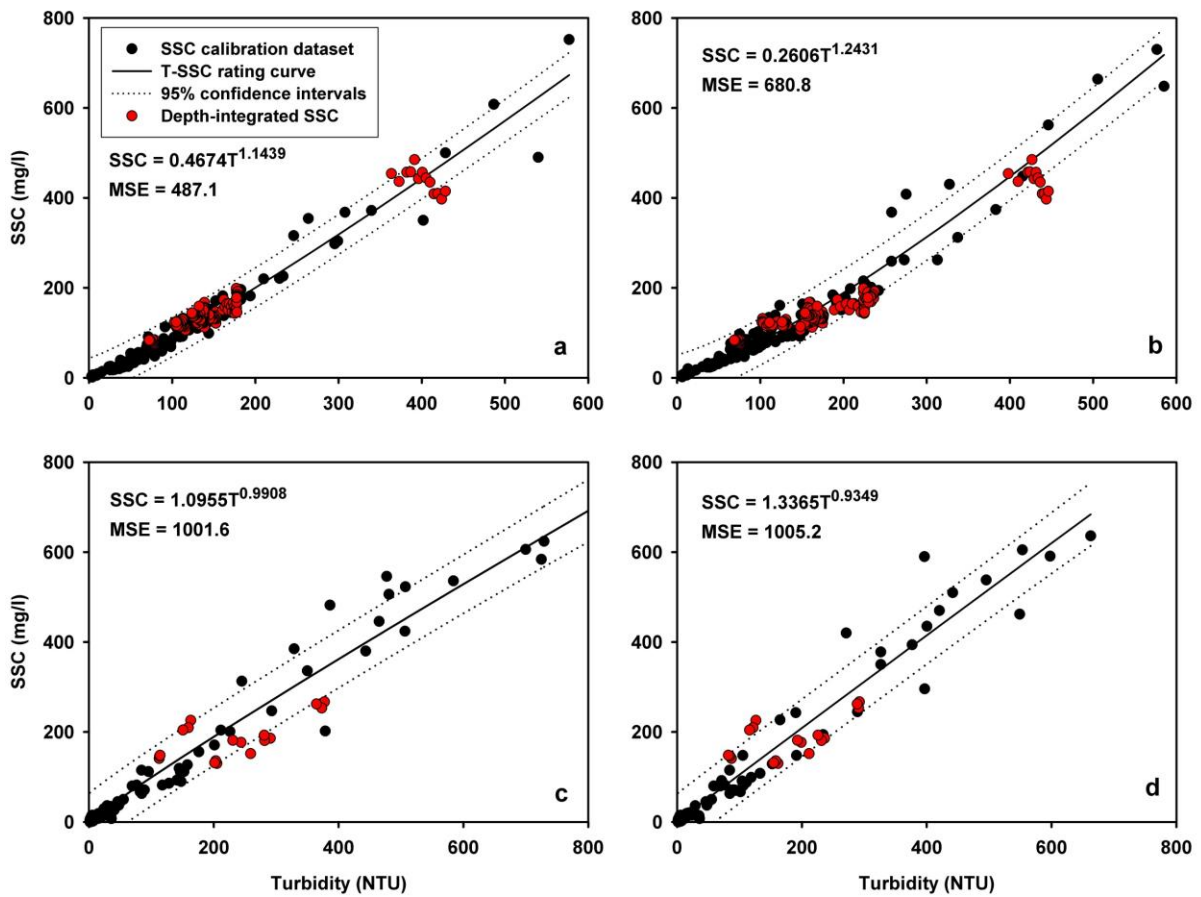
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720 **Figure 5:** Variability of instantaneous depth-integrated SSC measurements across the channel cross section

721 compared to the mean transect SSC a US DH-48 sediment sampler at a) Grassland B and, b) Arable B.

