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Variable response to phosphorus mitigation measures across the nutrient transfer continuum in a dairy grassland catchment

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ABSTRACT

Phosphorus (P) loss from soils to water can be a major pressure on freshwater quality and dairy farming, with higher animal stocking rates, may lead to potentially greater nutrient source pressures. In many countries with intensive agriculture, regulation of P management aims to minimise these losses. This study examined the P transfer continuum, from source to impact, in a dairy-dominated, highly stocked, grassland catchment with free-draining soils over three years. The aim was to measure the effects of P source management and regulation on P transfer across the nutrient transfer continuum and subsequent water quality and agro-economic impacts. Reduced P source pressure was indicated by: (a) lower average farm-gate P balances (2.4 kg ha^{-1} yr^{-1}), higher P use efficiencies (89%) and lower inorganic fertilizer P use (5.2 kg ha^{-1} yr^{-1}) relative to previous studies; (b) almost no recorded P application during the winter closed period, when applications were prohibited, to avoid incidental transfers; and (c) decreased proportions of soils with excessive P concentrations (32–24%). Concurrently, production and profitability remained comparable with the top 10% of dairy farmers nationally with milk outputs of 14,585 l ha^{-1}, and gross margins of € 3130 ha^{-1}. Whilst there was some indication of a response in P delivery in surface water with declines in quick flow and interflow pathway P concentrations during the winter closed period for P application, delayed baseflows in the wetter third year resulted in elevated P concentrations for long durations and there were no clear trends of improving stream biological quality. This suggests a variable response to policy measures between P source pressure and delivery/impact where the strength of any observable trend is greater closer to the source end of the nutrient transfer continuum and a time lag occurs at the other end. Policy monitoring and assessment efforts will need to be cognisant of this.

1. Introduction

The driver of food needs and the pressure (DPSIR framework; EEA, 1999) of intensive agriculture may impact water quality, particularly via nutrient loss from land to water, causing eutrophication (Sharpley and Rekolainen, 1997). The EU policy response is the nitrates directive (ND) (OJEC, 1991) and consequent Nitrates Action Programmes (NAPs) to manage the risks of such losses. In the Republic of Ireland, the NAP (SI 31, 2014) includes measures for both N and P, as P is an important trophic pressure in freshwaters (EPA, 2012). Reviews of the NAP follow a four-year cycle and are contingent, amongst other things, on water quality status and the agricultural contribution to the pressure.

Dairy farming is a key sector of Irish agriculture and operates a relatively low cost, efficient and profitable system based on maximizing grazed grass in the cow's diet (Ryan et al., 2011). However, dairy farms also tend to be more intensive with higher stocking rates than other grass-based systems, supported by higher P applications to grassland and higher farm P imports in

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feed and fertilizer, potentially creating a greater nutrient source pressure.

Under the NAP, P inputs and management are constrained by measures intended to improve P use efficiency (PUE) and minimise P loss risks (Table 1), such as a winter closed period for spreading fertilisers and maximum field-level P application rates based on crop type and soil P concentration (Morgan P) (Wall et al., 2013). Application of P to soils with excessive P is prohibited in most scenarios, the intention being that, with continued P offtake by crops, concentrations will decline to optimum and reduce the environmental risk. Previous research to gauge the effectiveness of NAP measures has focused on the time required to decline to optimum (e.g., 2–20 years in Wall et al., 2013), including modeling (Schulte et al., 2010) and plot scale monitoring (e.g., Dodd et al., 2012; Blake et al., 2003; Burkitt et al., 2002). However, few studies have examined soil P changes over time in whole farm systems and less so on a catchment or watershed scale – the scale of the farming landscape as it interacts with hydrological processes. Wall et al. (2011) proposed that monitoring of policy impacts be cognisant of the nutrient transfer continuum (Lemunyon and Gilbert, 1993; Haygarth et al., 2005), or nutrient cascade (Smith et al., 2013), from source to impact, including nutrient changes and attenuation along pathways and their influence on ecological impacts.

Ultimately, optimising agronomic output and lowering environmental risk is the policy goal (Buckley and Carney, 2013) but discrete monitoring programs report on national scale pressures and states separately and few consider the links between these two. Biophysical processes, such as variable climatic processes (Mellander et al., 2014) and non-agricultural pressures (Withers et al., 2013), can lead to lag effects between changes in nutrient source pressure and water quality impact (Sharpley et al., 2013). Positive river water quality responses to agricultural practice change in meso-catchments (1–100 km^2) may commonly take up to 10 years to occur, and even longer to measure (Melland et al., 2014). The time-scale of affecting change may not, therefore, match the expectations of policy makers and this may have consequences for DPSIR reviews of policy effectiveness. Furthermore, the water framework directive (WFD; OJEC, 2000) makes explicit comment on the economic consequences of such policy measures (Martin-Ortega, 2012), which, at the farm scale, is the impact on productivity and profitability. These considerations are increasingly embedded in catchment management reviews (e.g., Kragt et al., 2011; Roberts et al., 2012).

The advantage of examining the P cascade in catchments is that the integrated effect of farm management practices and environmental processes, that both generate and attenuate P loss in the landscape, are captured (Sharpley et al., 2013). This study examined farm-level P balances and use efficiencies, field-level P inputs, management and soil P status, P pathways and losses to water, stream biological quality and farm agro-economics in a dairy-dominated catchment with free-draining soils over three years. The P cascade elements were measured to gauge the effects of the NAP measures on P source pressure and subsequent water quality and agro-economic impacts.

2. Materials and methods

2.1. Study area

This study took place in south-west Ireland, in a 7.6 km^2 catchment (Fig. 1) with mostly well-drained Brown Earth soils (Cambisols) (85%) with lesser areas of more poorly drained surface-water gleys and gleyic alluvial soils (stagnosols) and peats (histosols). Bedrock geology consists mostly of Devonian (old red sandstone) interbedded sandstones, mudstones and siltstones and upper Devonian–Carboniferous sandstones and mudstones (Sleeman and Pracht, 1994). The climate is cool temperate oceanic with a mean annual rainfall of 1207 mm (Table 1). The catchment area is ca. 90% agricultural, dominated by grassland (79% in 2010 and 70% in 2011) used for dairy production in a system based on grazed grass that is characteristic of the more intensive dairy production systems in southern Ireland. The remainder is in arable crops; principally maize, spring barley and spring wheat. There are 42 land owners in the catchment with 15–20 accounting for most of the agricultural area. Twelve farms (34% of the catchment area) have stocking rate derogation. The catchment has an overall

Table 1
Characteristics of the study catchment, including nutrient loss mitigation measures under the NAP (AF = all farms, DF = derogation farms, NDF = non derogation farms).

