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Intensive management in grasslands causes diffuse water pollution at the farm-scale

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Abbreviations: BMP, best management practice; DOC, dissolved organic carbon; DIC; dissolved inorganic carbon; EU WFD, EU Water Framework Direction; EU FFD, EU Freshwater Fisheries Directive; FYM, Farmyard manure; NVZ, Nitrate Vulnerable Zone; NH_3^-, nitrate; NH_2^-, nitrite; NH_4^+, ammonium; SRP soluble reactive P; SS, suspended sediment; TC, total carbon; TOC; total organic carbon; TIC, total inorganic carbon; TPC, total particulate carbon; TON_N, total oxidized nitrogen-N; TP, total phosphorus; PP, particulate phosphorus

Abstract. Arable land-use is generally assumed to be the largest contributor to agricultural diffuse pollution. This study adds to the growing evidence that conventional temperate intensively managed lowland grasslands contribute significantly to soil erosion and diffuse pollution rates.

This is the first grassland study to monitor hydrological characteristics and multiple pollutant fluxes (suspended sediment [SS] and the macronutrients: total oxidized nitrogen-N [TON_N], total phosphorus [TP] and total carbon [TC]) at high temporal resolution (monitoring up to every 15 minutes) over one year. Monitoring was conducted across three fields (6.5 - 7.5 ha) on the North Wyke Farm Platform, UK. The estimated annual erosion rates (up to 527.4 kg ha^{-1}), TP losses (up to 0.9 kg ha^{-1}) and TC losses (up to 179 kg ha^{-1}) were similar to or exceeded the losses reported for other grassland, mixed land-use and arable sites. TON_N annual yields (up to 3 kg ha^{-1}) were less than arable land use fluxes and earlier grassland N studies; an important result as the study site is situated within a Nitrate Vulnerable Zone. The high resolution monitoring allowed detailed "system's functioning" understanding of hydrological processes and mobilization- transport pathways of pollutants. SS and TP concentrations frequently exceeded water quality guidelines recommended by the European Freshwater Fisheries Directive (25 mg L^{-1}) and the European Water Framework Directive (0.04 mg soluble reactive P L^{-1}), suggesting that intensively managed grasslands pose a significant threat to receiving surface waters.

Such sediment and nutrient losses from intensively managed grasslands should be acknowledged in land management guidelines and advice for future compliance with surface water quality standards.

Introduction

Approximately 40% of water courses in the USA (Evans-White et al., 2013), more than half of all European water courses (EEA, 2012) and two thirds of all surface waters in the UK do not reach drinking water status or good ecological status (McGonigle et al., 2012). In the UK alone, diffuse agricultural pollution contributes to water quality failures as the source for 75 % of sediments (Collins and Anthony, 2008) and high proportions of macronutrients: 55 % of nitrates (Hughes et al., 2008) and 20 % of phosphorus (White and Hammond, 2009). Carbon is not yet included in water quality guidelines in terms of freshwater ecosystem health, but rising carbon concentrations in surface waters are known to impact on water quality (Edwards et al., 2008). These pollutants cause human health problems (polluted drinking and bathing water) and high economic costs (Parris, 2011). Ecological deterioration of water courses, due to eutrophication, is also a problem: increased occurrence of algal blooms with subsequent dissolved oxygen depletion, reduced light availability and perturbation of the balance of organisms, generally associated with a decline in invertebrate and fish species (Bilotta and Brazier, 2008, Bilotta et al., 2008).

The effects of different agricultural land uses on pollutant sources, mobilization, transfer and delivery to water bodies need to be understood in detail (Haygarth et al., 2005a), to support the need for sustainable intensification of agriculture, whilst minimising diffuse pollution, and eventually to identify best management practices (BMPs). However, gaining such knowledge has been constrained for several reasons, discussed below.

1. Grasslands managed for dairy and meat production, despite covering more than half (65 %) of the agricultural land in the UK (Bilotta et al., 2008) and extensive areas of western Europe (Brazier et al., 2007), Australia and New Zealand (Nash et al.,

2000) and the United States (40%) (Peeters, 2004), have received less research attention than other land uses in terms of water quality research. Only recently has the general assumption been challenged that grasslands contribute little to total agricultural diffuse pollution in comparison to their arable counterparts (Brazier et al., 2007, Evans, 2005). Several studies have since demonstrated sediment and macronutrient losses from grasslands to be comparable with those of arable fields/catchments and to exceed EU water quality guidelines (Bilotta et al., 2008, Bilotta et al., 2010, Granger et al., 2010, Preedy et al., 2001).

2. Multiple diffuse agricultural pollutants are rarely considered in the same study. However, it is widely acknowledged that it is not the availability of single nutrients, but the relative availability of multiple nutrients that impact aquatic ecosystems. The impacts range from direct toxic effects on aquatic biota to effects on primary productivity, cascading through the entire aquatic food web (Dungait et al., 2012, Elser and Urabe, 1999). The acknowledgment of the relative importance of nutrients has challenged the general consensus that freshwater systems are always P limited (Conley et al., 2009, Elser et al., 2007). Additionally, most studies focus on inorganic forms of N and P as they were considered as the bioavailable forms and EU/ UK water quality regulations focus on those forms. Such studies fail to capture the total nutrient delivery of both directly bioavailable (inorganic forms) and organic forms that can become available by assimilation by bacteria (organic N, (See et al., 2006)) or hydrolysis (organic P, (Darch et al., 2013)). Water quality guidelines in the USA for example are set as total N and total P (USEPA, 2000).

3. The depth of scientific understanding has often been limited by the temporal resolution of the data. The recent development and increased use of near-continuous water quality sampling equipment promises great advances in hydrochemical process

understanding (Kirchner et al., 2004). Generally, the higher the data resolution and the longer the time-series, the more revealing the results are. Understanding fine temporal variability of sediment ant nutrient fluxes, estimating accurate pollutant loads (Jordan and Cassidy, 2011), and calculating accurate and precise calculation of water quality guideline exceedance-frequencies are examples of what is possible (Bilotta et al., 2010, Thompson et al., 2014).

4. Catchment scale water quality research, whilst conducted at a pragmatic 'hydrological' scale is often limited in providing information on pollutant sources within catchments, process information on the contribution of individual units of land use or information on the effectiveness of policy efforts towards reducing water pollution (Doody et al., 2012, Haygarth et al., 2005b). Each catchment has numerous pollutant sources and land uses, which are managed separately by different stakeholders (e.g. farmers, landowners or rural industry). Any effort towards reducing pollution is generally targeted towards individual pollutant sources and their managers. Therefore, the scale of management decisions and policy implementation is not the catchment scale, but in the case of agricultural diffuse sources, it is the individual farm scale. Here, we suggest that field to farm scale research is a logical scale to guide agricultural diffuse pollution research. Field to farm-scale research within a catchment might give insight into a) the likely contribution a single type of agricultural land use has on overall water pollution and therefore improve pollution source apportionment within catchments, b) the effects of individual soil and nutrient management practices, and c) the effectiveness of mitigation measures. The usefulness of farm scale experiments has been reviewed before (Garcia et al., 2008, Pilgrim et al., 2010), but there are few, comprehensive field to farm-scale monitoring experiments underway, especially in intensively managed grasslands.

