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Ranking connectivity risk for phosphorus loss along agricultural drainage ditches

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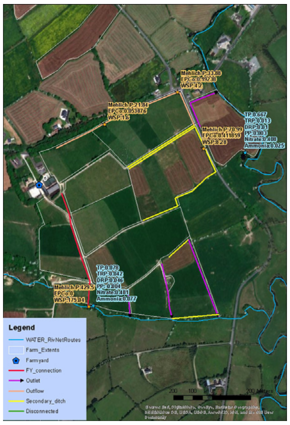
HIGHLIGHTS

- Varying levels of connectivity exist between agricultural surface drainage ditches and surface waters.
- Landscape position and sediment P chemistry describe the risk of P loss in 5 categories of ditches.
- Highest risk attributed to ditches connecting farm yards and outlets to streams.
- Legacy P accumulated in ditch sediment from yards and at outlets over time.

GRAPHICAL ABSTRACT



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ABSTRACT

Agricultural drainage systems comprising both in-field pipe drains and surface ditches are typically installed to remove excess water from agricultural land. These drainage networks can provide connectivity between phosphorus (P) sources and surface waters thereby increasing the risk of P loss to rivers and streams. The objective of this study was to derive a farm-scale drainage ranking that categorises drainage ditches in terms of P loss risk based on connectivity and physic-chemical characteristics. Ten pilot farms were selected to characterise drainage networks through ground survey and, sediment and water sampling. Five drainage ditch categories were derived based on landscape setting and connectivity. Each category recorded soluble and reactive P concentrations above environmental water quality standards. To assess the risk of surface ditches as a connectivity vector between agricultural P and surface waters ditches were ranked in order of P loss risk by integrating landscape position and sediment P chemistry. Elevated sediment P with high equilibrium P concentration (EPCo) were associated with ditches connected to farm yards, and in sediment sampled at ditch outlets, suggesting P deposition over time indicative of a legacy P source. The greatest risk of P loss was attributed to ditches connecting farm yards to streams, and ditches that connected the drainage network to surface waters, or Outlets. These results rank connectivity risk for P loss along agricultural drainage ditches for farm level risk assessment to target P loss mitigation measures to the appropriate locations.

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1. Introduction

Phosphorus (P) is responsible for the pollution of a large proportion of surface waters across the globe due to its role in eutrophication and its negative environmental impacts (Torrent et al., 2007). The transfer of P to surface waters can be summarised as a continuum of interlinking stages from source, via mobilisation and delivery, to pollution impact on receiving waters as described by Haygarth et al. (2005). The mobilisation stage involves the separation of P molecules from their source via geochemical desorption, biological solubilisation or physical detachment with the rate and extent of these processes increasing under certain soil conditions, particularly high moisture and organic matter (OM) contents, and management regimes (Daly et al., 2001; McDowell et al., 2001; Daly et al., 2015). This mobilised P is typically carried via surface and subsurface pathways and delivered to surface waters where it acts as a pollutant (Haygarth and Jarvis, 1999; Deakin et al., 2016; Reid et al., 2018).

The extent and efficiency of the pathways that transport nutrients to surface waters is a measure of the connectivity of the surrounding landscape. Connectivity is defined as the transfer of energy and matter between two landscape zones or within a system as a whole (Chorley and Kennedy, 1971) and hydrological connectivity at the catchment scale has been discussed extensively by Fryirs et al. (2007), Wainwright et al. (2011) and Masselink et al. (2017). Conversely, dis-connectivity is the isolation of such landscape zones and intercepting the nutrient flow pathway is often considered the most effective measure to mitigate nutrient transport to surface waters (Deakin et al., 2016). There are two principal components of hydrological connectivity. Firstly, the spatial distribution of connected zones and secondly, the magnitude or frequency and duration, of the connections (Wainwright et al., 2011). This concept can be applied at a range of scales from catchment to field. At the farm and field scale hydrologic connections (aka preferential flow pathways) can take the form of an array of features including micro-topographical features within a field such as field boundaries, farm tracks and roadways. Furthermore, critical and variable source areas that are prone to saturation throughout the year also form important hydrologic connections on farms.