Physical			
Climate	Cool temperate oceanic	Elevation (m AOL)	17–127
Topography	Rolling to flat	Mean annual temperature[a] (C)	9.4
Mean annual rainfall[a] (mm)	1207	Mean winter (Nov–Jan) rainfall[a] (mm)	394
Mean summer (May–Jul) rainfall[a] (mm)	217		
Mean annual catchment rainfall 2010–2013 (mm)	1144	Mean annual stream runoff 2010–13 (mm)	563
Mean annual runoff coefficient 2010–2013	0.48		
Dominant soil type	Well-drained brown earth (Brunisol)	Bedrock geology	Devonian sandstones and mudstones
Unconsolidated sediments	Fluvial and glacial deposits	Stream order	3
Land use			
Total area (km^2)	7.6	Grassland area (%)	75
Agricultural area (%)	90	Arable area (%)	15
Livestock density (LU ha^{-1})	1.94	Other area (%)	10
Mitigation measures			
Organic N limit 170 kg ha^{-1}	NDF	Organic N limit 250 kg ha^{-1}	DF
Organic fertiliser closed period (15 Oct –12 Jan)	AF	Inorganic fertiliser closed period (15 Sep–12 Jan)	AF
16 weeks manure storage facilities	AF	Max field N and P rates	AF
Ploughing restrictions (1 July –15 Jan) and green cover requirements	AF	Limits on farm P import	AF
Soil sampling and analysis at least every 4 years	DF	Nutrient management plan	DF
No fertilizer close to streams, rivers, lakes or groundwater sources	AF	No fertilizer on waterlogged, frozen or steep ground or when heavy rain is forecast	AF
Soiled water storage facilities for 10 days	AF	Good yard facilities and management	AF

[a] 30 year average (1962–1991) from nearest synoptic station (Cork Airport).

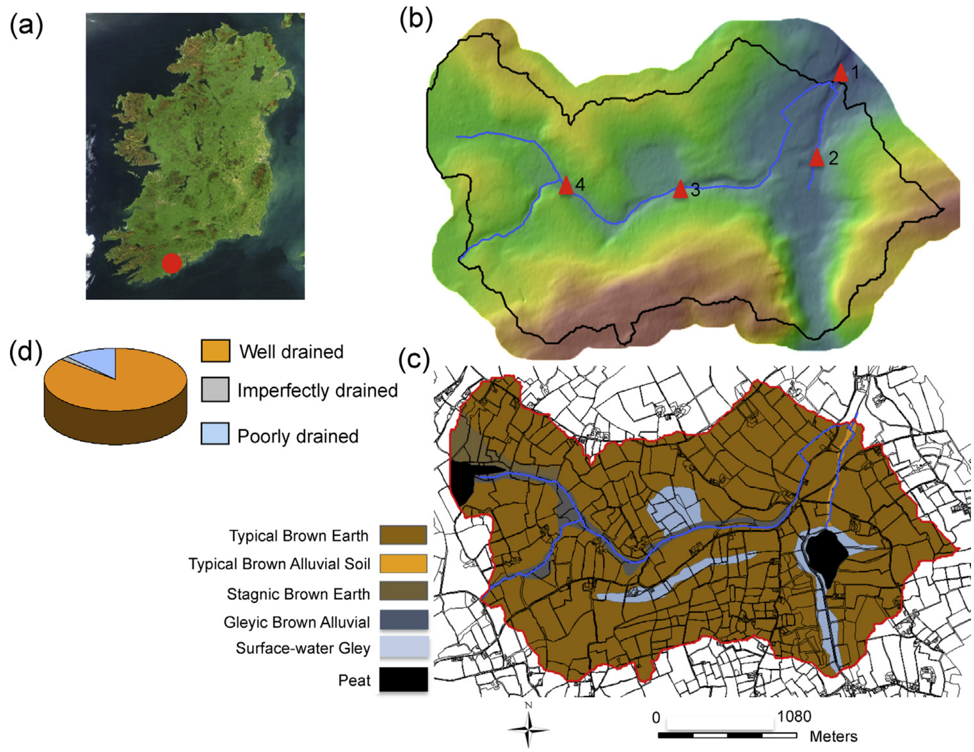

Fig. 1. Location (a), topography and stream network (b), soils and field boundaries (c) and soil drainage distribution (d) of the study catchment. Red triangles in (b) indicate stream biological sampling sites at 1. outlet, 2. tributary, 3. main stream, and 4. headwater. Satellite image source: NASA Earth Observatory.

stocking rate, based on national inventories, of 1.94 LU ha^{-1} (165 kg organic N ha^{-1}).

2.2. Phosphorus sources

Farm-gate P imports (inorganic fertilizer, feed, manures, livestock) and exports (milk, livestock, crops sold, manures exported) were recorded (Wall et al., 2012) for 11 catchment farms (9 dairy, 1 suckler beef, 1 tillage/sheep/beef), for 2010–2011, accounting for 586 ha; equivalent to 87% of the catchment agricultural area. Some areas of these farms were outside the catchment but it was assumed that these farms were representative of the farmed catchment area. Of the 9 dairy farms, 6 had derogation from the NAP organic N input limit of 170 kg organic N ha^{-1} (in Ireland, equivalent to a stocking rate of 2 livestock units (LU) ha^{-1}). Farm data were calculated on a whole-farm area basis, unless stated otherwise. The farm-gate P balance was calculated as the balance of farm-gate imports minus farm-gate exports in a year. Phosphorus use efficiency was calculated as the farm-gate P exports expressed as a percentage of the P imports. To calculate farm and field P balance components, assumptions of standard P content (Coulter and Lalor, 2008; DEFRA, 2010; Sauvant et al., 2004) of different feed types, animals, manures and crops and animal liveweight based on type and age were made, where necessary.

Field nutrient data, including application timing, rate and form of P, were recorded, for fields within the catchment boundary, by the catchment farmers and their advisor in nutrient management records (NMRs) (Wall et al., 2012) for 2010–2011; this equated to 97 and 88% of the agricultural area, respectively. Farm and field nutrient data were thus largely synchronous with the other data collected but did not cover 2012. However, these two years of data, covering most of the catchment area, establish a good representation of farm nutrient management in the catchment. Soil P status was assessed for every field in the catchment with sample areas no greater than 2 ha in late 2009–2010 (mostly in early 2010) (Wall et al., 2013) and again in early 2013 using identical sampling areas. Soil was sampled to 10 cm depth using soil corers with at least 20 cores per sample area, taken in a 'W' pattern across the sample area. These 20 cores were bulked, dried at 40 °C, passed through a 2 mm sieve and analysed for Morgan P (Morgan, 1941). The total area covered by both sampling events was 648 ha, or 96% of the agricultural area.

2.3. Phosphorus pathways and delivery

High temporal-resolution water discharge and P concentration data were collected at the catchment outlet (Jordan et al., 2012) from 2010–11 to 2012–13 (three April-March hydrological years). Discharge was calculated from water level, rated at a flat-V Corbett non-standard weir. Phosphorus concentration, both total P (TP) and total reactive P (TRP), was measured using a Hach-Lange Phosphax-Sigmatax bankside analyser (Mellander et al., 2012). Total P loads for the three monitored years were collated with data completeness of 94%. Low, median and high flow ranges (Q90, Q80, Q50 and Q10, plus and minus 5 percentile ranges), and corresponding P concentrations, were also determined. Using the Loadograph Recession Analysis method of Mellander et al. (2012), hydrological and P transfer pathways (*Quick flow*,*Interflow* and *Delayed flow*) were quantified over the regulated closed periods for organic fertiliser (slurry) application (15th October–12th January) for 2010–2013. This is the period when residual P losses were assumed to dominate (i.e., no incidental losses from fertiliser application followed immediately by rain events). Grazing is allowed during the closed period; animals were housed between late October/late November and late January/early March. Therefore, there may have been some incidental P losses associated with late grazing early in the closed period, but these are likely to have been small in comparison to the total losses of the closed period. Flow-weighted mean transfer pathway P concentrations were estimated from daily P loads and water flow for each pathway.