Addressing the above limitations in water quality research, this study quantifies the hydrological characteristics and the fluvial fluxes of suspended sediment and the macronutrients total carbon, nitrogen (total oxidised nitrogen) and total phosphorus at high resolution (up to 15 minute sampling intervals) over one year in intensively managed grasslands. Thus, this study provides novel information to answer the following key questions:

Q1. How do rates of sediment and macronutrient delivery from intensively managed grasslands compare to other agricultural land uses?

Q2. What are the controlling factors on hydrology and how does hydrology affect fluxes and yields of sediment and macronutrients in intensively managed grasslands?

Q3. How does water quality from intensively managed grasslands compare to EU and UK recommended water quality standards at the farm-scale?

Materials and Methods

Field Site

This study was undertaken at the North Wyke Farm Platform in south-west England, UK (50046’10”N, 3054’05”W), described in more detail by Griffith *et al.* (2012). Figure 1a-b) shows the location and an aerial photograph of the North Wyke Farm Platform. Three large fields (6.6-7.6 ha) were chosen for sampling: Great Field (Field 2), Orchard Dean (Field 5), and Middle and Higher Wyke Moor (Field 8), as they represented the same scale of interest (Figure 1c - e). The total area of the farm is 68.4 ha.

During the sampling period for this study, the Farm Platform was managed as a conventional beef and sheep production system. Application of fertilizers at the Farm

Platform is in accordance with the Code of Good Agricultural Practice (DEFRA, 2010) and the Nitrate Vulnerable Zone guidelines (DEFRA, 2008) and is therefore considered to represent standard management practices for grassland systems. The grasslands of the Farm Platform are classified as intensively managed grassland. The three sampling fields were mainly used for cattle grazing (~30 cattle per field for ~ 75 - 95 days in all three fields) and sheep grazing (90 weaned lambs for 20 days in Field 8) during the summer of 2012. Additionally, the fertilizer N inputs during the sampling period were: Field 2: 522 kg (6.71 ha catchment size), Field 5: 503 kg (6.59 ha catchment size), Field 8: 495 kg (7.59 ha catchment size). No mineral P and K or Farmyard manure (FYM) were applied to the sampling fields during the monitoring period. Each field is hydrologically isolated (to a depth of 0.8 m) so that water leaving the field by sub-surface lateral flow or surface runoff is channelled via French drains into flumes, which are outfitted with water quality and quantity monitoring equipment. The hydrological soil types common on the North Wyke Farm Platform are representative of the most common hydrological soil types in England and Wales, covering approximately 13.9 % of the land area, (Boorman et al., 1995) and are typical for many areas under grassland management (Bilotta et al., 2008). The North Wyke soils are clay loams overlying shales of the Crackington formation with thin subsidiary sandstone bands (Bilotta et al., 2008, Harrod and Hogan, 2008) (soil series classifications according to different systems in Figure 1c). The soil series vary in their hydrological characteristics, but all have low water storage capacity and are slowly permeable up to 30 cm soil depth, where they have a clay-rich layer with very low hydraulic conductivities, allowing the (near to) hydrological isolation of the fields (Harrod and Hogan, 2008). The rainfall and annual temperature (1056 mm and 9.5

°C, respectively, mean of 40 years) at North Wyke are typical of much of the intensively managed grassland areas in the western UK (Harrod and Hogan, 2008).

Site Instrumentation

The Farm Platform instrumentation set-up is described in detail by Griffith *et al.* (2013). In short, every field is equipped with a rainfall and soil moisture monitoring station and the water leaving each field drains to a flume with a range of automated or semi-automated water quantity and quality monitoring equipment (locations shown in Figure 1c). All data are collected at 15-minute time intervals and transmitted by a remote telemetry unit (RTU) (www.adcon.at) via UHF radio telemetry to a centrally located base-station (a850 Gateway). The base station sends the raw data into the AddVantagePro software (www.adcon.at), which stores, processes and visualises these data (Griffith et al., 2013).

Hydrology and Macronutrient Monitoring

Hydrology data presented here were continuously monitored from April 2012 - April 2013. Each field is equipped with a tipping bucket rain gauge, with a 0.2 mm resolution (ADCON RG1, ADCON, Davis, USA). The discharge in each flume was measured by a 4230 Bubbler Flow Meter (Teledyne Isco, Lincoln, USA). The flow meter measures the flow height (h) in the flumes (www.tracomfrp.com), where the relationship between water height and flow rate is derived. Adcon SM1 soil moisture probes were used to measure soil moisture content (expressed as % soil moisture) through the soil profile: at 10, 20 and 30 cm soil depths (Griffith et al., 2013).

Hydrological data presented here were monitored from 1st April 2012 to 1st April 2013. Suspended sediment (SS) and the macronutrients carbon (total C), phosphorus (total P) and nitrogen (total oxidized nitrogen-N: TON_N) were monitored by a combination of different approaches, from making use of recent advances in continuous sampling equipment to manual sampling and laboratory analysis. Manual TC samples were taken from 1st April 2012 to 1st April 2013. Automatic and semi-automatic water quality data (SS, TON_N, TP) were used for the hydrological season, from 1st October 2012 - 1st April 2013, from when the sensors were fully installed, operational, and quality assured.

TON_N and SS were monitored continuously (flow permitting) by automated sensors located in a stainless-steel by-pass cell. The runoff water was automatically pumped into a by-pass cell every 15 minutes if flow was >0.2 L s^{-1} and the last sample was held in the by-pass cell when flow dropped <0.2 L s^{-1} (Griffith et al., 2013). The by-pass cell was required to keep sensors constantly submerged. TON_N was measured by an optical UV absorption sensor, NITRATAX plus SC (Hach Lange, Germany, Düsseldorf, (www.hach-lange.co.uk)) by passing UV light using a two-beam turbidity compensated photometer through the water and measuring the absorption. The absorption was then converted to TON_N concentrations, using the known UV light absorption rates of nitrate dissolved in water below the wavelength of 250 nm (Griffith et al., 2013).

As a proxy for SS, turbidity was continuously monitored using a YSI multiparameter sonde (6600V2, YSI, Yellow Springs, USA; www.ysi.com). To convert turbidity measurements (Nephelometric Turbidity Units, NTU) into SS (mg L^{-1}), the relationship between NTU and SS measured on the same samples was established (Table 1). Flow-proportional samples were taken via auto-sampler (ISCO 3700, Teledyne Isco, Lincoln, USA) and then analyzed in the laboratory for SS by filtering a known volume of sample through a pre-weighed, dry, glass fibre filter paper (1.2 mg pore size, Whatman GFC) followed by drying at 105 °C for 60 min and re-weighing to determine SS (Anon., 1980, Bilotta et al., 2008).

TP was measured by a Dr. Lange Sigmatax-Phosphax suite (Hach Lange, Hach Lange, Germany, Düsseldorf, Germany ; www.hach-lange.co.uk). The Phosphaxes are installed by the flumes and were manually switched on and off according to flow conditions. The Sigmatax-Phosphax suite extracts the water directly from the flume with the Sigmatax unit and homogenises a 100 mL sub-sample with a 3-minute ultra-sound homogenization. The sample is then digested with sulphuric acid-persulphate and added with molybdate antimony and ascorbic acid (Jordan et al., 2007).