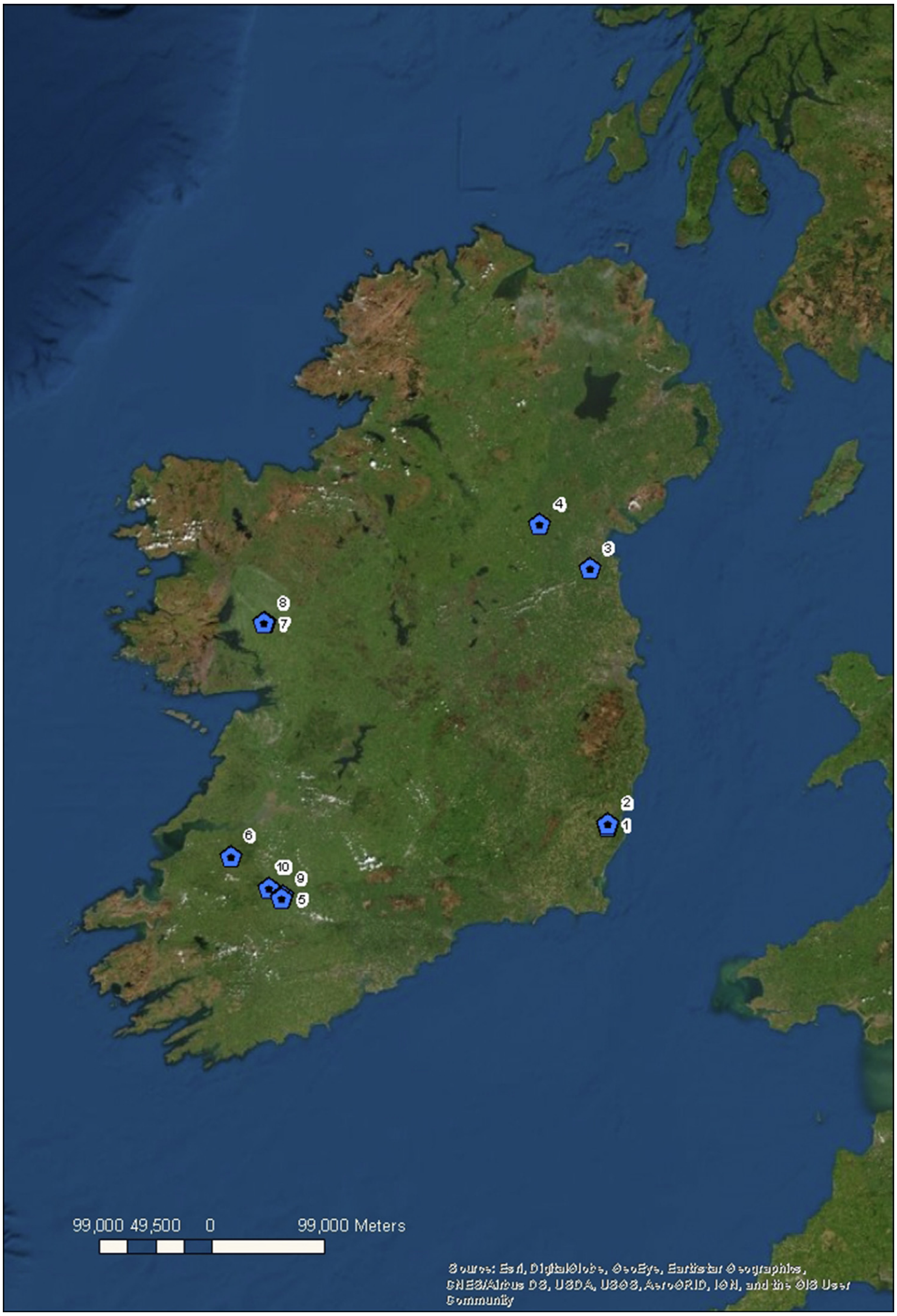
Agricultural production in Ireland is primarily based on pasture-based livestock systems where grass accounts for the majority of livestock feed. Optimising grass utilisation is critical to the profitability of these production systems (Dillon et al., 2005), however this can be constrained by soils with impeded drainage and low soil fertility (Shalloo et al., 2004). Across Ireland, networks of sub-surface field drains and surface ditches are common in areas that experience high annual rainfall and poor natural drainage (Shore et al., 2015). Therefore drainage networks were designed and installed to quickly remove excess water from the land and/or lower the water table to optimise grass production (Tuohy et al., 2016). This has led to a complex drainage network with field drains connecting with an open ditch network or discharging directly to surface waters. Due to the ad-hoc nature of drainage systems in Ireland not all of the surface ditch network may be connected to a drainage outlet. The level of connectivity or dis-connectivity is important in terms of P loss along the transfer continuum (Daly et al., 2001; Nguyen and Sukias, 2002; Daly et al., 2017). Furthermore, drainage networks can have contrasting effects on hydrological connectivity depending on their location in the landscape (Lane et al., 2004; Shore et al., 2013). A number of studies have found significant concentrations of P in drainage ditch sediments, but whether a ditch is mobilising or retaining P depends on a number of local biogeochemical factors. For example, Nguyen and Sukias (2002) found high P concentrations and varying P retention capacities in surface and subsurface drainage sediments and concluded that P retention capacity in these sediments was correlated with a number of sediment chemical characteristics including pH, OM and oxalate-extractable Al and Fe. In an Irish study, Shore et al. (2015) classified drainage ditches using their physical characteristics and how these may affect the mobilisation and transport of fine sediment and associated P to surface waters downstream. The authors reported ditch dimensions, slope and vegetation cover to all influence the release or retention of fine sediment and any associated P. Sediment can act as a source or sink for nutrients and in fresh water systems it plays a crucial role in regulating the hydro-chemistry of rivers and streams. Previous studies have reported that sorption/desorption processes will adjust until the system reaches an equilibrium P concentration (EPCo) value (McDowell et al., 2001; McDowell, 2015; Hongthanat et al., 2016). If sediment EPCo is higher than the freshwater dissolved reactive phosphorus (DRP) concentration, the sediment will release P until it reaches that EPCo value, and conversely, if EPCo values are lower than DRP, the sediment will act as a sink for P until the freshwater DRP reduce to the equilibrium value (McDowell, 2015). In drainage ditches this interaction could be an important characteristic, often overlooked in the context of the P loss continuum.

In Ireland, agriculture has been identified as the primary source of P in surface waters with P transfer from agricultural land responsible for the degradation of 53% of rivers that fail to achieve ‘good’ ecological status under the European Union Water Framework Directive 200/60/EC (EU WFD). Currently, legislation regarding the protection of waterways from agricultural nutrients focuses on regulation at the farm gate and management of risk at the field scale by managing soil P status, P input levels and the timing of applications (S.I. No. 605 of 2017). These policies may not be the most effective strategy for water quality protection as they do not consider the connectivity vector i.e. the drainage ditches, as a potential option for P mitigation. The characterisation of surface drainage ditches and the overall drainage network offers an opportunity to use these unique landscape features to disconnect nutrient transfers from agricultural landscapes to surface waters, and to utilise them in risk assessment and P loss mitigation. Therefore, the objective of this study was to derive a farm-scale drainage ranking that categorises drainage ditches in terms of P loss risk based on connectivity and sediment characteristics. To meet this objective a network of pilot farms were selected to characterise these systems through ground survey and, sediment and water sampling.

2. Materials and methods

2.1. Case-study farms

Ten farms were selected as case-studies to represent a range of agronomic (intensive and extensive production systems) and biophysical settings across Ireland (Fig. 1). An agronomic summary of each farm in this study is presented in Table 1. The main farming enterprises were dairy and drystock with farms ranging in size from 29 to 255 ha. The intensity of farms, as indicated by their N stocking rate, ranged from 102 to 246 kg N/ha. The predominant soil types were a mix of mineral and organic soils dominated by poor to moderate drainage which necessitated artificial drainage installation on an ad-hoc basis over decades. Information on soil type was gathered from the existing EPA soils and subsoils maps (Fealy and Green, 2009). Field soil P data were available at field level for each farm to describe agronomic plant available P from



Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Fig. 1. Location of case study farms in the Republic of Ireland.

previous studies (Roberts et al., 2017; Shortle and Jordan, 2017). In many countries, fertiliser P recommendations are based on P index systems where bands of soil test P values are used to guide application rates. In Ireland, the agronomic soil test for P is Morgan's Extractable P and for easier management and knowledge transfer at farm level these values have been categorised as indices; 1 (deficient), 2 (low), 3 (agronomic optimum) and 4 (excessive) (Lalor and Coulter, 2008). Excesses and deficiencies in soil P levels are typically detected in detailed soil testing and in this system Index 4 identifies excessively fertilised fields that could also act as a source of P loss to water. All farms were located in areas with high annual rainfall (>847 mm).

2.2. Ground survey and feature digitisation

Ground surveys of the ten case study farms were carried out between August and December 2018 in order to characterise their surface and sub-surface drainage network. Qualitative interviews were carried out with each farmer to discuss the extent, type and location of any drainage work that may have been completed recently or historically. Following the interview, a ground survey covering the extent of the farm was carried out to identify and record the location of surface ditches, sub-surface field drains and drainage outlets. During the ground survey a number of features were noted including surface ditches/in-field pipe drains

Table 1
Summary of case study farms.