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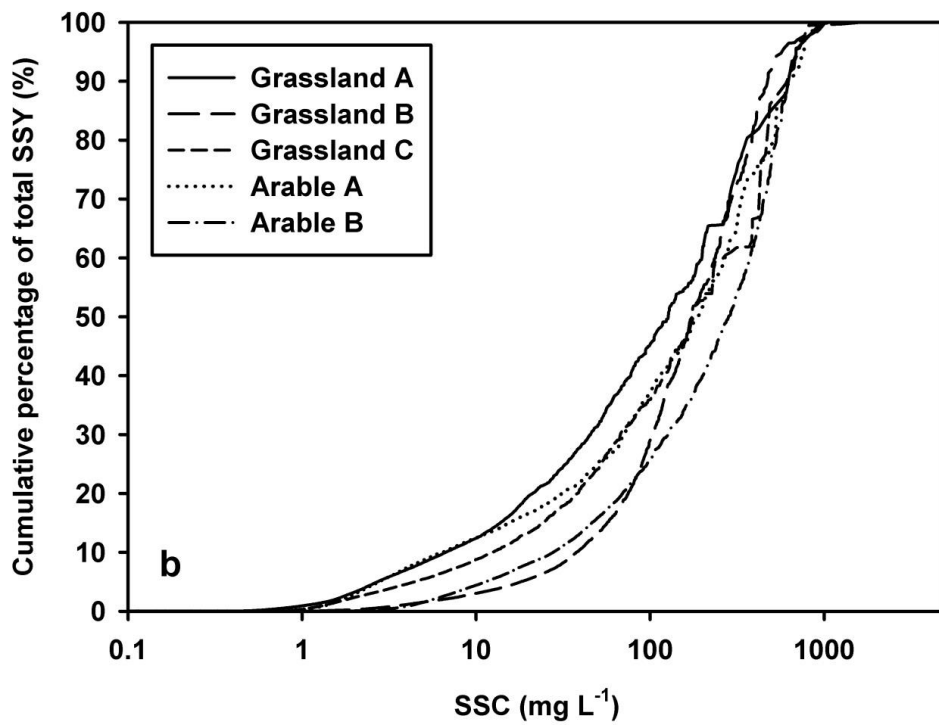
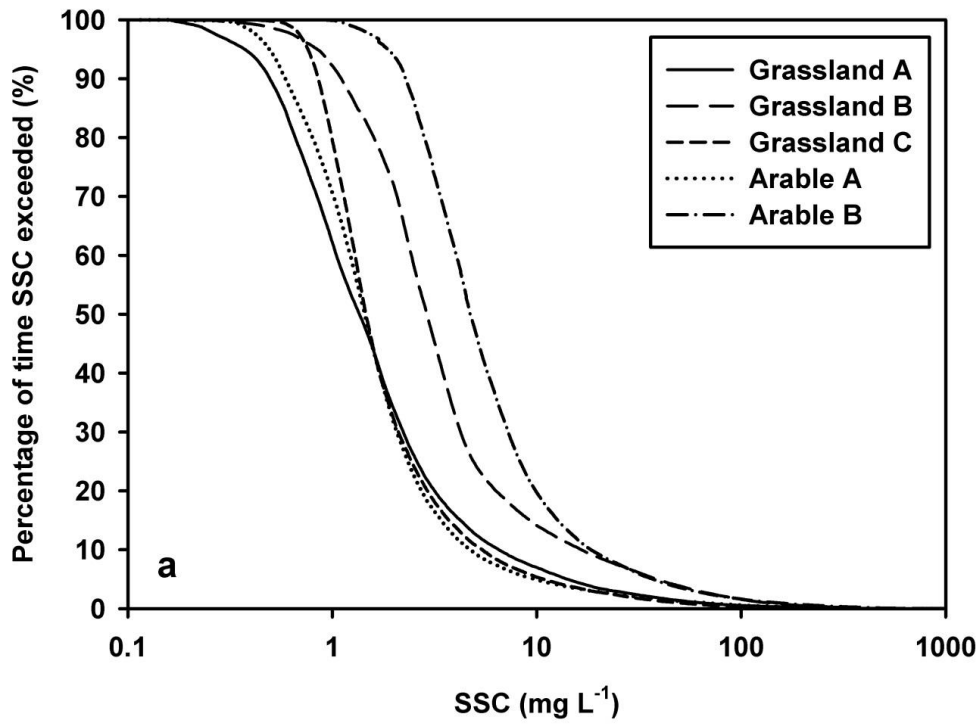
725 **Figure 6.** Turbidity-suspended sediment concentration rating curves, confidence intervals, calibration data and