2.4. Stream biological impact

Stream benthic diatom and macroinvertebrate quality was measured in September (post summer) from 2009 to 2013 and in May (post winter) from 2010 to 2013 at four sites within the catchment. Sampling sites were distributed across several sub-catchments in order to capture the range of 1st–3rd order stream settings benthic communities in the catchment and were located to avoid impacts from local point sources where possible. Selected sites also had incomplete shade, a hard streambed substrate and riffle conditions suitable for the sampling methods. Benthic macroinvertebrate communities were collected using standard 2-min, traveling, kick-samples in the riffles using a long-handled pond net (250 mm width, mesh size 1 mm) according to ISO standards (ISO 7828; 1985) and national protocols (McGarrigle et al., 1992). The composition and abundance of invertebrate communities with different sensitivities to pollution were used to calculate a Q-value (McGarrigle et al., 1992). Diatoms were collected from natural substrates near to macroinvertebrate sampling sites and were assessed in accordance to the Diatoms for Assessing River Ecological Status (DARES) protocol to calculate Trophic Diatom Index (TDI) (Kelly et al., 2008).

Q-values and TDIs were expressed as ecological quality ratios (EQRs) by comparing the observed values with values expected if the site was at reference condition. An EU-wide intercalibration of EQR values against WFD water quality status classes (McGarrigle and Lucey, 2009) was used to assign WFD status classes. Because the sites used in this study were smaller than, and not part of the national monitoring program, results are considered as indicative of ‘potential’ rather than ‘actual’ WFD status classifications. General trends in the macroinvertebrate and diatom EQRs were analysed by site and season.

2.5. Economic impact

Economic and production data were recorded across 8 specialist dairy farms, a sample covering approximately 40% of the catchment, between 2010 and 2012, using an online tool, the ‘eProfit Monitor’ system (Teagasc, 2015). Some areas of these farms were outside the catchment but it was assumed that these farms were representative of the farmed catchment area. All of these farms had derogation from the stocking rate limits of the NAP. All but two of the farms were also in the farm gate P balance analysis. To ensure confidentiality, results from these 8 farms are reported cumulatively. Production patterns and gross margin returns were compared with the top 10% economically performing specialist dairy farms in the Teagasc National Farm Survey (NFS) in 2012 (Moran, 2014). The Teagasc NFS is collected on an annual basis as part of the Farm Accountancy Data Network requirements of the European Union (FADN, 2005). Comprehensive farm accounts and enterprise level transactions are chronicled on a random representative sample of farms throughout the Republic of Ireland by a team of trained recorders. Data were collected through the NFS on a total of 258 specialist dairy farms in 2012. Each farm is provided with a population weight to be representative of a number of farms in the national population. Population weights were generated by the Central Statistics Office (CSO) of Ireland (Teagasc, 2013). All data for this comparative analysis were derived based on the farm area assigned to dairy production based on the proportion of dairy livestock units on the farm.

3. Results

3.1. Farms and farm-gate P balances and use efficiencies

The 9 dairy farms included in the farm-gate P balance analysis were specialist dairy farms with minimal non-dairy enterprises (Table 2). Dairy farm-gate P balances ranged from −3.0 to 8.0 kg P ha^{-1} with a mean of 2.4 kg P ha^{-1} (Table 3). Farm-gate P use efficiency (PUE) ranged from 59 to 126% with a mean of 89%.

Part of the P applied to soils is intended to build up soil P fertility in soils that have agronomically sub-optimal P concentrations. In Ireland, 20 kg P ha^{-1} yr^{-1} is advised for index 1 soils and 10 kg P ha^{-1} yr^{-1} for index 2 for build up (Coulter and Lalor, 2008). Accounting for the estimated area of index 1 and 2 soils on these farms and their associated potential P build-up requirements to achieve optimal soil P concentrations (index 3), an “Optimal P Balance” was estimated. This Optimal P Balance indicates how the actual annual farm P imports compare to the annual imports that would be required to build the levels of soil P fertility on the farm to the agronomic optimal level (index 3), with negative values indicating that more P imports would be required. Dairy farm-gate Optimal P Balances ranged from −7.2 to 5.1 kg P ha^{-1} with a mean of −1.7 kg P ha^{-1} (Table 3). An eco-efficiency metric was calculated as the surplus P per ha/t milk solids (MS) produced and ranged from −0.06 to 0.45 with a mean of 0.11 kg surplus P ha^{-1} t MS^{-1} yr^{-1}.

3.2. Field P management and soil P concentrations

The NMRs indicated good compliance with the NAP regulations with almost no P applied within the relevant closed period (Fig. 2). Also, P application was skewed towards the earlier part of the grass growing season with 63 and 67% of P applied before June in 2010 and 2011, respectively. The average total P applied to monitored fields in 2010 was 34.0 kg P ha^{-1} yr^{-1}, ranging from 0 to 131.6 kg P ha^{-1} yr^{-1} and in 2011 it was lower at 26.2 kg P ha^{-1} yr^{-1}, ranging from 0 to 132.7 kg P ha^{-1} yr^{-1}. In 2010, 18% of monitored fields received P applications in excess of the maximum allowable 64 kg P ha^{-1} yr^{-1} for grassland (allowed for a P index 1 soil at pasture establishment under grazing at 2.5–2.9 LU ha^{-1}) and in 2011 this was lower at 10%. Organic fertilizer P dominated, making up 94 % of P applied to grassland fields in 2010 and 83% in 2011, and most of the excess P applications were associated with high rates of organic fertilizer application. This was mostly applied as cattle slurry, farm yard manure and soiled water.

Table 2
Characteristics of 9 specialised dairy farms for which farm-gate P balances were constructed.

	Mean	SD	Min	Max
Farm area (ha)	61	38	13	134
Grassland area (ha)	54	32	13	114
Grass area (% of total)	92	8	79	100
Milking platform[a] (ha)	36	15	12	57
Milking platform[a] (% of total)	68	18	42	91
Total livestock units (LUs)	140	78	39	245
Stocking rate[b] (SR) (LU ha^{-1})	2.4	0.5	1.7	3.1[c]
Grassland SR (LU ha^{-1})	2.6	0.5	2.0	3.5
Milking platform SR (LU ha^{-1})	3.9	1.4	2.0	6.0
Milk production[b] (l ha^{-1})	9761		6900	13097
Milk production[b] (kg MS ha^{-1})[d]	710		455	1005

[a] Area dairy cows can access on foot from the milking parlour.
[b] Calculated on a whole-farm area basis.
[c] Two farms were above the maximum 2.9 LU ha^{-1} for farms with a derogation. This was due to restrictions on livestock exports from these farms due to tuberculosis controls.
[d] MS = milk solids

Table 3
Farm-level P imports, exports, balances and use efficiencies for catchment farms in 2010–2011 (total of 17 farm years).

