Received: 19 April 2021 | Accepted: 25 August 2021 | Published online: 5 October 2021

DOI: 10.1002/jeq2.20285

Journal of Environmental Quality

REVIEWS AND ANALYSES

Applying the nutrient transfer continuum framework to phosphorus and nitrogen losses from livestock farmyards to watercourses

Sara E. Vero[1] | **Donnacha Doody**[2]

[1] Dep. of Science and Computing, Waterford Institute of Technology, Co., Waterford, Ireland

[2] Agri-Food and Biosciences Institute, 18A Newforge Lane, Belfast, Northern Ireland

Correspondence
Sara E. Vero, Dep. of Science and Computing, Waterford Institute of Technology, Co., Waterford, Ireland.
Email: saraevero1@gmail.com

Assigned to Associate Editor Tamie Veith.

Abstract

Farmyards are commonly conceptualized as point sources of nutrient pollution nested within the wider agricultural landscape. However, within farmyards there are individual sources and delivery pathways, each of which is affected by a range of management practices and infrastructure. Rainfall mobilizes these nutrients, which may then be delivered to a receptor or to the wider drainage network. As such, the nutrient transfer continuum (NTC), which has been established as a framework to understand and mitigate nutrient loss at a landscape scale, can be similarly applied to disentangle the stages of nutrient transfer from farmyards. The NTC differentiates nutrient transfer into source, mobilization, delivery, and impact stages. This differentiation allows targeting of mitigation measures and evaluation of costs and benefits. This review paper applies the NTC template to farmyard nitrogen and phosphorus transport to conceptualize causative factors and to identify mitigation options.

1 | INTRODUCTION

Farmyards are commonly conceptualized as point sources of pollution nested within the wider agricultural landscape that deliver contaminants to waterbodies at clearly identifiable locations. However, within a farmyard there are multiple different contaminant sources (e.g., nutrients, organic matter, pharmaceuticals, etc.) and delivery pathways, each of which are affected by various management practices and infrastructure. Farmyard structure, use, and management varies across farm system types and geographical areas, and the importance of farmyards as a hub of farm activities varies across time and place. Within the United Kingdom and Ireland, pasture-based livestock farms typically include outdoor farmyards consisting of livestock housing, handling facilities (such as dairy parlors and holding and drafting pens), waste storage for manures and effluents, equipment sheds and workshops, and feed stores including silage pits or bale stores (Aitken et al., 2003). Farmyards typically include an impervious hardstanding. This is the platform for many activities, including holding and handling of livestock, movement of feed and bedding, storage of vehicles and equipment, among other incidental activities. Although there is a shift toward increased housing of livestock, particularly toward "zero-grazing" systems (van den Pol-van Dassalaar et al., 2020), livestock and dairy production systems in Ireland and the United Kingdom are still largely outdoor, with pasture-based systems representing 98 and 92% of Irish and British dairy farms, respectively (Crump et al., 2019). Housing of livestock in these systems is largely restricted to the winter period, during which weather conditions are harsh or during which soil is vulnerable to compaction and poaching. Within these systems, farmyards remain a primary hub for farm activities, and

Abbreviations: BOD, biochemical oxygen demand; DOC, dissolved organic carbon; FYM, farmyard manure; ICW, integrated constructed wetland; NTC, nutrient transfer continuum.

failure to manage them correctly can detrimentally affect air and water (Misselbrook et al., 2001; Vero et al 2019). Under the EU Nitrates Directive (European Commission, 1991), member states are required to, among other things, ensure that "the capacity and construction of storage vessels" for livestock manures, effluents, and silage are sufficient and that runoff and seepage is prevented. The precise specifications of storage facilities and buildings and the measures by which the farmyard and nutrient sources are to be maintained vary between nations. For example, in Ireland, farmyards are regulated under the Code of Good Agricultural Practice (S.I. 605; Office of the Attorney General, 2017), with the equivalent legislation in Northern Ireland (U.K.) being the Code of Good Agricultural Practice for the Prevention of Pollution of Water, Soil and Air (Department of Environment, Food & Rural Affairs, 2009).