TC was measured by taking flow-proportional grab-samples via auto-sampler, and then analyzed in the laboratory for TC. The sample bottles were collected and transferred to the laboratory within 24 hours, immediately refrigerated and analyzed within 48 hours of the event. A Scalar Formacs analyzer was used (Formacs^{HT} TOC/TN Analyzer, Scalar, Breda, Netherlands). The samples were not filtered and stirred before needle injection into the analytical instrument in order to provide a measurement as close to the total amount of carbon as possible (both dissolved organic and inorganic and particulate organic and inorganic C), here simply called total C. Within the Scalar Formacs, samples were heated to 850 °C and all forms of carbon were oxidised to carbon dioxide with cobalt oxide. Carbon dioxide was then determined by infrared detection. Total inorganic carbon (TIC) and total organic carbon (TOC) was automatically measured/ calculated in this procedure. The average % of TIC and TOC in all TC samples was calculated.

The quality of the data were assured by a) regular equipment calibration and drift checks, b) running a quality assurance programme, which incorporates sensor drifts, as well as filters out values that are above and below the known limits of detection by the sensors (www.rothamsted.ac.uk/farmplatform) and c) using the established relationship by cross-validation between automated measurements and laboratory analysis from samples taken within the same 15 minute slots during hydrological events to correct the data (Table 1).

All quality assured data was considered as the entire time series. To establish an event-based dataset, data taken during hydrological events were extracted from the continuous datasets by using a rule-based method (based on (Bilotta et al., 2008) defining hydrological events as a period of rainfall and discharge if the discharge increases in response to a period of precipitation above a certain threshold, based on field size. A storm event was considered to start when rainfall occurred after it had been dry for 2 hours and that rainfall was followed by either a net flow rise within the next 8 hours or a rapid flow rise (>0.5 L s^{-1} ha^{-1} within an individual 15-minute time slot) and the flow rose >1 L s^{-1} ha^{-1} within the event (Bilotta et al., 2008). A storm event was considered to end 4 hours after the last rainfall associated with an event. In total, 184 hydrological events (61 for Field 2, 64 for Field 5 and 59 for Field 8) were recorded between 1st April 2012 and 1st April 2013 using these rules.

To minimise uncertainty in the automated water quality data (SS, TON_N), hydrological events were ignored if there was >1h continuous missing data in the event time series, unless data was missing at the start of an event, when flow levels were too low for water to be pumped into the by-pass cell housing the sensors. All TP and TC events were used.

(Q1) To compare rates of sediment and macronutrient delivery from intensively managed grasslands other agricultural land uses, annual yields were calculated and compared to those reported in other grassland studies, mixed agricultural land-use studies and from arable land. To estimate annual yields, the entire time series was used: event-based data as well as data taken between hydrological events. Annual yield estimation is generally highly uncertain, as pollutant concentrations over unmonitored time steps have to be estimated (Bilotta et al., 2010), even with the near-continuous sampling regimes employed in this study. To estimate total yield of SS and TON_N, rating curves with 95 % confidence intervals were calculated and used to estimate concentrations of unsampled time steps (Bilotta et al., 2010, Glendell and Brazier, 2014). To estimate annual yield of TC and TP with low data coverage, the Walling and Webb Method 5 was used (Walling and Webb, 1985). This method provides the least biased yield estimation from time-series with little data coverage, but a continuous discharge record (Glendell and Brazier, 2014, Littlewood, 1992, Stutter et al., 2008, Walling and Webb, 1985).

(Q2) To examine the controlling factors on hydrology and how hydrology affects fluxes and yields of sediment and macronutrients, a) rating curves between discharge and pollutant concentrations were established throughout the entire time series, and b) stepwise-multiple linear regressions were conducted to understand event-based relationships between hydrology and responses of individual pollutants (considered significant at the $p \leq 0.05$) and c) the sedigraph/chemograph shapes and hysteresis effects were examined in relation to the hydrographs. Non-normally distributed variables were log transformed. All statistical analysis was conducted in Genstat (15th edition) (VSN International Ltd., Hemel Hempstead, UK).

(Q3) To compare water quality form intensively managed grasslands to EU/UK recommended water quality standards, the percentage of 15-minute time-step data that exceeded concentrations recommended by water quality guidelines was calculated (Bilotta et al., 2010). Such concentration-frequency curves express the pollutant delivery data in an ecologically relevant way: providing an indication of the duration of exposure to certain pollutant concentrations (Bilotta et al., 2010, Thompson et al., 2014). Water quality guidelines used are summarised in Table 2).

Results and Discussion

Figure 2 shows the whole hydrological time series, rainfall and discharge, and defines the hydrological events for each sampled field for the entire sampling duration (April 2012 - April 2013). The sampling year was wet, with 132 % of the average annual rainfall (of the past 40 years) occurring at the site. Results therefore illustrate the response of a landscape that is well connected to its surface waters, a scenario that is likely to occur more often in the future with wetter winters being predicted to occur in the UK with climate change (Jordan et al., 2012).

Table 3 summarises the hydrological characteristics for the three fields. Hydrographs across the Farm Platform demonstrate a rapid response to rainfall, with high variability across the year with runoff-coefficients being as low as 10 % during the summer and reaching as high as 133 % during the winter. The flashy hydrographs are typical for most areas of managed grasslands with similar soil types and topography (Bilotta et al., 2008, Harrod and Hogan, 2008). The soil moisture (%) at 10 cm depth was the most variable in all three fields (21 - 46 %), showing the lowest soil moisture levels during summer and the highest during winter/ wet periods. Moving down the soil profile, the soil was driest at 20 cm (24 - 34 %) (apart from dry periods, when the topsoil was driest) and was most constant at 30 cm soil depth (29 - 39 %). Soil conditions were considered as dry when soil moisture at 10 cm <40 %, 20 cm < 30 %, 30 cm <36 % and near saturation when soil moisture levels were above these thresholds, accounting for high variability and measurement errors. The high rainfall-runoff coefficients reflect both the fast response of these soils caused by the low water storage capacity that heavy clay soils support and the artificial drainage system. A number of events (6 out of 184 in total) had runoff-coefficients higher than 100 %, which may be due to a) spatial variability of rainfall rates, so that a lower total rainfall is calculated for a hydrological event than actually fell on the field, b) errors in rain gauge measurements, c) elevated “baseflow” at the start of storm events which were still affected by previous rainfall and d) water previously stored in the drainage system or the soil being flushed out in rapidly flowing events.

How do rates of sediment and macronutrient delivery from intensively managed grasslands compare to other agricultural land uses?

Table 4 summarises event-based diffuse pollutant fluxes, yields and the estimated annual yields from the three grassland fields. Apart from four TC events, all events for which water quality parameters were measured occurred when the soils were wet/near saturation or saturated (October- March).

Individual hydrological events in these grassland fields can yield up to 29.5 kg SS ha^{-1} (>1.2 µm), 84.5 g TON_N ha^{-1}, 49 g TP ha^{-1}, and 7.1 kg TC ha^{-1}. Estimated over the entire sampling duration (April 2012 - April 2013), these grassland fields yield up to 527.4 kg SS ha^{-1}, 3 kg TON_N ha^{-1}, 0.9 kg TP ha^{-1} and 179 kg TC ha^{-1} annually (Table 4). TC was approximately composed of 34 % TIC and 66 % TOC.