	Dairy farms				Beef farm	Arable farm
Imports (kg P ha^{-1} yr^{-1})	Mean	SD	Min	Max		
Inorganic fertilizer	5.2	4.6	0.0	12.7	2.9	4.8
Stock	0.3	0.5	0.0	1.5	0.0	0.0
Bulk feed	2.3	3.5	0.0	11.2	0.6	0.0
Concentrates	7.4	3.0	3.9	13.4	0.7	0.0
Organic fertilizer	1.0	3.0	0.0	9.1	0.0	23.9
Total imports (kg P ha^{-1} yr^{-1})	16.2	3.5	9.4	20.9	4.2	28.7
Exports (kg P ha^{-1} yr^{-1})						
Milk	9.3	2.1	7.1	12.3	0.0	0.0
Stock	4.3	1.0	2.4	6.0	4.3	1.0
Slurry/manure	0.0	0.0	0.0	0.0	0.0	0.0
Crops	0.1	0.4	0.0	1.2	0.0	24.9
Total exports (kg P ha^{-1} yr^{-1})	13.7	2.6	10.9	17.7	4.3	24.9
Balance (kg ha^{-1} yr^{-1})	2.4	4.1	−3.0	8.0	−0.1	3.8
Optimal P Balance (kg ha^{-1} yr^{-1})	−1.7	3.6	−7.2	5.1	−7.5	0.5
P use efficiency (%)	89	25	59	126	103	87
Milk solids exported (kg ha^{-1} yr^{-1})	713	189	501	1005	NA	NA
Eco-efficiency (kg surplus P ha^{-1} t MS^{-1} yr^{-1})	0.11	0.19	−0.06	0.45	NA	NA

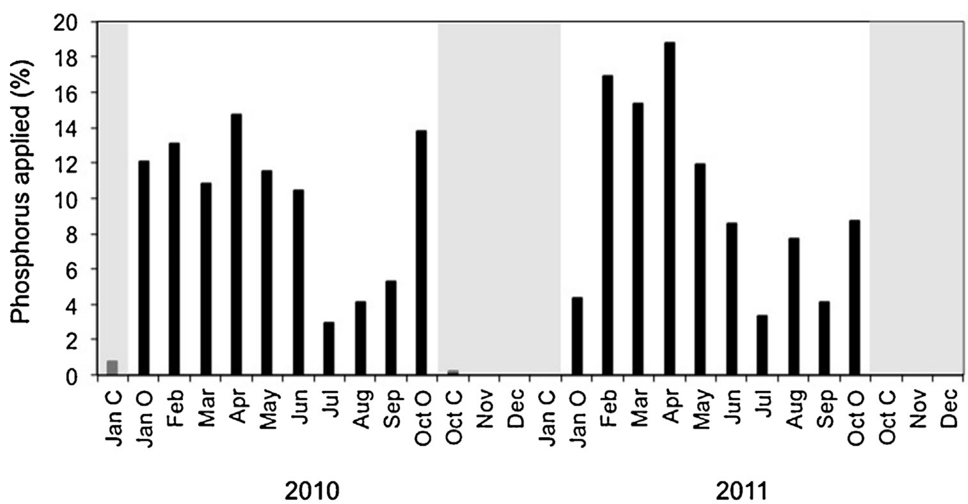

Fig. 2. Proportion of P applied to fields in the catchment for 2010–2011, showing regulated dates of closed periods for spreading organic fertilizers in grey. The proportion of P applied to fields during the closed (C) and open (O) periods for spreading organic fertilizers in October and January are indicated.

The total P load applied to soils across the whole catchment area (excluding deposition by grazing animals) was estimated, assuming similar P management practices on grassland and arable areas that were not captured in the NMRs as those that were, and assuming no P application to non-agricultural areas. Loads applied were markedly higher in 2010 at 28,768 kg P, or 38.0 kg P ha^{-1}, compared to 2011 at 19,820 kg P, or 26.1 kg P ha^{-1}. It was noted that P is also deposited directly by animals at pasture; however, the focus of the NAP is on field management of fertilizer P. In any event loads applied are likely to be correlated with P deposited.

For soil sample areas that were sampled in 2010 and resampled in 2013, the area-weighted mean soil P concentration decreased from 6.9 mg l^{-1} to 6.4 mg l^{-1}. Importantly, the proportion of area in index 4 (excessive) decreased from 32 to 24% (Fig. 3) and the area in index 3 (optimum) increased from 27 to 36%, while the area in index 2 (deficient) decreased from 25 to 22%, indicating a convergence of soil P concentrations on the agronomic optimum. However, the area in index 1 (very low STP) increased from 16 to 18%.

Over the 3 year period the initial index 1 and 2 soils showed a small increase in P concentration, on average, while initial index 3 soils remained relatively constant (Fig. 4). Initial index 4 soils showed the largest change, with an average decrease of 2.1 mg l^{-1}. The majority (67%) of the P load in 2011 was applied to soils of <8 mg l^{-1} (index 1–3 for grassland) but P was also applied to soils above the 8–10 mg l^{-1} threshold, albeit largely as recycled organic

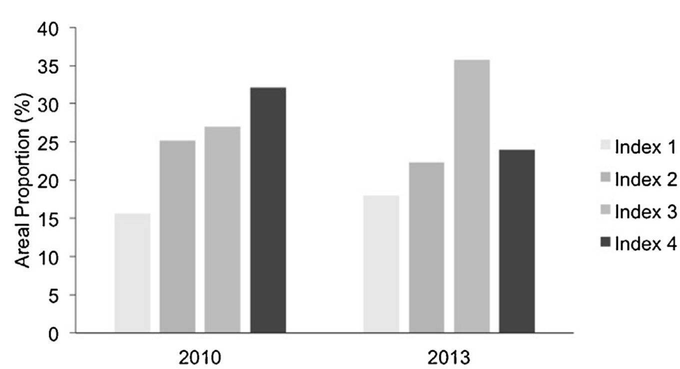

Fig. 3. The areal proportion of soils in each soil P index in 2010 and 2013.

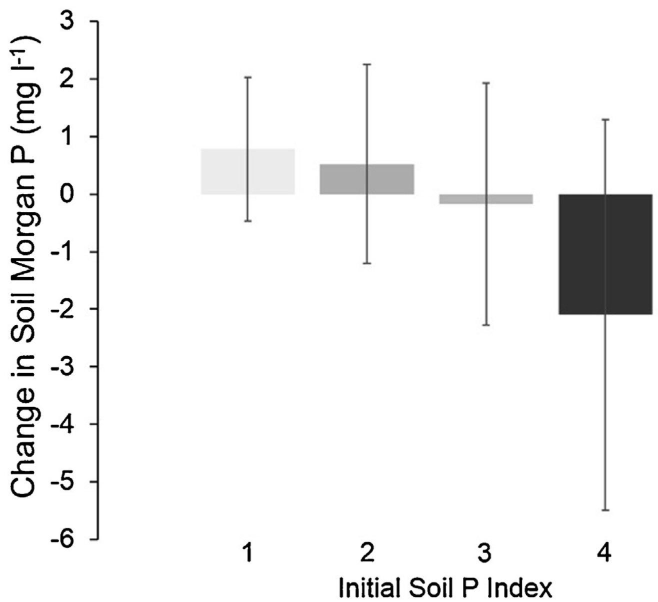

Fig. 4. Mean change in soil Morgan P concentration over three years by initial soil P Index. Error bars indicate standard deviation.

fertilizers and only accounting for a small proportion of the total P applied in the catchment.