Several farm- and catchment-scale longitudinal surveys of nutrient concentrations across river and drainage networks have detected elevated phosphorus (P) and nitrogen (N) in both water and bed sediment in watercourses and artificial drains that receive farmyard discharges (Harrison et al., 2019; Moloney et al., 2019; Vero et al., 2019; Withers & Hodgkinson, 2009). Edwards and Hooda (2009) measured P and N concentrations up- and downstream from a mixed-enterprise farmyard (dairy and sheep production, 200 m^2) with two discrete point discharges into the watercourse. Significant temporal variabilities were observed in the discharges from each source, reflecting seasonal patterns and daily variation, where one source reflected periodic contributions from dairy washings. Losses from yards typically become more impactful during summer months when diffuse losses are suppressed; the relative contribution of farmyards to total nutrient loads has been observed to fluctuate from 5% during high-discharge periods to 90% during low-discharge periods (Tunney et al., 2000). Edwards et al. (2008) observed differences in impervious, or hardstanding runoff concentrations to vary by up to four orders of magnitude across four monitored farmyards. Dunne et al. (2005) reported that effluent volumes from a 42-head dairy farm varied from 3.6 to 18.5 m^3 d^{-1}, whereas annual nutrient exports for soluble reactive P and ammonium were 47 ± 10 and 128 ± 35 kg yr^{-1}, respectively. In a similar study, Forbes et al. (2011) found that for a 170-head dairy herd average discharge from the farmyard was 49 m^3 d^{-1}, with total P concentrations of the effluent ranging from 24 to 132 mg L^{-1} and ammonium concentrations ranging from 3.3 to 15.7 mg L^{-1}. Mustafa et al. (2009) observed SRP concentrations of 11.5 ± 10.1 mg L^{-1}, 39.6 ± 41.7 mg L^{-1} ammonia-N, and 3.8 ± 3.4 mg L^{-1} nitrate-N in effluent for a 0.5-ha farmyard for a 77-head dairy herd.

These studies highlight the significant contribution that farmyards can make to agricultural nutrient loads to waterbodies. Typically, point sources such as septic tanks are relatively simple, consisting of a single nutrient source or input and a distinct discharge point. These can be accounted for in nutrient and hydrological models using easily inferred or default values. Conversely, within each individual farmyard there are distinct nutrient sources, mobilization and transport factors, and variable temporal and quantitative factors of delivery depending on the timing of farmyard activities and weather. It would be more appropriate to conceptualize farmyards as subsystems nested within the wider agricultural system. Magette et al. (2007) acknowledged farmyard conditions as one controlling factor on P export within their catchment-scale risk index. In that system, expert opinion was used to assign risk to certain source factors (slurry, manure, and effluent storage factors), proportion of soiled hardstanding, and some management factors corresponding to mobilization/transport stages. Although that study presented a practical decision support tool for identifying risky areas and practices, it did not represent a "deterministic description of P movement in the terrestrial environment" (Hughes et al., 2005). Although farmyards have been included in a number of review papers (e.g., Hooda, Edwards, et al., 2000; Schoumans et al., 2014), none of those have disaggregated individual sources, pathways, and the risks associated with management practices within farmyards. The aim of this paper is to provide a review of nutrient loss from farmyard in the context of the nutrient transfer continuum (NTC), which was originally presented by Haygarth et al. (2005) in the context of the diffuse source. It is acknowledged that other contaminants originate within the farmyards, such as veterinary medicines (Sinclair et al., 2007), hormones (Cai et al., 2013), fecal indicator organisms (Aitken, 2003), disinfectants (Brewer et al., 1999), and effluents having high biochemical oxygen demand (BOD) (Cumby et al., 1999; Novotney, 1999), but these contaminants are not addressed in this paper.

Core Ideas

- The nutrient transfer continuum framework is applied to farmyard N and P losses.
- Nutrient sources include urine, faeces, silage effluent, and parlor washings.
- Mobilization is controlled by the timing and magnitude of rainfall.
- Delivery is controlled by connectivity of the farmyard with the drainage network.
- Mitigation measures include management and infrastructure.