Erosion rates from the Farm Platform intensively managed grasslands (annual yields 182.2 - 527.4 kg SS ha^{-1}) are comparable to similar grasslands (during individual events (up to 14.9 kg SS ha^{-1} in Bilotta et al., 2008a)); 540 - 1210 kg SS ha^{-1} $year^{-1}$ (SS > 0.7 µm) (Bilotta et al., 2010)), mixed land use (640 kg ha^{-1} $year^{-1}$ average erosion rate for 56 mixed land-use catchments (Boardman et al., 1994), 255 - 588 kg SS ha^{-1} $year^{-1}$ in a mixed land-use catchment in the south-west UK (estimated for a 9 month period (Glendell and Brazier, 2014), 116 kg SS ha^{-1} $year^{-1}$ in a mixed-agricultural tributary to the River Dee in Scotland (Stutter et al., 2008)), and are within the ranges reported for arable land (average erosion rates for clay soils 611 kg ha^{-1} $year^{-1}$ (Deasy et al., 2009), 85 - 650 kg SS ha^{-1} $year^{-1}$ on an arable site in

England (Withers et al., 2006), 410 - 1910 kg ha^{-1} $year^{-1}$ on the UK Woburn Erosion Reference Experiment (Quinton et al., 2006) and 71 - 88 kg ha^{-1} $year^{-1}$ on Danish arable land (Kronvang et al., 1997)). The erosion rates presented in this study were measured as particles > 1.2 µm. If smaller particles had been included in the measurements, which are known to be preferentially transported from grasslands to surface waters, erosion rates may have been even higher (Bilotta et al., 2007). The high erosion rates from intensively managed grasslands reported here add to the growing evidence that grasslands have previously been underestimated in their erosion rates (Bilotta et al., 2008, Brazier et al., 2007), which should be acknowledged in land management guidelines and advice.

The TON_N losses from the Farm Platform were low (annual yields 0.92 - 3 kg ha^{-1}). Concentrations in discharge waters were similar to those from comparable grasslands (Granger et al., 2010, Sandford et al., 2013), but annual yields were lower than the losses reported in nearby grassland studies in the past (35.8 - 133.8 kg NO_3^- ha^{-1} $year^{-1}$ (Scholefield et al., 1996). The reduction in N leaching may be attributed to reduced N inputs, controlled by NVZ guidelines in the UK (DEFRA, 2008). Grasslands monitored by Scholefield et al. (1996) received more than double the N inputs (btw. 200 - 400 kg N ha^{-1} $year^{-1}$) that the sampled fields in this study received (80 kg N ha^{-1} $year^{-1}$). The annual TON_N losses from the Farm Platform were significantly lower than those reported for mixed land-use (17.4 kg NO_3^- ha^{-1} $year^{-1}$ in a mixed grassland and arable catchment in Scotland (Stutter et al., 2008)) and arable land-use (6 - 60 kg NO_3^- ha^{-1} $year^{-1}$ in Denmark (Erikson and Vinther, 2002), average losses of 30 kg NO_3^- ha $year^{-1}$ on the Rothamsted Research Broadbalk Wheat experiment, UK, (Goulding et al., 2000), >20 kg NO_3^- ha $year^{-1}$ from tile drained arable land in USA (Schilling et al., 2013)).

TP losses reported herein (up to approx. 0.9 kg ha^{-1} $year^{-1}$) compare well with typical TP losses reported for UK agricultural catchments (1 kg P ha^{-1} $year^{-1}$ (Heathwaite et al., 2005)), P losses measured in nearby grasslands (0.21 kg SRP ha^{-1} $year^{-1}$ of soluble reactive P (Hawkins and Scholefield, 1996) event yields of up to 50 g P ha^{-1} (Bilotta et al,. 2008)), other grasslands (0.54- 0.7 kg TP ha^{-1} $year^{-1}$ in Irish grassland catchments (Jordan et al., 2012)), mixed land-use (0.37 kg TP ha^{-1} $year^{-1}$ in a Scottish mixed catchment(Stutter et al., 2008)), and arable land (Irish arable land 0.18 - 0.79 kg TP ha^{-1} $year^{-1}$ (Jordan et al., 2012), 2 kg TP ha^{-1} $year^{-1}$, UK, (Smith et al., 2001), 0.17 - 0.73 kg TP ha^{-1} over one winter on a clay soil, UK, (Deasy et al., 2009). The TP losses add to the growing evidence that grasslands contribute significantly to P levels in surface waters.

The comparison of TC yields from the Farm Platform with yields from grasslands or other land uses is constrained because a) the majority of C studies are conducted on peaty or natural catchments and b) studies mostly measure dissolved organic carbon (DOC) (Worall et al., 2012) rather than the TC data presented here. Results show that TC was approximately divided into 34 % TIC and 66 % TOC. Despite constraints in comparing DOC and TC/TOC yields, C exports from these grasslands (up to 179 kg ha^{-1} $year^{-1}$ (≈ 118 kg TOC ha^{-1} $year^{-1}$)) exceeded those of typical UK/ European catchments (the average of 85 UK catchments 7.7 - 103.5 kg DOC ha^{-1} $year^{-1}$ (Hope et al., 1997b)) and the range reported for an agricultural watershed in Midwestern USA 14.1 - 19.5 kg DOC ha^{-1} $year^{-1}$ (Dalzell et al., 2007), as well as the range reported for German grasslands (8 - 55 kg DOC ha^{-1} $year^{-1}$ (Don and Schulze, 2008)). In addition, the TC annual rates are comparable to the combined rate of DOC and total particulate carbon measured over a 9 months period in an agricultural catchment in south-west UK (27.18 - 130.46 TPC+DOC kg ha^{-1} (Glendell and Brazier, 2014)). The high rates of C export support the notion that agricultural catchments, particularly pastoral systems, yield higher C losses than forested or semi-natural catchments (Graeber et al., 2012). Increased C losses from agricultural catchments may be caused by manure applications and grazing causing an increase in soil dissolved organic matter concentration through stimulation of microbial activity and increased oxygenation of the agricultural soils (Chantigy, 2003, Heitkamp et al., 2009) and higher erosion rates (Hope et al., 1997a, Quinton et al., 2010) transporting high rates of particulate carbon. The few studies that quantified DOC and particulate carbon in agricultural landscapes showed a high contribution of particulate carbon to overall carbon losses by low DOC-TOC ratios (up to 1.01) (Glendell and Brazier, 2014) and 58 % and 42 % of organic C losses as DOC and POC, respectively (Stutter et al., 2008). Therefore, studies measuring C cycling and C losses from agricultural land to freshwater should include total carbon as up to 50 % of C losses may not be accounted for (Quinton et al., 2006).

It is important to note that erosion rates and losses of TP and TC were equivalent to those reported for mixed agricultural land and arable land use, even though the fertilization rates and stocking densities were relatively low for intensively managed grasslands and the fields had not been ploughed recently, a common management practice in intensively managed grasslands. Therefore, results presented herein demonstrate that sediment, TP and TC losses from intensively managed grasslands are significant.

What are the controlling factors on hydrology and how does hydrology affect fluxes and yields of sediment and macronutrients in intensively managed grasslands?

Total event rainfall described a significant amount of variance in total event discharge in Field 2, Field 5 and Field 8 (R^2= 0.61, 0.62 and 0.77, respectively). Also, variance in discharge peaks in all fields were significantly related to total rainfall and peak rainfall intensity in the case of Field 2 and Field 5, but to a lesser extent (R^2= 0.47, R^2= 0.52, respectively). Testing for antecedent soil moisture conditions as a control, sampled events were divided into events that occurred when soils were wet and dry. The amount of total discharge described by total rainfall was higher in Field 2, 5 and 8 when soils were wet (R^2= 0.85, 0.87, 0.94, respectively) and in Field 2 and 5 when soils were dry (R^2= 0.82, 0.63, respectively) (Figure 3).