3.3. Phosphorus loss in streamflow

Over the April–March monitoring periods, TP and TRP loads, respectively, delivered to the stream outlet were 0.62 kg ha^{-1} and 0.37 kg ha^{-1} in 2010–2011, 0.41 kg ha^{-1} and 0.25 kg ha^{-1} in 2011–12 and 1.56 kg ha^{-1} and 0.77 kg ha^{-1} in 2012–2013. These loads were associated with runoff of 532 mm, 427 mm and 869 mm, respectively. Closed period P loads were lowest in 2011–2012 (0.19 kg ha^{-1} TP and 0.12 kg ha^{-1} TRP) and highest in 2012–2013 (0.35 kg ha^{-1} TP and 0.19 kg ha^{-1} TRP) (Table 4). Mean annual TRP concentrations were equal to or below the national good water quality status threshold of 0.035 mg l^{-1} for the low flow Q85–95 discharge range but were above it for all other ranges (Table 5).

The calculated percentage of water flow pathways and phosphorus transfer pathways in the closed period was similar over the three years (Fig. 5). On average, 13% of the water discharge was generated as *Quick flow*, 5% as *Interflow* and 82% as *Delayed flow*. *Quick flow* transferred, on average, 46% of the TP and 39% of the TRP, and *Delayed flow* transferred 46% of TP and 53% of TRP. Only a small percentage (5–8%) of the water discharge and P transfer occurred as *Interflow*. Closed period TP load was highest in *Quick flow* (except in 2011–2012) and TRP load was highest in *Delayed flow* (Table 4). Flow-weighted mean TP and TRP concentrations, calculated from the daily data, decreased in the *Quick flow* and *Interflow* pathways over the monitored period but remained similar, with only a slight decrease, in the *Delayed flow* (Fig. 6). Therefore, observed rates of decrease appear to be lower in deeper pathways associated with longer travel times.

Table 4
Phosphorus loads separated for *Quick flow* (QF), *Interflow* (IF) and *Delayed flow* (DF) for three closed periods.

	TP (kg ha^{-1})				TRP (kg ha^{-1})			
Closed period	QF	IF	DF	Total	QF	IF	DF	Total
2010–2011	0.113	0.018	0.104	0.235	0.060	0.012	0.072	0.145
2011–2012	0.078	0.013	0.102	0.194	0.039	0.008	0.071	0.118
2012–2013	0.173	0.028	0.143	0.345	0.081	0.017	0.091	0.188

Table 5
Mean stream P concentration by discharge ranges. No discharges were recorded at Q85–95 for the 2012–2013 period. Concentration range is shown in brackets.

	2010–2011	2011–2012	2012–2013
Q85–Q95 (0.014–0.024 m^3 s^{-1})			
TP (mg l^{-1})	0.053 (0.020–0.660)	0.052 (0.033–0.391)	–
TRP (mg l^{-1})	0.035 (0.014–0.509)	0.032 (0.021–0.302)	–
Q75–Q85 (0.024–0.038 m^3 s^{-1})			
TP (mg l^{-1})	0.058 (0.021–0.753)	0.064 (0.033–0.696)	0.062 (0.032–0.450)
TRP (mg l^{-1})	0.038 (0.011–0.649)	0.043 (0.019–0.482)	0.038 (0.018–0.343)
Q45–Q55 (0.087–0.124 m^3 s^{-1})			
TP (mg l^{-1})	0.085 (0.019–1.432)	0.080 (0.042–0.989)	0.088 (0.042–0.837)
TRP (mg l^{-1})	0.056 (0.012–1.097)	0.053 (0.033–0.578)	0.056 (0.029–0.449)
Q5–Q15 (0.245–0.375 m^3 s^{-1})			
TP (mg l^{-1})	0.102 (0.042–1.062)	0.089 (0.041–0.581)	0.099 (0.043–0.709)
TRP (mg l^{-1})	0.064 (0.027–0.704)	0.057 (0.028–0.416)	0.062 (0.025–0.328)

3.4. Stream biological quality

The quality of the stream benthic flora and fauna community was generally impaired post-summer with just one site achieving 'Good' diatom status and just two sites achieving 'Good' macroinvertebrate status (Fig. 7). Biological quality tended to be better post-winter with all sites achieving 'Good' diatom status at some stage and three of the sites achieving 'Good' macroinvertebrate status every (outlet and main stream site), or almost every (headwater site) year. These seasonal trends of better biological quality post winter were stronger than any other identifiable inter-annual trend over the five years. There were some early indications of significant improvements (i.e., crossing WFD status boundaries) in post-summer diatom quality at the main and tributary sites over the five years. However, there were also some early indications of significant declines in post-summer macroinvertebrate quality at the tributary and outlet sites over the same period. Longer-term data-sets are required to determine if these early indications progress into clear trends.

3.5. Comparative economics

Expansion occurred on the sample of catchment dairy farms between 2010 and 2012 with more land and an increased number of dairy cows in milk production. However, the average stocking rate decreased from 2.56 to 2.48 LU ha^{-1} due to the increased quantity of land. In 2012, production and gross margin returns

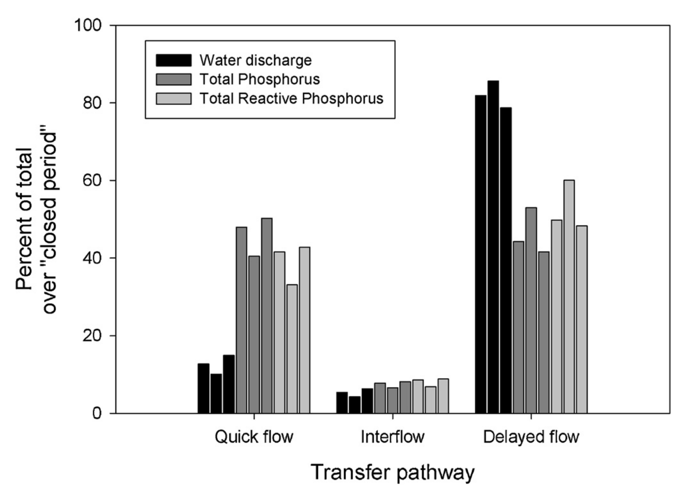

Fig. 5. Proportion of stream water discharge and TP and TRP load at the catchment outlet during three closed periods in 2010–2013 for flow and transport pathways.