2 | APPLYING THE NUTRIENT TRANSFER CONTINUUM FRAMEWORK

The NTC presents a four-stage conceptual framework describing the transport of P to watercourses (Haygarth et al., 2005).

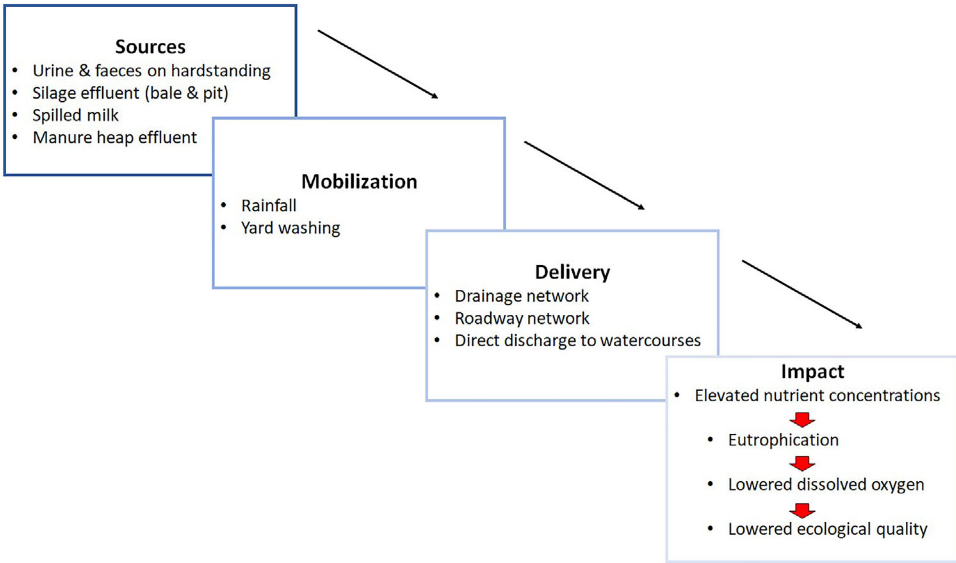

FIGURE 1 Conceptual illustration of the nutrient transport continuum framework adapted from Haygarth et al. (2005) as applied to the farmyard

This conceptual framework can be used to disaggregate the overall phases contributing to nutrient loss, allowing identification of key contributing factors, and targeting of mitigation measures (Wall et al., 2011). Previously this conceptual model has been applied to non-point source P losses at the landscape or catchment scale (Haygarth et al., 2005; Murphy et al., 2015; Wall et al., 2011). The present paper applies this framework to farmyard nutrient transfer by considering the farmyard as a subsystem positioned within the wider landscape. Delivery of nutrients from farmyards to watercourses may still occur at specific, identifiable points, but internally, source, mobilization, and transport factors can be differentiated. The impact phase occurs only in the watercourse itself and, as with any application of the NTC, may be distant from the start of the continuum (Holden et al., 2017). The NTC phases within the context of farmyards are illustrated in Figure 1. Briefly, sources refer to inputs of nutrients to the system. Mobilization is the initiation of transfer, during which nutrients are physically detached or solubilized from their initial position. Delivery refers to the routing of entrained nutrients from their source to the receptor. Finally, impacts are the changes observed in the receptor as a consequence of nutrient addition.