Farm number[1]	Location	Farm size (ha)	Enterprise	Stocking rate (kg N/ha)	% Fields with P index 4	Soil type	Soil OM% range	Annual rainfall (mm)[2]
1	Ballycanew	71.77	Dairy	183	0	Poorly drained surface water gleys	<20	890
2	Ballycanew	54.95	Dairy	246	5		<20	890
3	Dunleer		Dairy		38	Poorly drained luvisols	<20	767
4	Sreenty/ Corduff		Drystock		8	Acid brown earth/luvisols and gleys	<20	1027
6	Athea	60.56	Dairy	140	0	Surface water gleys and peat	10–26	1310
7	Black	33.93	Drystock	103	42	Brown earths and blanket peat	14–85	1191
8	Black	29	Drystock	102	45		11–88	1191
5	Allow	68.05	Dairy	194	31	Poorly drained surface water gleys/blanket	10–16	1154
9	Allow	65.44	Dairy	172	29	peat	10–18	1154
10	Allow	32.04	Dairy	243	22	and transitional soils in upland areas	11–26	1154

[1] Six of the case study farms had a derogation for N (Farms 1, 2, 3, 5, 9 and 10).
[2] Mean annual rainfall between 1985 and 2017.

connecting the farmyard to the drainage network, the proximity of the drainage network to nearby surface waters, any points of connection between the drainage network and local surface waters, groundwater springs and the general connectivity throughout the drainage network. Results of each farm survey were digitised using ArcGIS software to map the extent and characteristics of the drainage network and an example is presented in Fig. 2. The total length of the drainage ditch network on each farm was measured in ArcGIS and is presented in Table 2 as a proportion of the summed field perimeters of each farm. The length of each ditch category as a proportion of the total ditch length is also presented.

2.3. Grab sampling procedure

Grab samples of base sediment (n = 105) and surface water (n = 150) were taken from surface ditches throughout the drainage network. Surface ditch sediment samples were collected using a marked trowel to a depth of 5 cm (Shore et al., 2016). At time of survey on a number of farms ditches contained a large volume of water and/or were not accessible thus the collection of bed sediment was not feasible. This was the case on farms 7 and 8. In ditches where water was present a 50 ml sample each of unfiltered and filtered water was collected. For the filtered samples water was filtered through a 0.45 μm filter in the field. All water and sediment samples were stored and transported to the laboratory in cool boxes for analysis within 24 h of sample collection. Sediment and water sampling locations were digitised and are illustrated on Fig. 2.

2.3.1. Surface water analysis

Surface water samples were analysed to determine their dissolved, reactive and particulate P fractions. Filtered water samples were analysed calorimetrically for dissolved reactive phosphorus (DRP), using a nutrient analyser (Aquachem Labmedics Analytics, Thermo Clinical Labsystems, Finland). A second filtered subsample was analysed for total dissolved phosphorus (TDP) using acid persulphate. Unfiltered water samples were analysed for total phosphorus (TP) with an acid persulphate digestion and total reactive phosphorus (TRP) using the Aquachem Analyser. Particulate phosphorus (PP) was calculated by subtracting TDP from TP. All samples were tested in accordance with the Standard Methods (APHA, 2005).

Unfiltered water samples were used to determine NO_2-N, NH_4-N, total organic nitrogen (TON) and Cl concentrations using an Aquakem discrete analyser while concentrations of NO_3-N were calculated by subtraction of NO_2-N from TON.

2.3.2. Sediment analysis

Sediment samples were analysed for water soluble P (WSP) and Mehlich3-P (M3P) to assess their soluble and stored P fractions. Equilibrium P concentrations were estimated from isotherm analysis to describe the likelihood of sediment P release to overlying water. Phosphorus sorption isotherms were derived for sediment samples to derive P sorption capacity and binding energies using the Langmuir sorption model.

Prior to laboratory analysis, sediment samples were air-dried before being sieved to 2 mm. Mehlich3-P was determined on all base sediments using the modified Mehlich3-P test (Mehlich, 1984) to extract P, Al, Fe and Ca at a 1:10 sediment/solution ratio using Mehlich3 reagent (0.2 M CH_3COOH + 0.25 M NH_4NO_3 + 0.015 M NH_4F + 0.13 M HNO_3 + 0.001 M EDTA) with a 5-min reaction time and quantified by inductively coupled plasma (ICP-OES) spectroscopy (Sims and Heckendorn, 1991). Water soluble P was determined on a 1 g sample suspended in 40 ml distilled water and equilibrated for 1 h on a reciprocating shaker (Van der Paauw, 1971) before filtration using Whatman No. 4 filter paper and quantified colorimetrically. Equilibrium phosphorus concentration was determined using six base solutions which contained initial P concentrations of 0, 0.1, 0.25, 0.5, 1.0, 3 and 5 mg P L^{-1} (as KH_2PO_4). One gram of sediment was weighed into a 30 ml centrifuge tube and 20 ml of base solution added. Samples were then equilibrated for 24 h using a reciprocating shaker before centrifugation at 3600×g for 10 min. The supernatant was filtered through a 0.45 μm filter and the P concentration was quantified by colorimetry on a nutrient analyser (Aquakem Labmedics Analytics, ThermoClinical Lab Systems, Finland). Phosphorus isotherms were derived for all sediment samples using a modified version of the standard batch technique by (Nair et al., 1984). Sediment samples were equilibrated with P in solution and the data was evaluated using the Langmuir model (Pautler and Sims, 2000). Eight solutions of P concentration 0, 5, 10, 15, 20, 25, 35 and 50 mg/l^{-1} were added to 2 g sediment in 50 ml centrifuge tubes, in duplicate. The suspensions were shaken at room temperature for 24 h, centrifuged and filtered, and the concentration of P in solution was determined colorimetrically. Phosphorus sorbed to the sediment was calculated as the difference between the initial concentration and P concentration measured at equilibrium. Adsorption data and affinity constants were determined by fitting sorption data to the Langmuir equation using S (P sorbed) (mg P kg^{-1}) and C, the equilibrium P concentration (mg P l^{-1}). The Langmuir model is expressed as: $C/S = 1/Smax * k + C/Smax$. The linear form of this equation was used to derive Smax and k, the sorption maximum (mg P kg^{-1}) parameters and the constant relating to the binding energy (l mg^{-1}). A single point sorption index was derived using 1 g sediment shaken with 20 ml solution containing 75 mg P l^{-1}. Samples were shaken on an end-over-end shaker for 18 h and P concentration in solution was measured colorimetrically (John, 1970). A P sorption index (PSI l mg^{-1}) was expressed by the equation X/Log C where X is P sorbed (mg kg^{-1}) and C is the final P concentration at equilibrium (mg l^{-1}).