The flashy discharge response to rainfall indicates a fast lateral movement of water either on the soil surface or through the surface layers via cracks and macropore flow (Granger et al., 2008). The topsoil (10 cm) is the most responsive to rainfall and temperature changes. Upon reaching field capacity in the topsoil, water is most likely to run off laterally down slope rather than further infiltrating into the driest 20 cm soil layer. The 30 cm clay-rich soil layer maintained the most constant soil moisture. Antecedent soil moisture conditions were a controlling factor on discharge, despite soil moisture measurements being highly variable. The highest runoff coefficients occurred when the soil was near saturation prior to a rainfall event, suggesting that hydrological events are mostly driven by saturation excess processes during wet periods (most of the sampling year as it was a particularly wet year) and only a small proportion of events was driven by infiltration excess processes during the summer months (Granger et al., 2010).

Throughout the entire time series, discharge explained significant amounts of variation in concentrations of SS (Field 2: R^2=0.62, Field 5: R^2=0.71, Field 8: R^2=0.65), TON_N (Field 2: R^2=0.5, Field 5: R^2=0.76, Field 8: R^2=0.24) and TP (Field 2:

R^2=0.53, Field 5: R^2=0.59, Field 8: R^2=0.84), but no variation in TC concentrations. Total storm event discharge explained a large amount of variation in event loads for SS, TP and TC, but only very small amounts or no variation at all for TON_N (Figure 4). Storm event analysis could not be split into events with dry or wet antecedent soil conditions, because almost all events monitored for water chemistry occurred when soil conditions were wet.

SS, TON_N, TC and TP showed different behaviours throughout hydrological events. Sedigraphs, TON_N and TP chemographs showed a consistent response in relation to the hydrograph in all hydrological events (unimodal, bimodal and multimodal). The hydrographs, sedigraphs and chemographs for an example hydrological event for all three fields are presented in Figure 5.

The sedigraph peak(s) generally preceded the hydrograph peak(s), following a clockwise hysteresis pattern. Clockwise hysteresis in SS has previously been reported in field- to small catchment scale studies in temperate grasslands (Bilotta et al., 2008, Granger et al., 2010) as well as large-scale catchments (Asselman, 1999). Clockwise hysteresis indicates that it takes less time to mobilise, transport and deliver SS to the flumes than the time to generate peak discharge, because the sediment source is located in close spatial proximity to the flume or in closer proximity in relation to the source area generating the flow (Klein, 1984). Here it is suggested that sediment particles deposited in or near the drains from the last hydrological event or from periods of high flows in the same event are flushed out when discharge increases, showing an influence of past rainfall and antecedent moisture conditions and explaining why peak flow was not well correlated with peak SS concentration (Granger et al., 2010).

The TP peak either coincided with the SS peak, occurred between the SS and the hydrograph peaks, or coincided with the hydrograph peak, leading to clockwise hysteresis or no hysteresis. After the initial TP peak, concentrations dropped rapidly, but not as rapidly as the concentration drops observed for SS. Throughout the entire time series, SS explained significant amounts of TP (62 – 72 %). Higher TP concentrations during hydrological events as opposed to base flow indicates TP sources are mobilized by hydrological transfer from flashy, or fast response runoff (Haygarth et al., 2004, Jordan et al., 2012). Clockwise hysteresis for P has been shown before for TP as well as PP and SRP in agricultural catchments (Bowes et al., 2005, House and Warwick, 1998). The proportion of PP and SRP in TP could not be inferred from the data in this study. TP and SS concentration peaks often coincided and a control of SS over TP was confirmed by significant correlation throughout the entire time series, but SRP is still likely to play a role in TP losses, as TP storm concentrations after the first peak (likely to be driven by SS associated P) were still elevated compared to baseflow concentrations. In addition, the proportion of TP and SRP is likely to change both spatially and temporally throughout events and between events (Granger et al., 2010, Haygarth and Jarvis, 1997). TP accumulates at the soil surface either adsorbed to sediment particles or stored in dissolved form in the soil water in the topsoil layer. TP in both particulate or dissolved forms can then be mobilized by rainwater either through physical mobilization of sediment particles, desorption of P from particles or dissolved P enriched interflow transported through macropores; explaining the likely contribution of SRP and PP to overall TP losses (Gächter et al., 2004).

TON_N decreased as discharge increased and increased again towards the end of the event; TON_N concentrations often fell below the limits of analytical quantification within events (<0.1 mg TON_N L^{1}). The reduction of TON_N throughout hydrological events with discharge indicates a dilution of TON_N by flow. Such N dilution by rainwater has been reported elsewhere for both components of TON_N : nitrate (NO_3^-) (Gächter et al., 2004, Granger et al., 2010) and nitrite (NO_2^-) (Granger et al., 2010). Both nitrate and nitrite are formed and held within the soil and are mobilized from soil water by diffusion to water moving through the soil system (Gächter et al., 2004). When water moves rapidly over the soil surface or through the soil macro pores, it has limited opportunities for diffusion (Granger et al., 2010). Additionally, soils with high clay content are retentive of NO_3^- (Don and Schulze, 2008), with North Wyke soils having relatively high clay contents (up to 25 % clay content (Harrod and Hogan, 2008, Peukert et al., 2012). However, note that TON_N can show a different response to discharge shortly after fertilizer N was applied (Granger et al., 2010).

The TC chemograph varied between storm events, but often TC concentrations decreased slightly as discharge increased and TC concentrations showed a peak prior to the peak discharge, often coinciding with the SS peak. Mean TC concentrations were generally reduced throughout the year in all fields. The TC dataset is more uncertain because it had larger data gaps than the other pollutants monitored. Most importantly, if TC data were missing at the start and the end of an event, little indication was given about between-event concentrations. Therefore, less understanding of TC dynamics can be inferred compared to the other pollutant dynamics, highlighting the advantages of high frequency monitoring. Erosion associated C losses, are likely to play an important role in overall C losses with TC peaks often coinciding with SS peaks and concentrations being overall higher than in studies on analogous grasslands measuring only DOC (Sandford et al., 2013). However, SS did not explain any variation in TC throughout the entire time series, which may be attributed to the reduction of TC concentrations throughout the year. The reduced TC concentrations throughout the year may be explained by several factors: a) enhanced organic matter decomposition during summer followed by a flushing of C upon rewetting, b) enhanced C inputs by grazing animals and manure additions during the summer, c) higher temperatures and or d) enchytraeid worms influencing microbial activity by soil aeration (Evans et al., 2005).

How does water quality from intensively managed grasslands compare to EU and UK recommended water quality standards at the farm-scale?