2.1 | Sources

2.1.1 | Urine and feces

Livestock waste produced from housed animals is frequently stored in the farmyard until applied as a fertilizer when ground conditions are suitable and the probability of rainfall is low. Under the Nitrates Directive (European Commission, 1991), application of livestock waste is prohibited during winter months when plant utilization of nutrients is low, risk of erosion is high, and runoff is likely. Livestock waste may take the form of slurry (semi-liquid feces and urine stored in subsurface tanks, aboveground lagoons, or upright silos) or solid farmyard manure (FYM) stored in bunkers. Farmyard manure includes both urine and feces and bedding material. The most recent Irish survey indicated that 81% of manure is stored as slurry and 19% as FYM (Buckley et al., 2020). Correct storage of urine and feces is required to ensure they do not become sources of nutrient loss from the farmyard. Insufficient storage capacity can elevate risks of losses from the yard. In the United Kingdom and Ireland, storage capacity requirements are specified per head of livestock and with respect to geographic region, with those areas experiencing greater annual rainfall having greater storage requirements (Schulte et al., 2006). Although surplus storage capacity is mandated, during wet years the farmer may still be forced to house livestock for more prolonged periods. The pressure that this exerts on storage capacity is exacerbated where rainfall is allowed to ingress to storage tanks rather than being diverted from clean surfaces away from the yard. A study in Scottish catchments indicated a 44% increase in the volume of slurry to be land spread as a result of water intrusion (Aitken, 2003). Furthermore, >50% of farms in that study were found to have insufficient storage capacity across various storage types. This can result in accidental overflow of tanks, causing a large release of nutrients and potentially leading to serious pollution issues, including fish kills. Lennox et al. (1998) reported that between 11.7 and 41.3% of agricultural pollution incidents between 1987 and 1995 in Northern Ireland resulted from overflowing tanks (including livestock slurry, soiled water, silage effluent, and yard runoff). The acute effects of spills reflect the

TABLE 1 Silage effluent characteristics from the literature

				Nutrient concentrations	
Source		**Dry matter of ensiled grass**	**Avg. effluent vol.**	**P**	**N**
Silage pit					
Arnold et al., 2000				72–563[a]	
Haigh, 1998		16–17%	109–132 L t^{-1}		
Jones & Jones, 1995		17–22%	290–180 L t^{-1}		
Dunlea & Dodds, 1988				14.1[b]	43.3[c]
Laboratory silo					
Keady & O'Kiely, 1998		first cut	16.6%	80 L t^{-1}	
		second cut	17.7%	45 L t^{-1}	
Baled					
Jones & Jones, 1995	one layer		24 L t^{-1}		
	two layers		41 L t^{-1}		
	three layers		45 L t^{-1}		

[a]μg ml^{-1} orthophosphate. [b]g kg^{-1} total P. [c]g kg^{-1} total N.

high BOD of these effluents, which results in rapid oxygen depletion.

Farmyard manure can be stored in concrete bunkers in a farmyard or in field heaps. If stored in bunkers, the effluent generated should be treated as slurry and collected for later land spreading. Although this can further increase the pressure on slurry storage capacity, direct losses from badly constructed or damaged bunkers can also contribute to nutrient export from yards. A study of FYM stored in field heaps observed about 2.5% of total P in FYM lost in effluent (Sommer, 2001). Storage in field heaps as opposed to in farmyard bunkers allows infiltration of leachate into subsoil rather than more direct transfer to watercourses via runoff (Doody et al., 2013; MAFF, 2000).

Direct deposition of urine and feces onto hardstanding occurs when cattle are moving through farmyards or waiting for milking, dosing, loading, and other handling operations. The volume of deposition depends upon the length of time the animal remains on the hardstanding. Likelihood of urination/defecation in the yard area corresponds to length of time spent in the area (Stefanowska et al., 1999; White et al., 2001). Consequently, prolonged milking duration, which can result from insufficient parlor facilities or labor, promotes both longer congregation in the yard and increased deposition of feces and urine.