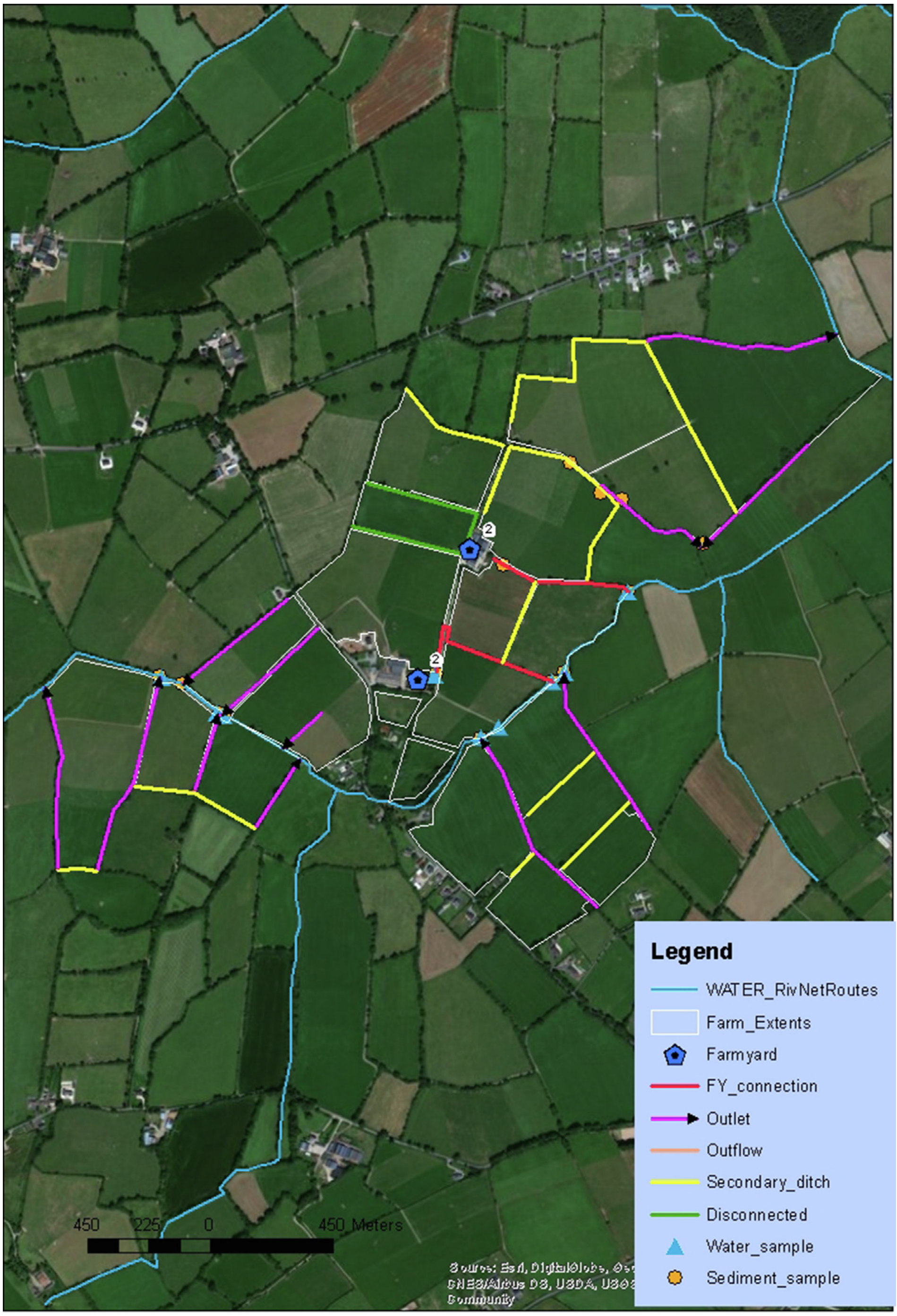


Fig. 2. An example of the maps produced after the ground survey on each survey farm indicating location of water and sediment sampling and characterisation of the surface ditch network. This figure describes Farm 2 of the study.

3. Results and discussion

3.1. Landscape features and setting

Drainage ditches accounted for an average of 27% of the total field boundary length, ranging from 6% on Farm 3 to 43% on Farm 2 (Table 2). Furthermore, the data in Table 2 suggests no clear relationship between the size of a farm (ha) and the extent of its ditch network. This is because drainage systems are generally ad-hoc and focus on drainage problems which have occurred over time. The size of these areas will differ between farms depending on local landscape and meteorological conditions. This was confirmed during the ground surveys where ditches occurred more frequently in low-lying areas likely prone to waterlogging as a result of low permeability soils or where the water table is shallow. The increased connectivity at this landscape position as a result of artificial drainage may increase the interaction with groundwater and thus diminish the potential for nutrient attenuation. On most farms the purpose of surface ditches is to alleviate waterlogging by removing excess water however if water quality and its protection is the goal then it is common practice to not drain such areas and indeed create saturated buffer zones to increase attenuation potential (Avery and Agency, 2012).

Table 2
Summary of ditch data including the proportion of the ditch network accounted for by each ditch category for each case-study farm.

Farm number	Field perimeter (m)	%perimeter that is ditch	Total ditch length (m)	Proportion of total ditch length (%)				
				1. Farmyard connection	2. Outlet	3. Outflow	4. Secondary	5. Disconnected
1	25,604	22	5725	13	0	13	74	0
2	30,581	43	13,242	9	48	0	36	7
3	36,835	6	2183	0	37	0	22	41
4	10,095	38	3789	6	0	60	34	0
5	27,458	14	3744	16	15	24	45	0
6	33,777	33	11,247	8	11	11	43	27
7	15,572	40	6218	0	24	10	66	0
8	16,736	31	5268	0	10	10	80	0
9	25,539	30	7729	9	17	7	68	0
10	15,981	14	2160	29	20	0	11	40

Five drainage ditch categories were derived from the ground survey that represent the physical characteristics and landscape setting of surface ditches across the case-study farms. These are listed in Table 3. Shore et al. (2015) highlighted the influence of physical characteristics on a ditch's capacity to retain or transfer fine sediment, surface and subsurface run-off, and associated P. Furthermore, surface ditch location can indicate where its load is being transported to, the pathway at play and, whether it transports rainwater only or does the ditch also interact with groundwater or springs. The ditch classification system introduced in the current study accounts for interactions between landscape setting and physical ditch characteristics and indicates the role that particular ditches within a network play in connecting pathways and agricultural nutrient sources to local surface waters.

The most obvious example of drainage ditches connecting agricultural nutrient sources to surface waters is seen in the Farmyard Connection category (Category 1). This is an easily identifiable category in which a ditch connects a farmyard either to the main drainage ditch network or directly to a body of water. Farmyard connections will typically occur in the form of a surface ditch or subsurface pipe drain that collects runoff from the farmyard, a slurry/runoff storage tank or a silage clamp. Although they only represent an average of 13% of the ditch network on farms where present (Table 2), Farmyard Connections by their nature, have potential to deliver high nutrient loads to surface waters and represent a significant nutrient point source as outlined later in this section.