726 cross-section depth-integrated suspended sediment concentration samples for, a) Grassland B  $T_{OUT}$ , b)

727 Grassland B  $T_{IN}$ , c) Arable B  $T_{OUT}$ , d) Arable B  $T_{IN}$ .

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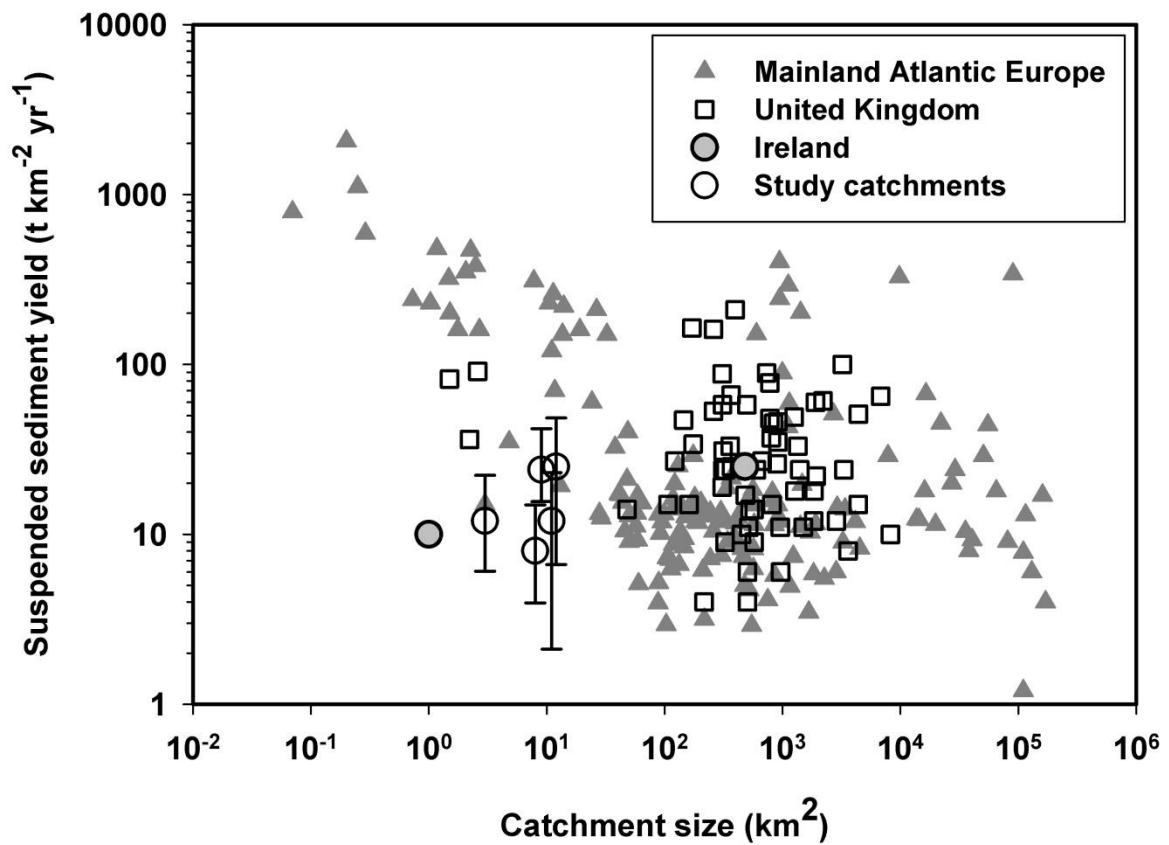


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731 **Figure 7.** Frequency-duration graphs of, a) suspended sediment concentration exceedance with time and, b)