2.1.2 | Silage effluent

During the ensiling process, soluble carbohydrates within the harvested crop are fermented anaerobically. This lowers pH (target of 3.8–4.2 pH) and thereby inhibits microbial growth and preserves the crop. Effluent is produced as a by-product of ensiling as the cell walls of the crop break down and release their contents, particularly when initial moisture content is high (Gebrehanna et al., 2014). Although a relatively low overall volume of silage effluent is generated, it contains significant concentrations of P and N (Table 1). Most effluent generation occurs within the initial 21 d post-ensiling. Volume of effluent is related to the grass dry matter at the time of ensiling, with minimal effluent produced at >25% dry matter (Dunlea & Dodds, 1988). Overall volumes can be increased by rainfall where pits are not adequately covered. Nutrient contents of silage effluent vary greatly due to factors including the diversity of silage production techniques, grass varieties, and environmental conditions and is summarized in Gebrehanna et al. (2014). Even after generation of effluent has ceased, losses can still occur if it is flushed from the pit or bales by rainfall or due to leaky silo, tanks, and pipes. By law, in many countries silage effluent from pits must be collected using gutters, and baled silage may not be opened adjacent to watercourses. Between 1987 and 1995, silage effluent was responsible for 8.4% of fish kills in Northern Ireland, second only to poultry waste (Lennox et al., 1998), where leaking silos, leaking tanks, and broken pipes accounted for 27.7, 15.9, and 4.2%, respectively, of recorded pollution incidences in Northern Ireland (Lennox et al., 1998), while lack of facilities or overflow of available facilities were responsible for 7 and 23.7%, respectively. Nutrient concentrations from silage effluents reported in the literature are presented in Table 1. From an ecological perspective, silage effluent is particularly impactful due to its high BOD (Friberg et al., 2010), which results from high amounts of readily oxidized organic matter (Dunlea et al., 1989). Values of BOD reported in the literature range widely, reflecting the success of ensiling and variability in ensiled herbages; Gebrehanna et al. (2014) reported 33,800–72,000 mg L^{-1} BOD for grass silages. These values exceed those observed for other farmyard wastes, including dairy washings.

2.1.3 | Soiled wastewater

The definition of what constitutes soiled water, also commonly referred to as "dirty water" (Brewer et al., 1999), varies between political jurisdictions. Soiled water is defined under Irish regulations as runoff water coming from impervious areas, having a BOD <2,500 mg L^{-1} and dry matter <1%, and being stored separately from slurry (DAFM, 2019). In the United Kingdom, it is defined as having BOD <2,000 mg L^{-1}, dry matter <1%, and total N of <0.3 kg m^{-3} (DEFRA, 2009). Brewer et al. (1999) described the composition of soiled water as originating from bulk tank rooms (dairy farms), parlor washings (dairy farms), cattle holding areas adjacent to the parlor (dairy farms), soiled outdoor yards, and silage pits. The contribution of each of these sources to the overall loading of the farmyard varies; however, soiled hardstandings were a common feature in almost all surveyed farms (Brewer et al., 1999). Fouling of this area to some degree is inevitable. As such, soiled water reflects nutrient contributions from a variety of individual sources within the farmyard itself. Regarding fouling by livestock, there is a significant correlation (R^2 = .99) between the proportion of time spent on the hardstanding with the number of livestock defecation events (White et al., 2001). The volume of soiled water produced on farms varies enormously and does not directly correlate with the size of the herd, but other factors including annual rainfall, area of the farmyard, and its layout may also contribute (Brewer et al., 1999).

Whereas the studies of Forbes et al. (2011), Mustafa et al. (2009), and Dunne et al. (2005) reported on soiled water lost from farmyard, Minogue et al. (2015) and Martinez-Suller et al. (2010) measured nutrient contents in wastewater held in farmyard collection tanks (Table 2). Each of these studies evaluates collected soiled water and so may not precisely reflect actual incidental losses to a watercourse. Rather, if nutrient and organic matter–rich material such as manure or urine is collected in the tank, the concentrations observed there will be greater, although concurrent losses to the environment will be reduced. The dry matter (%) of farmyard wastewater has a significant correlation (R^2 = .94) with P concentration (Martinez-Suller et al., 2010). Cumby et al. (1999) similarly found strong correlation between total suspended solids and phosphate (R^2 = .81), Kjeldahl N (R^2 = .79), and biological oxygen demand (R^2 = .87). That study also observed a seasonal effect, with higher concentrations of solids and nutrients during the summer. These studies demonstrate that runoff from the yard may have greater or lesser nutrient contents, depending on the timing of precipitation, partitioning or convergence of overland flow, the presence