Surface ditches that fall into the Outlet (Category 2) or Outflow (Category 3) categories are broadly similar in appearance however can be defined by their position in the landscape: an Outlet flows directly into a water body while an Outflow transports drainage water across the farm boundary onto neighbouring land. These ditches typically occur in low-lying parts of the landscape and are the confluence of a number of smaller Secondary ditches (Category 4). They therefore have potential to transport water drained from a large area of land. In this study Outlet and Outflow ditches were present on eight and seven of the ten case study farms, representing 23 and 19%, respectively of the total ditch network of these farms (Table 2).

Table 3
Definition and description of ditch categories.

Ditch category	Description
1. Farmyard connection	A ditch/pipe that connects a farmyard to the drainage network or directly to a surface water body
2. Outlet	A ditch that connects the drainage network to a surface water body
3. Outflow	A ditch that carries drainage water across the farm boundary through neighbouring land
4. Secondary	A ditch that typically flows perpendicular to the slope of the land connecting two larger ditches. Can also occur as an open ditch running through a field in order to collect and remove large excesses of surface water
5. Disconnected	A ditch that is not connected to the overall ditch network

Secondary ditches were the most commonly recorded category across the farm surveys being present on all ten case study farms representing an average of 48% of the ditch network of each farm. It is logical that Secondary ditches are more numerous than any of the other ditch categories in this study as their primary function is to collect excess surface and subsurface water from poorly drained areas of land and transport it to larger Outlet or Outflow ditches. Due to their widespread abundance Secondary ditches are key features in determining the extent of connectivity of the drainage network and play a significant role in linking agricultural land to surface waters.

Disconnected ditches (Category 5) are those that are not connected to the overall surface ditch network of a farm and can be disconnected by design, and therefore typically transfer surface water to subsurface groundwater, or may have become disconnected as a result of blockage due to lack of maintenance. This ditch category occurred on only four of the ten case study farms as ditches are generally maintained to some extent in order to remove excess water from poorly drained areas.

3.2. Hydro-chemistry

Hydro-chemical parameters measured in water samples from ditches and streams are prone to regular fluctuation over time as a result of fluctuating local conditions and thus are ideally measured at regular intervals over a prolonged sampling period. The results obtained from a single sampling event in the current study however provide a snapshot of ditch hydro-chemistry at the time of survey. The summary data for DRP, TRP, PP, TDP and TP presented in Table 4 shows breaches of the environmental quality standards (EQS) set by the EU WFD across each of the five ditch categories and demonstrates the role of drainage ditches as a connectivity vector between agricultural P and nearby surface waters.

Total phosphorus is an important water quality parameter as it quantifies the reactive and unreactive species in a sample. Values in the current study range from 0.007 to 325 mg l^{-1} with a mean of 7.502 mg l^{-1} recorded across all surveyed ditches (Table 4). This maximum value, and the next highest value of 8.120 mg l^{-1}, were recorded in Farmyard Connection ditches and reflect the high nutrient inputs from farmyard runoff (Fenton et al., 2011). The mean TP value recorded in Farmyard Connections in the current study exceeds previously reported values (1.5 mg l^{-1}) from similar agricultural ditches reported by Harrison et al. (2019). Furthermore, Harrison et al. (2019) reported greater TP concentrations in drains connected to farmyards than the remaining drainage ditches sampled, which aligns with data reported for Farmyard

Table 4
Summary statistics of phosphorus and nitrogen concentrations and speciation in water samples and soil P properties in sediment samples taken from ditches.

Ditch category		Water							Sediment				
		DRP[1]	TRP[2]	PP[3]	TDP[4]	TP[5]	Nitrate	Ammonium	WSP[6]	M3P[7]	EPCo[8]	Smax[9]	k value[10]
		mg l^{-1}							mg kg^{-1}		mg l^{-1}	mg kg^{-1}	l mg^{-1}
1	Min	0.018	0.017	0	0	0.022	0.481	0	1.16	15.51	0.048	400	0.059
	Max	342	348	6	342	325	380	379	175.04	429.50	2.19	1666.67	1.33
	Mean	42.8	43.5	0.8	58.2	55.6	65.6	63.2	24.92	159.31	0.480	783	0.836
	Median	0.050	0.046	0	0	0.084	1.580	0.051	6.52	115.83	0.275	769	0.828
2	Min	0.013	0.008	0	0	0.025	0.008	0	0.00	5.14	0.007	345	0.192
	Max	0.407	0.449	0.232	2.350	0.429	9.351	0.264	45.56	294.18	27.470	833	2.667
	Mean	0.111	0.118	0.020	0.213	0.197	2.416	0.026	7.64	57.05	2.370	578	0.804
	Median	0.031	0.041	0.003	0.042	0.242	1.125	0.003	3.18	43.77	0.130	588	0.724
3	Min	0.002	0.002	0	0	0.014	0	0	0	4.23	0.013	370	0.433
	Max	0.233	0.280	0.250	0.244	0.345	5.393	2.433	6.96	54.62	0.338	1000	3.333
	Mean	0.033	0.066	0.030	0.072	0.175	2.166	0.655	2.43	27.15	0.095	542	1.205
	Median	0.019	0.021	0.001	0.052	0.175	0.902	0.068	1.52	26.50	0.054	526	1.063
4	Min	0.002	0.001	0	0	0.007	0	0	0.00	3.58	0.006	323	0.277
	Max	1.847	1.852	0.028	1.050	2.060	6.877	3.298	22.32	114.70	0.418	909	3.800
	Mean	0.305	0.305	0.005	0.197	0.478	1.844	0.301	8.03	45.80	0.139	569	1.134
	Median	0.060	0.056	0.002	0.070	0.163	0.309	0.036	8.28	38.83	0.078	526	0.571
5	Min	0.014	0.016	0	0	0.035	1.417	0	1.08	40.92	0.055	417	0.329
	Max	1.679	1.707	0.028	0.024	0.164	3.332	0	15.92	103.10	1.114	714	2.800
	Mean	0.451	0.459	0.008	0.022	0.100	2.375	0	8.07	65.92	0.402	573	0.911
	Median	0.056	0.056	0.001	0.022	0.100	2.375	0	7.92	61.68	0.300	588	0.538
All categories	Min	0.002	0.001	0	0.008	0.007	0	0	0.00	3.58	0.01	322.58	0.06
	Max	342	348	6	342	325	380	379	175	430	27	1667	4
	Mean	5.189	5.288	0.104	7.738	7.502	10.425	8.463	9.17	64.13	0.94	598.41	0.98
	Median	0.030	0.042	0.002	0.053	0.164	1.117	0.005	3.82	41.91	0.14	571.90	0.75