Table 5 summarises guideline exceedance frequencies from the monitored grassland fields. Guidelines set for SS and soluble molybdate reactive phosphorus were frequently exceeded by the concentrations of SS and TP measured in this study. 11.6 - 13.9 % of all SS samples (low and high flows) exceeded the FFD guideline of 25 mg L^{-1}. When the SS standard was lowered to 10 mg L^{-1}, 31.4 – 54.1 % of the SS samples exceeded that standard. This lower 10 mg L^{-1} is close to the water quality guideline set for turbidity in agricultural ecoregions in the USA (USEPA, 2000). If particles <1.25 μm had been included in the suspended sediment measurements, the percentage of all samples exceeding the guideline concentrations would have potentially been even higher. SS on a nearby grassland site, more intensively managed than the Farm Platform, exceeded the FFD guideline for approximately 50 % of the hydrological season (SS >0.7 μm (Bilotta et al., 2010)) and two Irish catchments showed an annual exceedance of 8.3 - 17.8 % (Thompson et al., 2014). However, note that the suitability of the blanket guideline (across all water bodies in the UK, all seasons) of 25 mg SS L^{-1} has recently been heavily criticised (Bilotta et al., 2012). The high erosion levels from these grasslands are likely to contribute significantly to the documented water quality issues in the receiving Taw River. Low salmonid fish numbers in the Taw are generally attributed to high sedimentation levels and physical degradation of spawning gravels by siltation (EA, 2009, Haygarth et al., 2005b).

Approximately half of all the TP discharge water samples failed to meet what is considered as good ecological status (<0.04 mg SRP L^{-1}) and 4.2 – 18.3 % failed to reach moderate ecological quality (<0.15 SRP mg L^{-1}) (TAG, 2008). Therefore, grasslands, which cover 80 % of the Taw River catchment (similar management and soil types to the North Wyke Farm Platform) contribute to the documented P issue in the Taw river (EA, 2009, Haygarth et al., 2005b). Note that TP exceedance of the guideline set for soluble reactive P (SRP) may overestimate the true exceedance. However, not only SRP is bioavailable in surface waters, but particulate P and organic P also have the potential to be recycled within rivers and become bio-available through time (Darch et al., 2013). Elsewhere, for example in the USA, standards are set as TP concentration thresholds; in agriculturally dominated ecoregions around 0.03 - 0.08 mg TP L^{-1} (USEPA, 2000), sitting just between the values of good and moderate status that TP concentrations were compared to in here. Furthermore, climate change scenarios predict wetter winters and drier summers for the UK in the future, which are likely to increase winter discharge events and reduce river flow levels during summers (Jordan et al., 2012), both factors that already cause high P loadings in the Taw River.

TON_N concentrations in discharge waters did not exceed the Nitrates Directive Nitrate-N guideline of 11.3 mg L^{-1}, which is a positive result, as the entire Taw River catchment is designated as a NVZ. However, the fact that TON_N levels were acceptable under the current guideline does not necessarily mean that these grasslands do not pose a threat to surface waters in terms of N: a) Nitrite, even though only accounting for a small proportion of TON_N (4 % in Granger *et al*. (2010) on a near-by site), is potentially toxic to aquatic fauna even in very low concentrations and impacts on ecosystem health by leading to increased biological oxygen demand within surface waters (Lewis and Morris, 1986) and b) there is growing evidence that organic N can be rapidly available to plankton and microorganisms in estuarine and marine environments (See et al., 2006). Therefore, the export of organic N that was not accounted for in this study, may have significant impacts on the receiving Taw estuary, already classified as eutrophic (Maier et al., 2012).

There are no known EU/UK C guidelines in terms of freshwater and marine ecosystem health. The TC concentrations from these grasslands may make a potentially significant contribution to reducing dissolved oxygen in the water and therefore indirectly affecting aquatic fauna and flora (Edwards et al., 2008, Sandford et al., 2013). Excessive biological oxygen demand have been reported for stretches of the Taw river (Haygarth et al., 2005b). Such results support the need for explicit guidelines for TC in surface waters.

The concentrations found in discharge waters in this study were comparable to the range of concentrations monitored in the Taw River (weekly measured concentrations approx. 400 m downstream from the flume at Field 2: SS: 0 - 31 mg L^{-1} (particle sizes not presented), PP + total dissolved P: <0.02 - 0.37 mg L^{-1}, NO_3 + NO_2^-= TON: 4 - 10.2 mg L^{-1}, DOC: 1 - 16 mg L^{-1}) (Jarvie et al., 2008). Additionally, a study in the Taw catchment estimated that diffuse sources of P contributed at least 60 % of the annual P flux of the Taw River (Wood et al., 2005). Therefore, the sediment and macronutrient concentrations seen in the discharge waters and yields from these intensively managed grassland fields contribute significantly to overall concentrations and yields in the receiving surface water and dilution is not likely to be high.

Conclusion

This study adds to the growing evidence that grasslands have been previously underestimated in terms of their erosion rates and TP losses. TON_N yields were less than those reported in earlier grassland N studies and arable sites, possibly due to reduced N inputs as a result of NVZ regulations. TC losses were particularly high, adding to the evidence that agricultural catchments and intensively managed grasslands in particular can yield high C losses (**Q1**).

Hydrological events were mostly saturation-excess driven in the winter as opposed to infiltration-excess driven during the summer. SS and TP were mobilized and transported by hydrological flushing from the soil surface, while TON_N was diluted by rainwater, because diffuse from its soil water is limited by the rapid rainwater movement. TC showed varied responses in hydrological events, with concentrations decreasing from summer to winter both dissolved and C likely to contribute to the overall TC concentrations and yields (**Q2**).

Intensively managed grasslands pose a significant threat to water quality. SS and TP concentrations frequently exceeded EU water quality guidelines and high TC exports are likely to reduce dissolved oxygen in surface waters. (**Q3**). This study has shown the usefulness of a farm-scale research platform (the North Wyke Farm Platform) and has established a baseline understanding of how this ecosystem functions in terms of multiple pollutants at management-relevant scales. Future work will monitor field to farm-scale perturbations of land management in relation to this baseline understanding to understand the effects of improved land management on these multiple pollutant losses

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Figure 1. Description of the sampling site: the North Wyke Farm Platform. a) Location of the Farm Platform and b) the three sampling fields within the Farm Platform. c) - e) individual sampling field topography, soil types, location of flumes, rain gauges and soil moisture probes for Field 2 (6.71 ha), Field 5 (6.59 ha) and Field 8 (7.59 ha), respectively. Names for soil types under international classifications: Denbigh/ Cherubeer (Avery, 1980): FAO Stagni-eutric cambisol, USDA: Dystic eutrochrept; Halstow (Avery, 1980): FAO Stagni-vertic cambisol, USDA Aeric haplaquept; Halsworth (Avery, 1980): FAO Stagni-vertic cambisol, USDA Typic haplaquept (in Harrod and Hogan, 2008).

Figure 2. Rainfall (mm hour^{-1}), discharge (L s^{-1}) and hydrological events for all three intensively managed grassland sampling fields for the entire sampling duration (April 2012- April 2013). All data was monitored and is expressed at 15-minute time steps. Hydrological events were defined on the basis of rainfall and a discharge response rising above certain thresholds, based on field sizes. Occurrence of hydrological events was also similar, but peak flow rates vary between the fields. Soil moisture levels (not included in this figure) were high or near to saturation in mid-end of April, mid-end of June and throughout most of October- March. Note the period of July-August, when no hydrological events occurred, despite high rainfall rates, because soils were dry and/or dried out quickly after rainfall.