TABLE 2 Soiled wastewater characteristics from the literature

Source	Volume	P concentration					N concentration					No. of farms
		TP	PP	TDP	DRP	DOP	NO_3–N	TN	TDN	NH_4–N	DON	
		mg L^{-1}										no.
Edwards & Hooda, 2008		0.13–5.71	0–5.49	0.06–4.64	0.01–4.16	0–1.08	0.40–18.5					1
		0.28–51	0.03–16.2	0.10–34.80	0.05–27.8	0–19	0.45–14.7					
Edwards et al., 2008				0.008–19.50	0.005–19.46	0.003–10.94	0.01–14.28		0.48–637	0.025–562	0.43–248	4
Edwards et al., 2012					0–0.94		0.03–13.8			0.01-31.4		1
Brewer et al., 1999 and Cumby et al., 1999[a]	191–6,080 m^3 yr^{-1}				340–490			700–950		310–580		20
Dunne et al., 2005	1.2–18.5 m^3 d^{-1}				15–21		36–61			36–61 (NH_4^+)		1
Forbes et al., 2011	ave. 31 m^3 d^{-1}	24–131								3.3–15.7		1
Martinez-Suller et al., 2010		21–103			0.07–24.9			128–987		0–106		1
Minogue et al., 2015	9,784 L cow^{-1} yr^{-1}	80			36			587		212		60
Mustafa et al., 2009	180 m^3 mo^{-1}				11.55		3.81			39.62		1

Note. DON, dissolved organic N; DOP, dissolved organic P; DRP, dissolved reactive P; PP, particulate P; TDN, total dissolved N; TDP, total dissolved P; TP, total P.
[a]These studies share participating farms.

of contaminants such as silage effluent, and the implementation/efficacy of yard hygiene routine. Consequently, the potential magnitude of losses is highly dependent upon correct farmyard management.

2.2 | Mobilization

Mobilization refers to the initial element of transfer, during which nutrients are physically detached or dissolved from the source and begin movement along a hydrologic pathway. Rainfall is the primary driver of nutrient mobilization from farmyards. It can be assumed that there is no infiltration from farmyards because they consist of impermeable hardstanding or roofs, with all rainfall running off the concrete if not collected, either into adjacent field areas or into the drainage network. There is a significant relationship between rainfall and runoff volumes from farmyards ($P < .01$ [Dunne et al., 2005]; $P < .001$ [Minogue et al., 2015]); however, the overall dependency of nutrient loss on rainfall is variable, depending on characteristics of the yard itself (Edwards & Withers, 2008). Average annual volume and frequency of rainfall are key risk factors; however, a survey of experts found that these ranked lowest among runoff factors, with delivery factors such as the presence of discharge points and distance to watercourses considered to be of greater importance (Vero et al., 2020). The timing of rainfall relative to soiling of the yard will influence the gross loads of nutrients mobilized. Hardstandings are typically cleaned by scraping, sweeping, or power-washing, with yard washing also a potential mobilization driver if not managed correctly. However, data on losses due to cleaning methods are limited. Minogue et al. (2015) observed that increases in scraping frequency (0–7 times per week) significantly increased the volume of soiled water and the concentration of total N and ammonium collected in storage tanks, though it was not significantly correlated with total P. Hence, infrequent scraping is likely to result in accumulation of waste on the hardstanding, which may be lost to watercourses during rainfall events. In a study of 20 farms in the United Kingdom, no correlation was found between the volume of soiled water production and number of dairy cows (Brewer et al., 1999). In addition to rainfall volume and frequency, factors that also contribute to the volume of soiled water produced may include the area of the farmyard, its position within the landscape, efficiency of milking (equating to time spent in the collecting yard and volumes of washing water used), and calving system (Minogue et al., 2015). Washing of the parlor and collecting yard may also cause mobilization and entrainment of nutrients if those washings are not collected. This was observed at catchment scale in an intensive dairy region during a drought period, during which diurnal spikes in total reactive P indicated that these peaks corresponded with morning and evening milking times (Mellander & Jordan, 2021).

Runoff from farm roofs has been documented to contain highly variable concentrations of N, P, and metals (Edwards et al., 2008). That study reported mean concentrations of 70.4 mg L^{-1} ammonium, 1.24 mg L^{-1} nitrate, 4.73 mg L^{-1} total dissolved P, and 3.9 mg L^{-1} dissolved reactive P; however, there was significant variability both between sites and events. Unlike runoff from the hardstanding, concentrations from roofs rapidly declined as the storms progressed, indicating depletion of limited nutrient sources. These sources reflect atmospheric deposition of N and nutrients arriving via bird droppings (Edwards et al., 2008). These sources are slow to replenish compared with soiling of the hardstanding and so may be of a relatively minor concern by comparison.