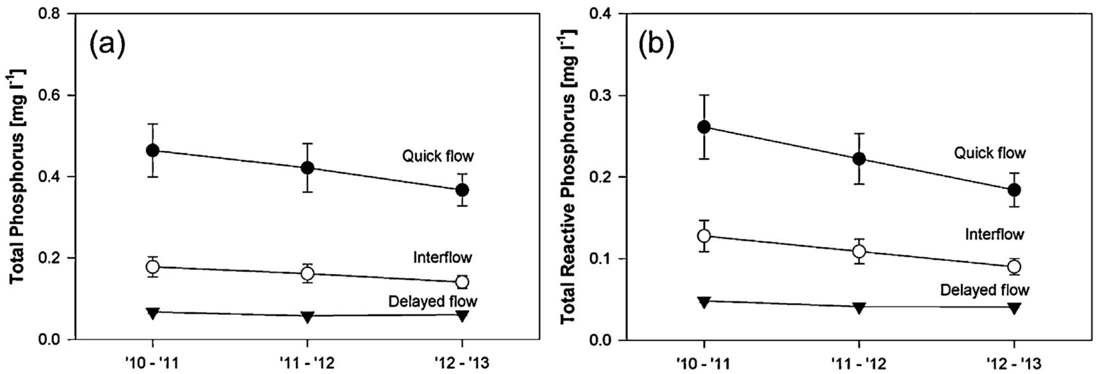

Fig. 6. Flow-weighted mean TP (a) and TRP (b) concentrations based on daily transfer pathways over three closed periods. Error bars indicate standard error.

were almost identical to the top 10% economic performing specialist dairy farmers nationally (Table 6). Farm stocking rate (2.48 versus 2.47 LU ha^{-1}), milk output of land devoted to dairying (14,585 versus 14,356 l ha^{-1}; 1125 versus 1045 kg milk solids ha^{-1}) and milk output per cow (5874 versus 5812l per cow; 450 versus 428 kg milk solids per cow) were slightly higher for the catchment farms while gross margin return to dairying (€ 3130 ha^{-1} versus € 3261 ha^{-1}) was marginally higher for the top 10% of specialist dairy farmers nationally.

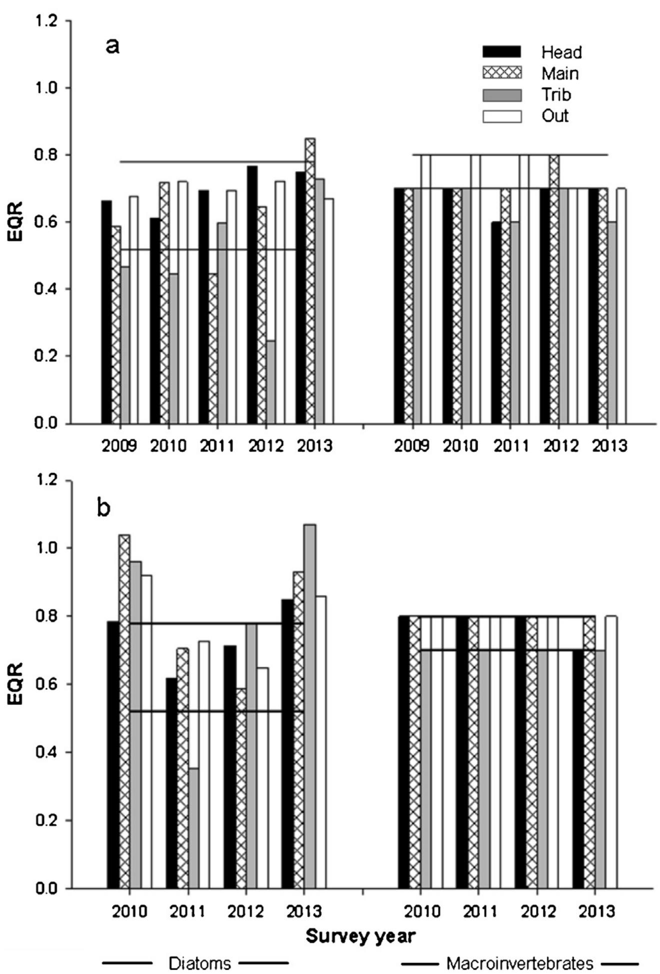

Fig. 7. Observed EQRs for diatoms and macroinvertebrates for (a) post-summer and (b) post-winter. Horizontal solid lines show poor-moderate and moderate-good status boundaries, where higher EQR values indicate better status. 'Head' = headwater site, 'Main' = main stream site, 'Trib' = tributary site and 'Out' = outlet.

4. Discussion

Compared to Irish dairy farms prior to the NAP measures (2006), P balances and PUEs on the dairy farms in this catchment (2.4 kg P ha^{-1} and 89%, respectively) appeared much improved, with Treacy (2008) reporting a P surplus of 5.6 kg P ha^{-1} and PUE of 68% for 21 commercial dairy farms (2003–2006) and Mounsey et al. (1998) reporting a P surplus of 19.5 kg P ha^{-1} and a PUE of 37% for 12 commercial dairy farms (in 1997). This suggests an improvement in P balance and PUE over time (Mihailescu et al., 2014). These results also compare favorably to other largely grass-based dairy farms in northwest Europe, with a similar range of milk outputs (4837–9847 l ha^{-1}), for which Raison et al. (2006) found P surpluses of 9.7–21.6 kg P ha^{-1} and PUEs of 33–57%. Feed and fertilizer P imports were low relative to many continental European dairy farms (Aarts, 2003; Nielsen and Kristensen, 2005; Raison et al., 2006). These results indicate that, while the dairy farms in this catchment are intensive, with a relatively high stocking rate and milk production per hectare, they may be exerting less of a P source pressure than similar continental European systems.

While it is not possible to attribute this improvement entirely to the impact of the NAP measures, as other factors such as P fertilizer and feed cost and agronomic advice impact are likely to be important, it would seem likely that the NAP measures, in placing restrictions on field application rates and farm-gate imports, have been effective in achieving at least some of this improvement in P management on dairy farms. The mean Optimal P Balance of −1.7 kg P ha^{-1} indicates that the nutrient source pressure may be even lower than the conventional balance suggests. This has been achieved while also achieving milk output and gross margin returns at the highest level and suggests a quite considerable improvement in resource use efficiency. The range in P balance and PUE observed in this study and in that of both Buckley et al. (2013) and Mihailescu et al. (2014) also indicates the potential to decrease P surpluses and improve PUE on dairy farms while maintaining, or even increasing, production through improved management.

As was highlighted by Mihailescu et al. (2014), this improvement has been largely due to a large decrease in the levels of inorganic fertilizer P import relative to comparable studies, from 23.5 kg P ha^{-1} in Mounsey et al. (1998) to 10.2 kg P ha^{-1} in Treacy (2008) to 7.6 kg P ha^{-1} in Mihailescu et al. (2014), 6.7 kg P ha^{-1} in Buckley et al. (2013) and 5.2 kg P ha^{-1} in the current study. As a result, imported feed P has become a much larger proportion of imported P, although absolute levels appear not to have changed significantly (Mihailescu et al., 2014). On the dairy farms in this catchment, this trend would appear to have gone further, with feed P now making up the largest part of overall P imports (60%). It is also consistent with this shift in P resources that the majority of P

Table 6
Trends in dairy production indicators and comparison with the top 10% of dairy farms nationally.