732 Cumulative percentage of suspended sediment yield with exceedance of suspended sediment concentration.

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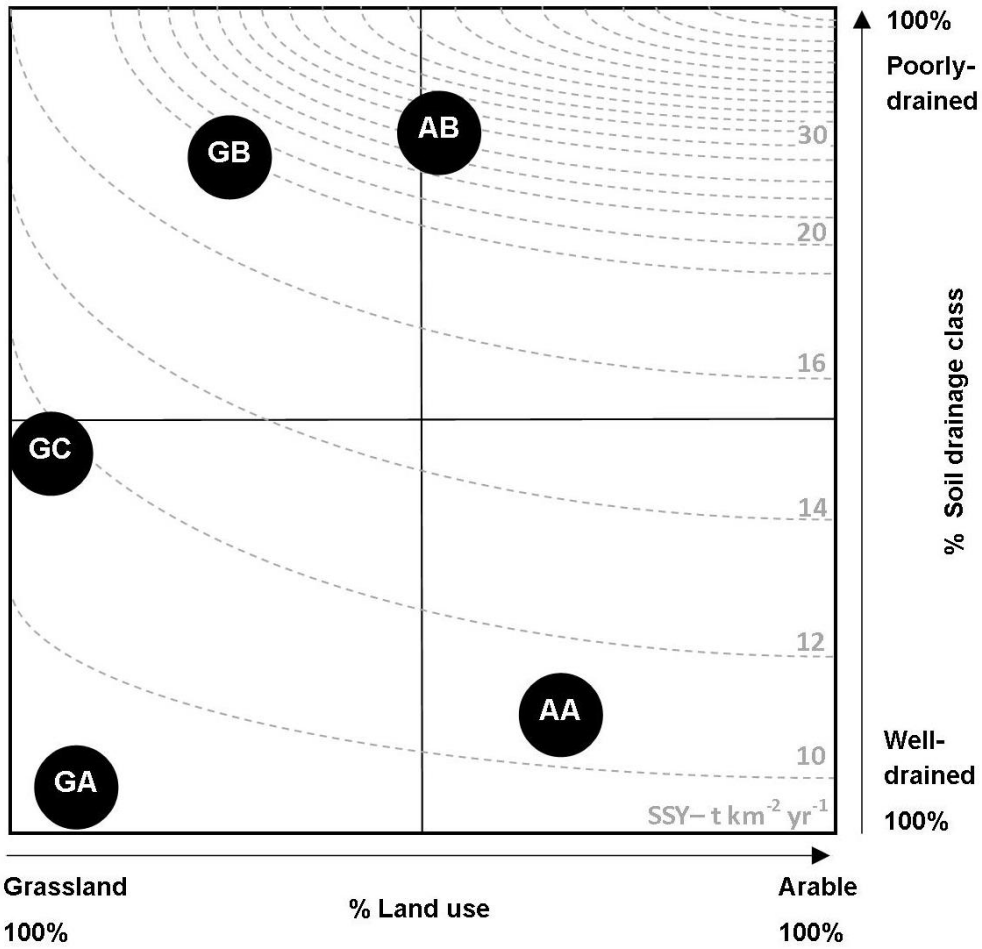


734

735 **Figure 8.** Catchment size and suspended sediment yield of European river catchments, study catchments  
 736 displayed with inter-annual range. Sources: Foster et al. (1986); Milliman and Syvitski (1992); McManus and  
 737 Duck (1996); Wass and Leeks (1999); Huang and O’Connell (2000); Verstraeten and Poesen (2001); Jordan et  
 738 al. (2002); Walling et al. (2002); Harlow et al. (2006); Oeurng et al. (2010); Zabaleta et al. (2007); Gay et al.  
 739 (2014).

740

741



742

743 **Figure 9.** Conceptual diagram of suspended sediment yield as represented by iso-lines according to land use and

744 dominant soil drainage class. Catchment abbreviations: GA- Grassland A, GB- Grassland B, GC- Grassland C,

745 AA- Arable A, AB- Arable B.