2.3 | Delivery

Although artificial land drainage is ubiquitous in Irish and U.K. farmland, it is largely nonstandardized and ad hoc, exhibiting wide variability in open and subsurface channels (O'Hara et al., 2020). Furthermore, farmyards do not adhere to a common design and typically are developed and added to over time. Hence, the connectivity of farmyards to watercourses via this secondary network is idiosyncratic (Moloney et al., 2019), and connections are often unrecorded. Category 1 ditches were defined by Moloney et al. (2019) as those that connect farmyards either to the drainage network or directly to a watercourse. A survey of 10 case study farms indicated 13% of total ditch length was classified as Category 1; however, 3 of the 10 sites were disconnected from the greater ditch network (Moloney et al., 2019). In these cases, overland flow from the farmyard is either captured in storage tanks or infiltrates through the soil in adjacent receiving fields.

Transfer of nutrients via the drainage network occurs during flow periods, and as such, when ditches are dry, they are not contributing to nutrient loads delivered to the watercourse at that time (Shore et al., 2015). This creates a time lag between the transfer of nutrients from the farmyard to the drainage network and their delivery to a waterbody. Subsequent to rainfall events, this lag might be relatively brief because the drainage network is hydrologically connected to the watercourse. However, deposition, attenuation, and resuspension of nutrients along drainage networks is common due to intermittent connectivity (Moloney et al., 2019), and prediction of lag is unreliable. Step changes in hydrochemical parameters in longitudinal surveys have been observed where delivery points from Category 1 ditches intersect with watercourses (Ezzati et al., 2020; Vero et al., 2019). Furthermore, drainage ditches receiving nutrient-rich organic effluents may act as a reservoir of sediment-bound P, which is vulnerable to resuspension subject to temperature, pH, and redox conditions (Aiping et al., 2015; Ezzati et al., 2020). Such systems may

TABLE 3 Nutrient concentrations in ditch water and sediment connected to farmyards

Study	Mean concentrations						
	Water				Sediment		
	DRP	TP	NO_3^-	NH_4^+	WSP	Morgan-P	Mehlich-P
	mg L^{-1}				mg kg^{-1}		
Moloney et al., 2019	0.018–342	0.084–325	0.481–380	0.051–379	1.16–175		15.51–429.5
Harrison et al., 2019	0.019–2.43	0.053–4.68	0–8.68	0.016–14.03			
Ezzati et al., 2020[a]	0.047–0.61	0.031–2.29					13.81

Note. DRP, dissolved reactive P; TP, total P; WSP, water-soluble P.
[a]Sample Point D receiving farmyard discharge, 0-to-30-cm sediment depth.

therefore contribute to P delivery even during periods of relatively low flow.

Grab sampling provides a snapshot of water composition at a specific time and flow period and is not necessarily prescriptive of fluctuations at other times. However, across several studies water samples in Category 1 (farmyard connected) ditches have been documented to contain high dissolved P and N concentrations (Table 3). These concentrations may reflect nutrients arriving at the ditch via runoff at that time (typically during or immediately subsequent to rainfall) or due to mobilization of legacy concentrations accumulated in the ditch sediment (Moloney et al., 2019; Vero et al., 2019). Sediment chemistry (chiefly equilibrium P concentration and Ca/Al/P ratios) within the ditch network and hydrologic connectivity (Ezzati et al., 2020; Moloney et al., 2019; Shore et al., 2015) thereafter control the delivery of nutrients to the watercourse.

Direct runoff to water courses via hydrologically connected overland pathways (e.g., saturated fields) reaching delivery points along the receptor may also provide a route for farmyard wastewater. Little commentary is available on this in the literature specifically regarding farmyard runoff, although it is reasonable to apply the same conceptual understanding of this delivery pathway as any other contaminant transported overland (Bracken et al., 2013). In such scenarios there may be opportunity for infiltration of runoff or attenuation of entrained nutrients along the pathway.