	Catchment farms 2010 $n=8$	Catchment farms 2012 $n=8$	NFS – top 10% dairy farms 2012
Production indicators[a]			
Milk production ($l\,cow^{-1}$)	5973 (5341–7569)	5874 (4210–7983)	5812 (4493–8556)
Milk production ($l\,ha^{-1}$)	15,178 (10,831–19,677)	14,585 (9941–17,962)	14,356 (11,047–20,336)
Milk solids (kgs protein and butterfat – cow^{-1})	457 (378–569)	450 (292–596)	428 (349–630)
Milk solids (kgs protein and butterfat – ha^{-1})	1146 (766–1462)	1126 (690–1470)	1045 (817–1387)
Whole-farm stocking rate ($LU\,ha^{-1}$)	2.56 (2.03–2.87)	2.48 (2.09–3.00)	2.47 (1.67–3.38)
Gross margin (€ ha^{-1})	2911 (2058–4379)	3130 (2105–4057)	3261 (2811–4036)
Production trend 2010–2012			
Total hectares	425	470	
Hectares devoted to milk production	309	322	
Total milk production (Ml)	4.69	4.69	
Total dairy cows	785	799	
Total LUs	1087	1166	

[a] Calculated on the basis of land assigned to dairy production unless otherwise stated. Mean and range (in brackets) are reported.

applied to fields in this catchment is in organic fertilizers (89% for grasslands). Irish dairy farms are apparently increasingly reliant on organic fertilizer P generated on-farm to supply the majority of the P needed to maintain soil P fertility and crop P requirement. It is also likely that farms are relying on “mining” the soil reserves of P built up over previous years to decades, as evidenced by the overall decreasing soil test P concentrations in this catchment. The negative P balances and PUEs > 100% of some dairy farms in this catchment would indicate such a reliance on soil P reserves.

This trend towards increasing reliance on organic fertilizer P may present challenges to achieving optimal management at the field scale as the P content of organic fertilizers is highly variable (O’Bric, 1991) and controlling application rates and timing can be more difficult than with inorganic fertilizers. The large variability in field application rates and magnitude of soil P changes in this study are consistent with this. The NAP regulations assume standard total P contents for organic fertilizers ($0.8\,kg\,P\,m^{-3}$ for cattle slurry). In addition, up until very recently, the NAP measures required an assumption of 100% plant-availability of P in organic fertilizers. This is not likely to be the case, at least within the year of application (Heming, 2008; Oberson et al., 2010). In addition, depending on the initial soil test P concentration, different soils have varying capacity to fix a proportion of the P applied, rendering some unavailable for plant uptake (Sheil et al., 2014). The assumed P availability for organic P fertilizers applied to low P soils (P index 1 and 2) has been changed to 50% in the most recent version of the measures (SI 31, 2014). Certainly, the dominance of feed P imports at farm-gate and organic fertilizer P application at field level, the low import and use of fertilizer P and low Optimal P Balances demonstrate quite a large shift in P sources, management and balances on dairy farms. Results suggest that, even if sufficient P was being imported at the farm-gate, in many cases, this P was not distributed adequately at field level to where it was required, highlighting potential for further improvement in P management to match P application to P requirement at the field scale.

Fig. 2 indicates good adherence to regulations in nutrient source management during the closed winter period, and prevention of P application to soils when the risk of incidental losses is highest (Preedy et al., 2001). Also, P application in both years was skewed towards the spring, allowing for better contribution to crop growth. The upper limits of annual field P application rates were breached in only a minority of cases and the majority of P was applied to soils of index 1 to 3, also indicating good practice and adherence to the nutrient source management regulations. However, P was applied to some index 4 soils; this is not a desirable outcome as it indicates excess application on some soils and suggests that some farmers may not be making best use of their P resource by applying based on soil P test results in all instances. Again, this highlights strong potential for further improvement in P management to match P application to P requirement at the field scale based on soil test results and advisory input.

Despite this, the decrease in fields receiving P in excess of the maximum allowed for grassland from 2010 to 2011 suggests that advisory input may be having an impact in this regard in building understanding of nutrient management plans at field level. Most excessive applications were associated with application of organic fertilizers and this suggests that more might be done to improve farmer awareness of the P content and value of organic fertilizers. In any knowledge transfer process a time lag can be expected between advisor–farmer engagement and farmer uptake of measures (Hodge and McNally 1998). Advisors began working closely with farmers in this catchment in 2010.

Overall, the observed decreases in mean surface soil P concentrations and the proportion of area in index 4 (Fig. 3), indicate quite a substantial decrease in soil P source pressure in the catchment, likely as a result of the P management outlined above. Wall et al. (2013) modeled the decline in area of index 4 soils in this catchment from the same baseline data of 2010 under differing field-level P balance scenarios and estimated that over 3 years at $-7\,kg\,ha^{-1}\,yr^{-1}$ the soil area in index 4 would decrease by ca. 60%. The data presented here shows that the area in index 4 decreased by 25% over 3 years at a mean farm balance of $2.4\,kg\,P\,ha^{-1}\,yr^{-1}$ (range -3 to $8\,kg\,ha^{-1}\,yr^{-1}$). It is notable that the reductions in P index 4 soils were achieved with no significant change in production metrics between 2010 and 2012. Nevertheless, any proposal for deeper P deficits to mine down soil P further would need to be cognisant of the tradeoffs between the speed of decline and the economic risks of constraining immediate crop supply. Frequent (at least every 4 years) soil sampling should ensure maintenance of adequate soil P concentrations; the NAP measures account for this and allow additional P to be applied to soils once they fall below the optimal index 3 level ($5.1–8\,mg\,l^{-1}$ for grassland soils). Also, the time lag between implementation of the NAP measures and the desired outcome of few or no fields with excessive soil P that was highlighted by Wall et al. (2013) is likely to be longer than the 10 years estimated in that study; the modeled rates of decline were subject to large uncertainties (Schulte et al., 2010).

The fate of the $2.4\,kg\,P\,ha^{-1}\,yr^{-1}$ mean farm-gate surplus will be determined by a range of factors but, in the long term, it will be incorporated into the soil/plant system and/or lost to water (surface or groundwater). Given that the mean Optimal P Balance was $-1.7\,kg\,P\,ha^{-1}$ it would seem likely that a large proportion would be incorporated into the soil/plant system. The 0.41 and 0.62 kg TP $ha^{-1} yr^{-1}$ losses in the stream in the first two years are small in comparison to the annual P surplus and are moderately low compared to those reported by Jordan et al. (2005) and Ulén et al. (2007) for six Irish grassland catchments (0.16–3.12 kg ha^{-1} yr^{-1}) and Kronvang et al. (2005) for two Norwegian catchments (0.5–5.8 kg $ha^{-1} yr^{-1}$) and similar to those reported by Melland et al. (2012) for two arable Irish catchments (0.27 and 0.72 kg $ha^{-1} yr^{-1}$). The mean TP loss over the three years of 0.86 kg ha^{-1} yr^{-1} is higher than that of a range of European catchments (0.43 kg $ha^{-1} yr^{-1}$) (Kronvang et al., 2003) but those catchments included some that were non-agricultural. The higher P losses in the third year, which increased the mean over the period, were consistent with an extremely wet year with approximately double the runoff of the previous year.