Figure 3. Relationship between total event rainfall (mm) and total event discharge (mm, normalised by field area) for events that occurred when a) soils were dry and b) soils were near saturation, in all three intensively managed grassland fields. Antecedent soil moisture conditions were defined as: dry soils when soil moisture was at 10 cm <40%, 20 cm < 30%, 30 cm <36% and near saturation when soil moisture levels were above these thresholds. A rainfall event of 40 mm total rainfall occurring when antecedent soil conditions are dry is expected to have discharges of approximately 10- 18 mm. The same rainfall event occurring when soils are wet is likely to trigger 22- 30 mm of discharge.

Figure 4. Correlation between total event discharge and a) SS event yield (Field 2: R^2=0.81, Field 5: R^2= 0.8, Field 8: R^2= 0.82), b) TON^N event yield (Field 2: R^2=0.14, Field 5: R^2= 0.34, c) TP event yield (Field 2: R^2=0.0.78, Field 5: R^2= 0.8, Field 8: R^2= 0.83), d) TC event yield (Field 2: R^2=0.65, Field 5: R^2= 0.49, Field 8: R^2= 0.57), e) correlation between SS event yield and TP event yield (Field 2: R^2=0.77, Field 5: R^2=

0.96, Field 8: R^2= 0.99). Trendlines are only shown and R^2 values given for significant correlations.

Figure 5. Detailed description of one particular storm event which occurred on the 25-26/01/2013 in the three sampled intensively managed grassland fields: Field 2 start of the histological event at 16:45, lasting for 10.25 h, Field 5 start of the hydrological event at 11:45, lasting for 15 h, Field 8 start at 16:15, lasting for 10.75h. For each field, rainfall and hydrographs are presented as well as showing hydrograph shape and chemograph/sedigraphs shapes (Field 2 a)-e), Field 5 f)-j), Field 8 k)-o). Each parameter was measured at 15 minute time-steps, apart from TC, for which manual samples are shown as individual dots. The x-axis shows hourly ticks.

Table 1.
Correlation functions between automated measurements and laboratory measurements (event-based samples taken in the same 15-minute slot). Note: the Nitratax was factory calibrated in Jan 2013, so the TON_N calibration functions are presented pre and post factory calibration. The two deployments of the Phosphaxes were: 1st deployment 14/12/2012-20/1202012, 2nd deployment 25/01/2013-12/02/2013.

	TON_N Nitratax-TON_N (pre factory calibration)	TON_N Nitratax-TON_N (post factory calibration)	TP Phosphax - TP (1st deployment)	TP Phosphax - TP (2nd deployment)	Turbidity NTU-SS
Flume 2	y=1.07x-0.51 (R^2=0.77 , N=12)	y=0.53x+0.06 (R^2=0.4, N=24)	y=1.53x-0.04 (R^2=0.89, N=16)	y=1.29x-0.005 (R^2=0.82, N=11)	
Flume 5	y=0.62x-0.32 (R^2=0.76 , N=17)	y=0.53x+0.002 (R^2=0.47, N=70)	y=0.78x+0.03 (R^2=0.69, N=16)	y=0.91x+0.02 (R^2=0.86, N=16)	y=1.18x+0.05 (R^2=0.75, N=256)
Flume 8	y=0.7x-0.01 (R^2=0.7, N=17)	y=0.83x-0.01 (R^2=0.68, N=27)	y=1.26x-0.02 (R^2=0.72, N=19)	y=1.08x-0.004 (R^2=0.72, N=17)	

Table 2.
The water quality guidelines and their specific pollutant concentrations used to compare to the pollutant concentrations measured on the North Wyke Farm Platform. Note that guidelines were set for specific fractions of the pollutants measured in this study. No guideline for TC or specific fractions of C could be found.

Pollutant in this study	Guideline used	Chemical guideline set for	Guideline concentration
SS	EU Freshwater Fisheries Directive	No specified SS size	25 mg L^{-1}
	lower SS standard than FFD used because FD was suggested to be too high (Bilotta et al., 2012, study on SS background levels of 638 UK water courses in reference state)	No specified SS size	10 mg L^{-1}
TP	"good ecological status" (GES) specifically for the upper Taw catchment (UK Technical Advisory Group (TAG) standard devised for the EU Water Framework Directive)	dissolved molybdate reactive Phosphorus (MRP)	<0.04 mg L^{-1}
	"moderate ecological status" (MES) specifically for the upper Taw catchment (UK Technical Advisory Group (TAG) standard devised for the EU Water Framework Directive)	dissolved molybdate reactive Phosphorus (MRP)	<0.15 mg L^{-1}
TON_N	EU Nitrates Directive	$Nitrate_N$ (NO_3^--N)	<11.3 mg L^{-1}
TC	n.a	n.a	n.a

Table 3.

General overview of the three fields' hydrological characteristics (mean, median, minimum and maximum) and number of events captured.

Field No.		Total R	Q total	Q peak	Rainfall-runoff coefficient
		mm	1000L	$L\ s^{-1}$	%
2	Mean	15.9	517.2	28.1	46.2
	Median	11.2	352.5	21.4	46.6
	Minimum	2.4	70.4	6.9	12
	Maximum	50	2124.4	97.6	82
	N	61	61	61	61
5	Mean	14.2	586.7	37	63
	Median	9.8	405.6	30.8	60.9
	Minimum	2	52.5	6.8	9.9
	Maximum	50.4	2424.9	105.8	132.7
	N	64	64	64	64
8	Mean	15.8	701.3	38.5	54.3
	Median	11	452.2	32.9	52.7
	Minimum	3.2	87.7	8.6	12.7
	Maximum	46.8	2583.6	103.9	128.7
	N	59	59	59	59

Table 4.

General overview of the three fields' sediment ant macronutrient annual yields, event yields, and peak event concentrations. All measurements are normalised by field area. Where appropriate, mean median, minimum, maximum and number of captured events are shown.

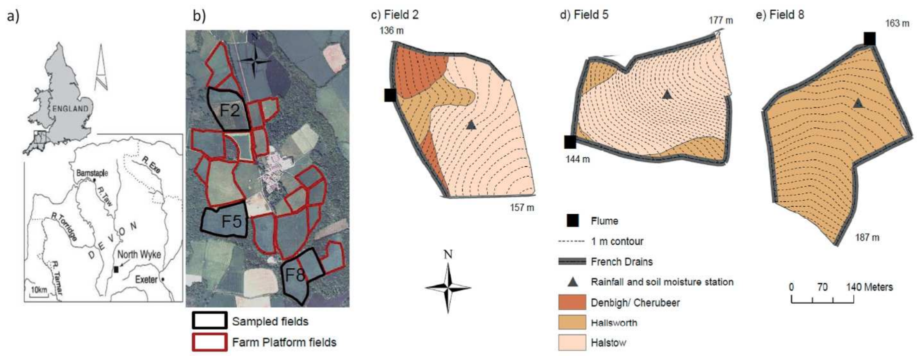
Field		Suspended Sediment (>1.2 µm)			Total oxidized Nitrogen$_N$			Total Phosphorus			Total Carbon		
		annual yield † kg ha^{-1}	event yield kg ha^{-1}	peak event conc. mg L^{-1} ha^{-1}	annual yield † kg ha^{-1}	event yield g ha^{-1}	peak event conc. mg L^{-1} ha^{-1}	annual yield ‡ kg ha^{-1}	event yield g ha^{-1}	peak event conc. mg L^{-1} ha^{-1}	annual yield ‡ kg ha^{-1}	event yield kg ha^{-1}	peak event conc. mg L^{-1} ha^{-1}
2	Mean	191.3	3.5	19.9	1.7	11.9	0.2	0.42	7.4	0.16	122.2	2.2	4.5
	Median		2.2	17.3		6	0.1		5.58	0.14		1.9	4.8
	Min	182.2	0.3	6.6	1.4	0.2	0		0.88	0.04		0.5	2.2
	Max	194.3	17.7	51.4	1.8	84.5	2		39.13	0.48		6.1	6.4
	N		41	41		41	41		18	18		11	11
5	Mean	441.4	5.9	38	3	16.2	0.2	0.87	10.65	0.22	179.1	2.4	5.2
	Median		4.3	29.3		11.5	0.2		8.1	0.23		2.4	5.3
	Min	433.9	0.5	9.5	2.9	4.3	0.1		1.09	0.06		0.3	2.7
	Max	527.4	25.5	109.2	3	70.1	0.3		49	0.37		7.1	6.9
	N		30	30		42	42		18	18		12	12
8	Mean	218.1	3.7	17.8	1	10.8	0.1	0.44	7.14	0.18	109.4	1.9	3.2
	Median		2.5	14.3		2.5	0		6.18	0.17		1.5	3.3
	Min	213.4	0.4	5.1	0.9	0	0		0.55	0.03		0.4	1.1
	Max	220.9	17.1	51.1	1	63.1	0.1		21.15	0.44		4.6	5.7
	N		37	37		19	19		17	17		12	12