Connections in this study. This is supported by simultaneously high ammonium concentrations at several locations in the current study. Other relatively high values recorded include TP values of 2.060, 1.250 and 1.210 mg l^{-1} recorded in Secondary ditches on Farms 7 and 8. Although these ditches are typically not connected to any point sources such as farmyards, they are surrounded by fields with elevated P index values (Table 1) and therefore likely to be a diffuse P source to water. Furthermore, the soil type on these farms is characterised as 'Peat' as evidenced by high % OM recorded from these fields. This soil type has a poor capacity to retain P applied as manure or inorganic fertiliser (Daly et al., 2001; Jiménez et al., 2019a,b), increasing the risk of diffuse P losses to water. The combination of soil types with poor P retention and high P index values on Farms 7 and 8 is likely to contribute to the high TP values recorded in a number of their drainage ditches illustrating the importance of biophysical and landscape setting in determining risk. Total reactive phosphorus and DRP values ranged from 0.001 to 348 mg l^{-1} and 0.002 to 342 mg l^{-1}, respectively, in the current study. The median DRP value of 0.030 mg l^{-1} is intermediate of the median values of 0.019 and 0.039 mg l^{-1} reported by Daly et al. (2017) and is lower than the EQS of 0.035 mg l^{-1} set by the EU WFD. With the exception of the Farmyard Connections, which are point sources, higher TRP and DRP values were primarily associated with Secondary and Disconnected ditches where the likely source of P is diffuse and derived from nutrient runoff from applied manure and/or inorganic fertilisers. Secondary ditches in particular occurred most frequently during the farm surveys (Table 2) making them quite efficient at collecting large volumes of overland flow from surrounding fields. Depending on weather conditions and the timing of the spreading of organic manure and/or inorganic fertiliser, this runoff can transport a large amount of P from the land to drainage ditches (Vadas et al., 2008, 2011).

Several locations across the study sites had elevated NO_3-N and NH_4-N concentrations above the thresholds set out by OECD (2001) (Table 4). The presence of these nutrients is indicative of connectivity between the sampling location and an organic N source from point or diffuse sources. This connection has been found elsewhere across multiple sampling locations (e.g. groundwater, end-of-pipe and drainage ditches) on dairy farms with heavy textured soils (Fenton et al., 2009; Necpalova et al., 2010; Baily et al., 2011; Clagnan et al., 2018). As nitrate converts to ammonium on these sites high nitrate concentrations may indicate a close proximity to the source or indicate more well drained subsurface conditions whereas high ammonium concentrations may indicate the occurrence of raw organic N (Huebsch et al., 2013).

3.3. Sediment phosphorus characteristics

The physic-chemically bound fraction of P in ditch sediment was represented by Mehlich3 extracts and values ranged from 3.58 to 429.50 mg kg^{-1} with a mean of 64.13 mg kg^{-1} recorded across all surveyed ditches (Table 4). Mehlich3-P is typically used as an agronomic test for field soils and a value of 50 mg kg^{-1} represents the lower band of the agronomic optimum range, whilst values >100 mg kg^{-1} are often an indication of high soil P reserves or legacy P, built up over time due to decades of excessive P applications (Vadas et al., 2018; Haygarth et al., 2014). In this study, a wider range of extractable M3P values was captured in ditch sediment compared to previous Irish studies (Daly et al., 2017; Shore et al., 2016) with a higher proportion of M3P values >100 mg kg^{-1} recorded in Farmyard Connections and Outlet ditches. Box plots in Fig. 3 illustrate the range across categories with maximum values of 429.5 and 293.8 mg kg^{-1} recorded in sediments sampled from ditches connected to farm yards and drainage network outlets. This indicates P deposition from point source inputs from farm yards and accumulation of P loads at an outlet point draining nutrients from the landscape along the network. Haggard and Stoner (2009) reported M3P values ranging from 44.3 to 250 mg kg^{-1} in benthic stream sediment downstream of rural effluent discharge and confirmed that sediment was still acting as a source of P, long after the point source had been removed. Elevated concentrations