The dominant closed period transport pathways were *Quick flow*, representing components of overland flow, shallow subsurface flow and rapidly connected field drainage, and *Delayed flow*, representing slower and deeper subsurface flow. The dominance of *Quick flow* in P transfer, despite accounting for only 13% of discharge, is consistent with acute P loss during high flow events, as found in many studies (e.g., Lazzarotto et al., 2005; Melland et al., 2012). The importance of delayed flow, however, is less commonly observed but is consistent with the findings of Jordan et al. (2012) who found that this catchment has an intermediate runoff flashiness (Q5:Q95 ratio), leading to hydrological buffering of the rainfall-runoff signal. It is also consistent with Mellander et al. (2012) who found that belowground P transfers were important and that baseflow and a long recession in water flow following events can deliver substantial loads of P due to elevated P concentrations. These trends are likely due to the high proportion of well drained soils (Fig. 1) and the permeability of sediments, weathered bedrock and rock fissures in the catchment. Phosphorus loss via groundwater and other subsurface flow in this catchment is significant and TRP dominated P in *Delayed flow* (67–72%), with implications for the effectiveness of measures targeted primarily at preventing loss via overland flow and particulate P, as has been discussed elsewhere (Mellander et al., 2012).

Higher mean TRP concentrations at higher discharges also indicates that P loss is not source-limited and exceedance of the national concentration threshold in rivers occurred at higher discharges. However, it should be noted that national monitoring data is much more coarsely sampled. The proportion of P lost during the closed period, despite the lack of nutrient application during this period, supports the closed period policy as a time of high P loss risk (Jordan et al., 2012). The lowest proportion of annual P lost during the closed period, however, occurred in the wettest year (2012–2013), indicating that P loss risks may be high at other times of the year, depending on weather. The 2012–2013 period was the wettest year recorded and notable for a wet summer period, which would increase incidental P losses.

The higher total P load applied to the catchment soils of 38.0 kg P ha^{-1} in 2010, relative to the lower load of 26.1 kg P ha^{-1} in 2011, coincided with the higher annual and closed period losses (Table 4) and higher concentrations of TRP and TP in all three pathways in 2010–11 (Fig. 6). Further P loading data were not available for the third year but the significantly lower coverage of index 4 fields suggests that this had not, at least, excessively increased.

Overall, successive declines in closed period *Quick flow*, *Interflow* and, to a lesser extent, *Delayed flow* TRP and TP concentrations suggest a response to this reduced P source pressure. A slower response is expected in *Delayed flow* groundwater pathways and the quicker response seen in *Quick flow* and *Interflow* would be consistent with the P in these pathways being more directly connected to, and responsive to, the concentration of P in surface soils. Baseflows from a *Delayed flow* pathway, however, were much higher during the third year. This was associated with greater runoff and annual (including incidental) and closed period P loads and highlights the difficulty in assessing the impact of nutrient source measures on stream water quality over time scales as short as 3 years (Wilcock et al., 2013). Inter-annual variability in hydrometeorological drivers of P loss may mask trends in source drivers of P loss that may only become apparent over longer time periods.

Furthermore, seasonal trends of better biological water quality post winter were stronger than any other identifiable trend over the three years. At this end of the nutrient transfer continuum, it would appear that any signal for improvement in nutrient sources has been largely attenuated, indicating a diminishing cascade response to changes in nutrient source pressures along the transfer continuum. This has important implications for policy makers and other stakeholders as it is at the stream impact end of the continuum that policy effectiveness is assessed, in terms of changes in water quality. Over time, it can be expected that improvements in management and decreases in nutrient source pressure will lead to improvements in water quality (e.g., O'Dwyer et al., 2013) and the decrease in closed period TP and TRP concentrations in *Quick flow* and *Interflow* suggests that the positive nutrient source change signal may be beginning to emerge at the nutrient delivery end of the continuum.

This study highlights some important policy choices. Firstly, the economics of production and stakeholder (farmer) buy-in are important. Notably, the reduction in P source pressure in this catchment was made whilst maintaining farm production and financial returns comparable to the top 10% of dairy farms nationally, indicating that nutrient source pressure can be reduced whilst maintaining or improving production and profitability. Secondly, where P transfer in sub-surface hydrological pathways is significant, such as in free draining catchments like this, P source pressure reductions may be slow to convert to definite downward trends in P delivery and impact (Melland et al., 2014); i.e., the message is to establish an accepted mitigation strategy and stick with it. Thirdly, a more engineered approach to attenuate P in pathways could be considered.

In any monitoring and policy evaluation approach, it appears to be extremely important to provide nutrient data across the continuum from source to delivery and impact and at a manageable scale (e.g., Wilcock et al., 2013) to ensure that the interpretation of water quality responses to policy are fully informed. Assessment at the source end of the continuum also allows early trends in nutrient source change to be detected before they become manifest in delivery and impact, as has been noted by others (e.g., Cherry et al., 2008; Meals et al., 2010; Sharpley et al., 2009). This is important to demonstrate the degree and direction of change in the nutrient transfer continuum at an early stage, such as the positive changes observed in this study.

5. Conclusions

The intensive dairy farming catchment studied here showed significant trends towards reducing the P source pressure on water quality while maintaining dairy farm production and profitability at a high level. Feed and fertilizer imports, P balances and PUE in these grazed grass-based systems also compare favorably with continental European dairy farms. This suggests that the NAP may be having the intended positive effects at the nutrient source end of the transfer continuum.

There was also some evidence of a response to this reduced P source pressure in closed period P concentrations and losses in the catchment stream, particularly in *Quick flow* and *Interflow* concentrations, but there were no clear trends of improving biological quality over the three years. This suggests a diminished response to policy between P source pressure and P delivery and impact, at least over the short time scale of this study (3 years), where the strength of any observable trend along this cascade of responses is greater closer to the source end of the nutrient transfer continuum. Therefore, significant time lags can be expected between implementation of policy, decreases in nutrient source pressures and improvements in water quality impact and policy monitoring and assessment efforts will need to be cognisant of this.

These results have important implications for DPSIR reviews of the effectiveness of programmes to improve nutrient management and water quality and expectations of water quality improvements. The attenuation of the nutrient transfer signal implies that early signs of policy effectiveness will most likely be observable closer to the nutrient source end of the continuum and that policy-makers need to be aware of the expected time-lags that can result between effective implementation of a program of measures, improvements in nutrient source management and resulting observable improvements in monitored water quality. Therefore, policy-support monitoring efforts might ideally include monitoring of the nutrient source end of the continuum, such as that outlined in this paper, to detect early signs of policy effectiveness. In the EU context, such monitoring may be particularly suitable to detect effectiveness of the NAPs under the ND, that specifically target measures at agricultural nutrient source management. Conversely, long term (greater than 3 years, at least, but likely considerably longer) monitoring at the delivery and impact end of the continuum appears necessary to detect policy effectiveness. Such monitoring may be particularly suited to assess achievement of the water quality targets under the WFD that sets the targets to be achieved at this end of the continuum. Importantly, a high intensity of monitoring across the nutrient transfer continuum has been necessary to carry out the assessment detailed in this study adequately and this has implications for the resources needed to implement an effective policy-support monitoring program.

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