† Calculated with rating curves

‡ Calculated with Walling and Webb Method 5 (Walling and Webb, 1985)

Table 5.
% of all SS, TP and TON_N samples exceeding water quality guidelines in the discharge water of three improved grassland fields. SS guideline: EU Freshwater Fisheries Directive and levels below this guideline, TP guideline: EU Water Framework Directive (note that the guidelines are set for soluble molybdate reactive Phosphorus SRP), guideline for TON_N: EU Nitrates Directive (note that the guidelines are set for NO_3^--N).

	Field 2	Field 5	Field 8
SS			
> FFD (>25 mg SS L^{-1})	11.64	13.9	13.37
> 10 mg l^{-1}	41.88	31.4	54.1
TP			
fail "good ecological status" (>0.04 mg SRP L^{-1})	49.1	52.3	59.27
fail "moderate ecological status" (>0.15 mg SRP L^{1})	7.11	4.2	18.31
TON_N			
fail Nitrates Directive (>11.3 mg NO_3^--N L^{-1})	0	0	0



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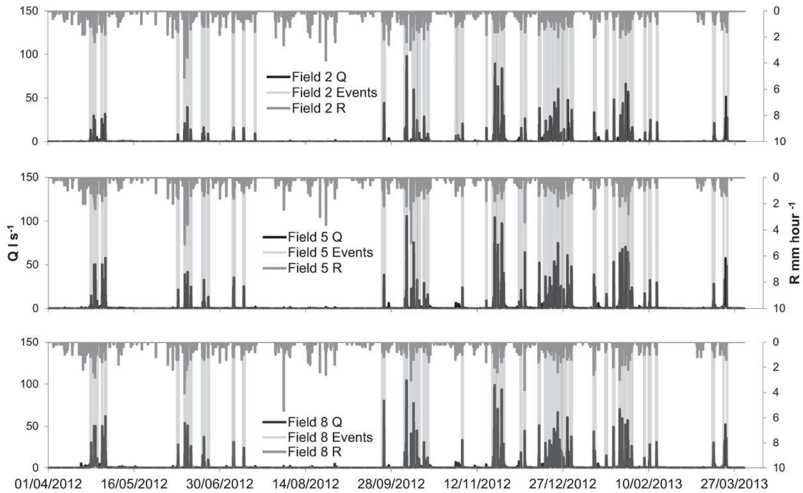

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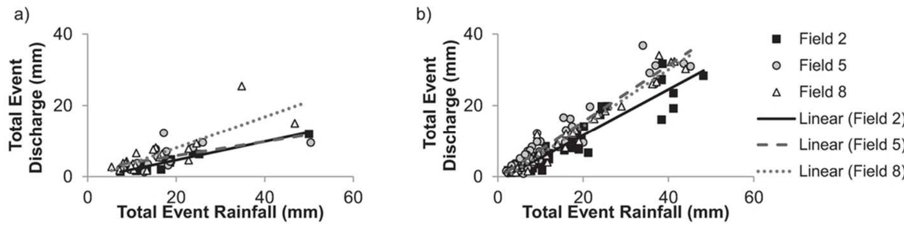

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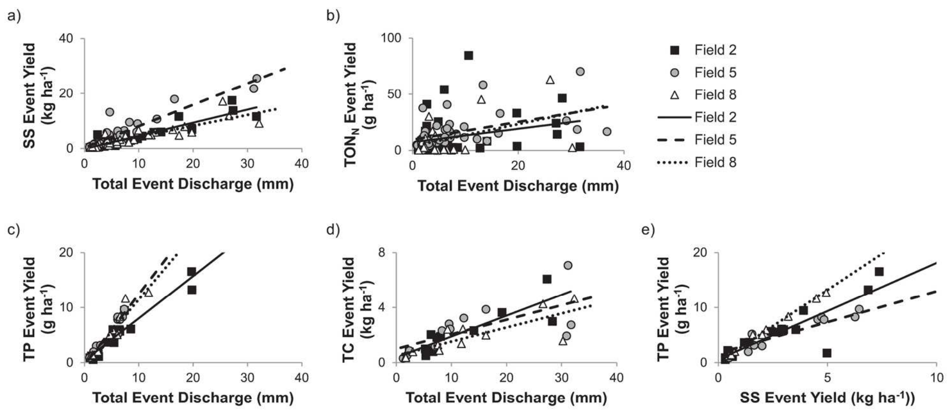

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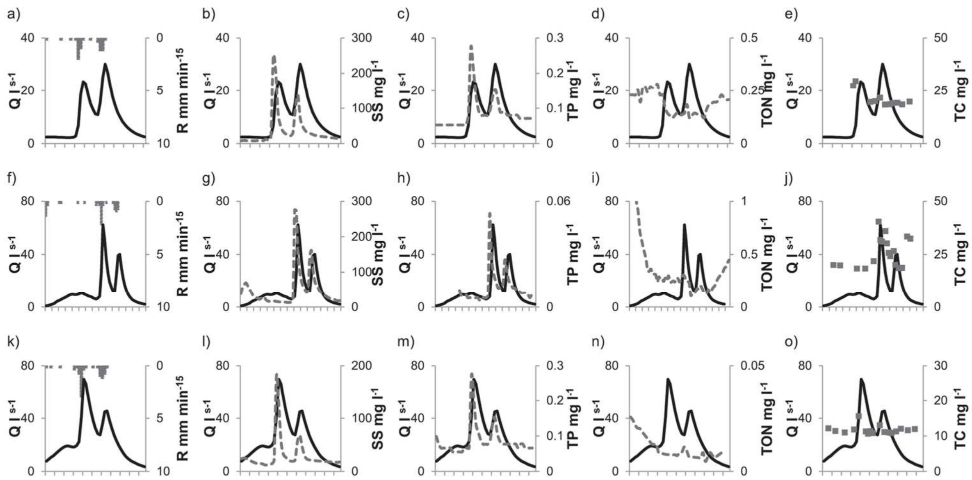

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