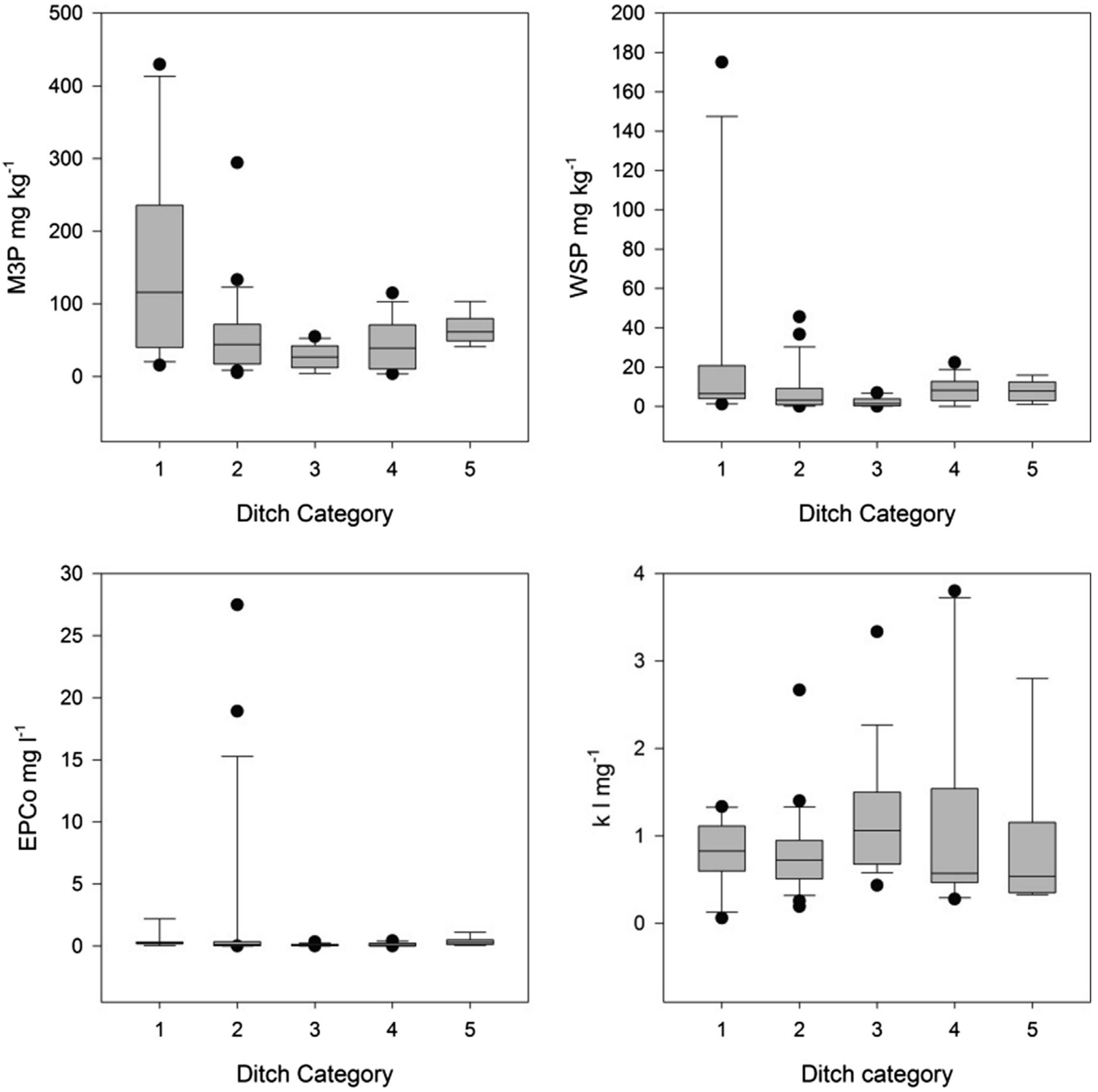


Fig. 3. Box plots showing median, 10th, 25th 75th and 90th percentiles as vertical boxes with error bars for Mehlich3 extractable P (M3P), water soluble P (WSP), equilibrium P concentrations (EPCo) and P binding energy (k) values measured in sediment sampled from each ditch category. Missing EPCo values in ditch category 1 were associated with samples with excessive M3P values that did not fit the isotherm and saturated the solution at equilibrium.

of M3P and WSP in Farmyard Connections and Outlet ditches, in this study, indicate legacy P stores that have potentially built up over long periods of P deposition from point and diffuse sources on the farms surveyed. Mobilisation of these legacy stores as soluble P from sediment to water is represented by WSP extracts from the ditch sediment and values varied from 0 to 173.5 mg l^{-1} with highest values recorded in Farmyard Connections and Outlet ditches, analogous with M3P values (Fig. 3). The range of extractable P (M3P and WSP) captured across the network of ditches surveyed in this study and illustrated in Fig. 3 supports our assumption that P entering ditches can accumulate in sediment, particularly when ditches are connected to point sources and diffuse P delivery points in the landscape.

The source/sink dynamic at the sediment-water interface was expressed as EPCo. This parameter represents a solution P concentration, at which, processes of sorption and desorption are in equilibrium and values were measured across the ditches surveyed in this study to assess the likelihood of P release or retention into water draining the landscape. Sediment samples (n = 3) with excessively high M3P collected from Farmyard Connections failed to provide a good fit to the isotherm and a value for EPCo could not be derived. The procedure for estimating EPCo is derived from the linear portion of the isotherm within a low concentration range, and it is likely that excessively high P samples saturated the solution thus confounding the model. Equillibrium phosphorus concentrations ranged from 0.006 to 27.47 mg l^{-1} with values <0.300 mg l^{-1} in the 75th percentile with some extreme values recorded outside the range (Fig. 3). Mean and median EPCo values of 0.940 and 0.140 mg l^{-1}, respectively, were recorded across all ditch categories but highest values were observed in Outlet sediments.

The influence of legacy P on EPCo values in ditch sediment is illustrated in Fig. 4, and supported by the linear regression between EPCo values and M3P ($R^2 = 0.6138$) and WSP ($R^2 = 0.7459$). These results indicate that P accumulated in ditch sediment due to inputs from yards and points on the landscape draining diffuse sources resulting in high EPCo values. These results are supported by previous studies that have identified external factors such as P inputs, high flow regimes and redox conditions elevating EPCo and sediment P release to water (House and

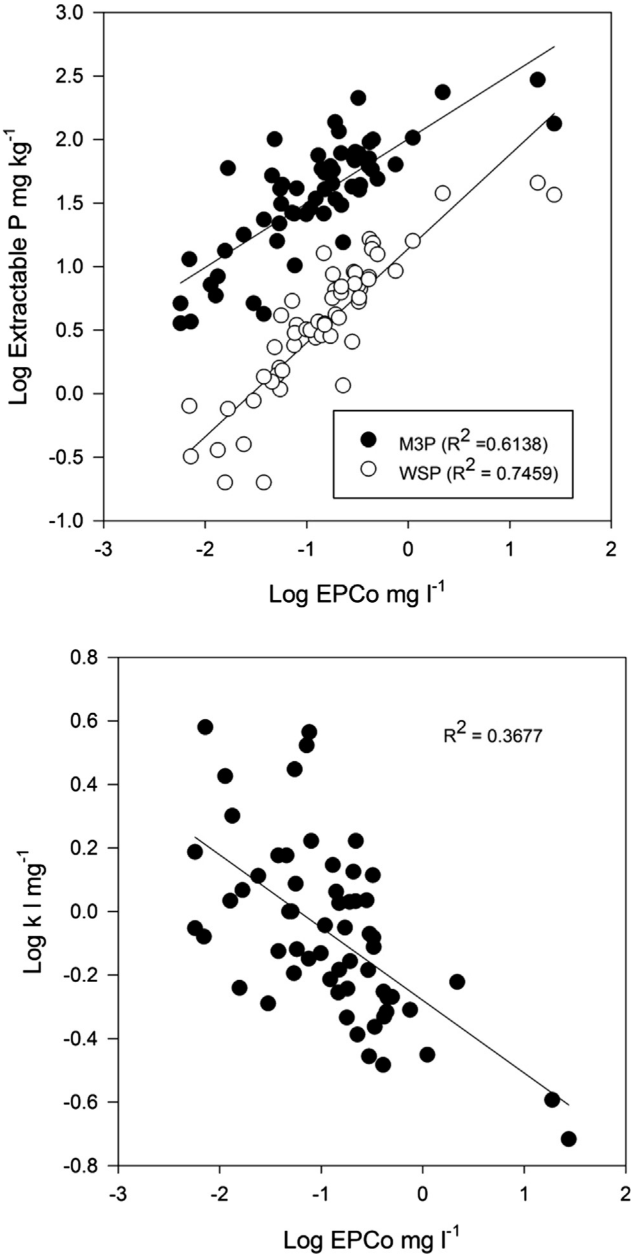


Fig. 4. Scatter plots with linear regression lines showing the relationships between equilibrium P concentration (EPCo) and extractable P measured as Mehlich3 (M3P), water soluble P (WSP) (Top) and between Langmuir P binding energy (k) for sediment samples collected during the survey.

Denison, 2000; Jarvie et al., 2005; Haggard and Stoner, 2009). Reducing conditions and high P saturation has also been linked with P binding energies (k) derived from the Langmuir sorption isotherm. House and Denison (2000) reported lower k values in sediment under reducing conditions which supports our results here. Phosphorus binding energies derived from ditch sediment were plotted against EPCo values and the negative correlation is illustrated in Fig. 4. As k values decreased, sediment EPCo values increased (r = −0.7000) with k values accounting for 36% of the variation in EPCo. High k values in this study were associated with Outflow and Secondary ditches, and lowest values were found in sediment sampled from Farmyard Connections. Overall, k ranged from 0.059 to 3.8 mg l^{-1}, lower than values reported in the literature for agricultural soils under aerobic conditions (Indiati et al., 1999; Pautler and Sims, 2000; Daly et al., 2015).

3.4. Landscape and physic-chemical interactions: Deriving a connectivity risk ranking

The results of this study demonstrate that surface ditches can act as both a conduit and a source of nutrients – particularly P. Connecting nutrients from point and diffuse sources via narrow channels and surface ditches can accelerate delivery to receiving water bodies thus presenting a risk of nutrient losses to surface waters from agricultural landscapes.

Ground surveys carried out during this study reported a variety of ditches that were defined by their landscape position and used to derive the categories described in Table 3. These five drainage ditch categories provide a useful tool for describing the magnitude of connectivity, and indeed dis-connectivity, in transferring point and diffuse sources of P to nearby surface waters based on landscape position. In addition, the ditch sediment studies indicate that sediment can act as a source of P, largely due to deposition and accumulation of P which can be released into overlying water and flushed along the network long after the original pollution event. This phenomenon has previously been highlighted by Sharpley et al. (2013) where it is discussed in detail. Therefore, risk of P loss from ditches can be defined by both landscape position and sediment P dynamics. Based on the ground survey and chemical analysis reported here, ditches are ranked from 1 to 5 in order of risk from highest to lowest.

Farmyard Connections represent the greatest risk to water quality as indicated by their landscape position and physic-chemical characteristics. The magnitude of risk in this ditch category is primarily associated with the direct connection between a point source and surface water body. In addition, the high risk status is confirmed by the high sediment P concentrations reported in this study and in similar surface ditches elsewhere (Harrison et al., 2019). Landscape factors such as ditch length and slope may influence nutrient attenuation rates in these ditches (Shore et al., 2015, 2016) but will likely not be sufficient to offset the significant amount of nutrients originating from the point source. Furthermore, the low binding energy (k) and high available P concentrations recorded in Farmyard Connection sediments suggest a legacy issue meaning that even after the connection between the source and surface water is broken, P already accumulated in the sediment will continue to be stored and released into the drainage water. The extent of this P storage has not been quantified and could extend deep into the sediment layer with McDowell et al. (2002) suggesting that such accumulated P can continue to act as a source to surface waters for many decades.

Landscape position is responsible for a significant portion of Outlet ditch risk as these ditches are typically located at the bottom of a slope where they are in contact with a number of nutrient pathways from both point and diffuse sources and often coincide with critical source areas. Furthermore, the nutrient status of the catchment area drained and contributing Secondary ditches, as well as the connectivity between both ditch types, will all significantly influence the ultimate risk associated with Outlet ditches. There is also evidence that the legacy P issues described for Farmyard Connections can occur in Outlet ditches with Outlet sediments in this study showing signs of P build-up as slope and water velocity decrease leading to P deposition (Sharpley et al., 2013). The same landscape by physic-chemical interactions associated with Outlet ditches apply to Outflow ditches. However, an additional factor associated with Outflow ditches is the distance drainage water must travel to reach a surface water after it has left the original farm thus, a greater distance may lead to greater opportunity for nutrient attenuation. The nature of Outflow ditches means the drainage water – and associated nutrients – originates from at least two different farms. This fact may lead to complications if ditch remediation measures are required. For example;

which landowner incurs the cost of any measures implemented? Can the cost be divided according to the nutrient contribution of each farm? These are among the factors that will need to be considered by policy makers regarding the implementation of P loss mitigation measures in drainage networks.

Although sediment and water quality data from this study indicate elevated P concentrations in Secondary (Category 4) ditches landscape/connectivity to receiving waters will likely have a greater influence on their risk than chemistry. Thus, whether Secondary ditches are sufficiently connected to larger Outlet or Outflow ditches will determine the water quality risk posed by this category of ditch. Disconnected ditches (Category 5) are an extreme example of this principal where despite often having elevated nutrient concentrations, there is no connection to the greater drainage network and thus pose the lowest risk to surface water quality based on connectivity.

The potentially significant lag time associated with the release of stored P described above will need to be considered by policy makers when implementing and assessing P mitigation measures in surface ditches. Such lag times in nutrient release will result in similar delays between the implementation of measures to remove/reduce a P source and the onset of measurable improvements in surface water quality. Therefore, unless adequate time is given before the assessment of mitigation measures to allow for the dissipation of stored legacy P it will likely be concluded (wrongly) that the measure has not been effective.

4. Conclusions and recommendations

Surface ditches are both a source and pathway for nutrients such as N and P from agriculture to surface waters as confirmed by the elevated P concentrations, connectivity and sediment characteristics reported in this study. The landscape position and level of connectivity of a surface ditch plays a major role in determining the magnitude of the P loss risk associated with a particular ditch. Surface ditches that provide a direct connection between farmyards, a point source, and surface waters pose the greatest risk while ditches associated with diffuse sources and/or have low connectivity to the drainage network (Outlets/Outflows, Secondary and Disconnected ditches) pose incrementally lower P loss risks. Furthermore, surface ditches can act as a P source as indicated by the results of this study presenting a legacy P issue where accumulated P can be released from sediment potentially after the source pathway has been broken. This is especially the case in Farmyard Connection ditches where extractable P concentrations in sediment were found to be high, and retention low.

Surface ditches are unique features in the landscape and this study demonstrates their important role in water quality. Identifying ditches that pose the greatest risk to water quality on a particular farm using the ranking system outlined in this study offers an opportunity for the more targeted implementation of mitigation measures. Furthermore, surface ditches allow for the implementation of these measures without incurring the costs associated with taking agricultural land out of production. Ideally, nutrient loss to surface waters should be mitigated by balancing P across the farm through careful nutrient management planning, however this is a long term option and can incur significant financial costs in the initial phases (Bragina et al., 2019). Farmers and landowners are likely to be more accepting of measures that incur less cost and have a more immediate visible effect (Micha et al., 2018) such as those commonly associated with drainage ditch remediation (McDowell and Nash, 2012; King et al., 2015). In future, ground surveys that map surface ditches may be more efficient than locating individual in-field drains as surface drains are conduits for the drained water from many in-field drains and are the main connections of these in-field drains to surface waters. Furthermore, risk assessment and P loss models of the future should include metrics of connectivity via surface ditches to help target the appropriate measures to the appropriate locations.

Measures mitigating P loss often involve the breaking of the nutrient pathway (Deakin et al., 2016) however results from this study indicate that legacy P built up in sediment will likely keep contributing to losses.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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