2.4 | Impact

There is a knowledge gap pertaining to the precise impact of farmyard nutrient losses on surface water quality, distinct from other point sources and diffuse losses. Impacts may occur distant and downstream from sources, and it may be challenging to disentangle the effects of multiple sources from signals at a catchment or subcatchment scale. For example, the contribution of individual point sources has been observed through longitudinal sampling to lead to cumulative loading moving from headwater to outlet (Vero et al., 2019) where riverine processes such as dilution or attenuation fail to ameliorate loads (Withers & Jarvie, 2008). Impacts may also be temporally buffered, first by the hydrologic time lag exerted upon the mobilization and transport phases of the NTC. Subsequently, in-stream delays occur as nutrients, particularly P, may be attenuated in sediment and later released (de Klein & Koelmans, 2011; Simpson et al., 2021) or when the effects on macroinvertebrates become manifested during ecologically sensitive periods such as summer (Piggott et al., 2012). Conceptualizing farmyard impacts are not limited to the physical or temporal point of delivery.

Direct monitoring of farmyard losses is relatively limited (compared with runoff and leaching studies at field or catchment scale) in part to wide variability in farmyard design and functioning and technological challenges in instrumentation (Withers & Hodgkinson, 2009). Arguably, greater emphasis on diffuse losses and related mitigation options in both research and policy may also contribute to the comparatively limited information on this area. Furthermore, the lack of a framework that conceptualizes farmyard nutrient dynamics and that can be integrated with catchment-scale models has led to significant uncertainty in the assumptions made. There is a need for further research to accurately quantify nutrient loads and timings from monitored farmyards nested within study catchments. Existing long-term monitored catchments may provide suitable infrastructure to support this. The weight of evidence presented here from across the literature shows that farmyards do contribute significant nutrient loads to watercourses but that their impact is likely to be heterogeneous across enterprises and variable through time.

3 | OPPORTUNITIES FOR MITIGATION

Within the framework presented herein, mitigation measures may affect different stages of the NTC. Although the measures presented here are not new, to date their discussion within the literature has not explicitly defined where they operate within the NCT framework. Farmyard mitigation measures can either reduce the quantity of nutrients generated or divert losses to natural or artificial sinks (e.g., storage tanks, wetland), which may require further management (e.g., application to land). Classification of farmyard mitigation measures based on the

NTC and requirement for further management will help to target measures more cost effectively (Vero et al., 2020). Mitigation options at each of the source, mobilization, and delivery stages of the NTC are discussed hereafter. Mitigation at the impact stage is omitted here in line with the preference for improvement earlier in the NTC. This is not a comprehensive list of all possible mitigation options but rather a commentary on some of the most prominent strategies described throughout the literature and seen in practice.

3.1 | Source mitigation

Preventing overflows of slurry or manure onto the hardstanding and ensuring there is sufficient capacity to collect deposited urine and feces is crucial for mitigating farmyard nutrient sources. Within the United Kingdom and Ireland, minimum storage capacities are specified in legislation on a regional basis, which reflects the likely required duration of winter housing. However, surveys have had limited ability to determine whether the actual storage capacities are adequate on a farm-by-farm basis (Hennessy et al., 2011) because rainfall patterns, water ingress, and opportunities to empty tanks result in dynamic requirements. It has been speculated that many holdings may have insufficient capacity, particularly during of extended housing of livestock (Aitken, 2003; Barnes et al., 2009; Humphreys, 2008). A contributing factor is the ingress of rainfall to tanks, thus diminishing available capacity. In Ireland there has been a positive year-on-year trend toward increased roofing of slurry stores (from 85% in 2016 to 89% in 2018) (Buckley et al., 2020). This may reflect private investment and also the effect of incentives such as the Targeted Agricultural Modernisation Scheme (DAFM, 2016) in the Republic of Ireland. Where roofing is not possible, the UK Code of Good Agricultural Practice proposes the use of float covers for open tanks or lagoons. Although an increase in total storage volume might be suggested as a simple improvement, it may not be technically feasible due to limitations in available land and disruption of existing yard facilities. In addition, this will result in a greater volume of slurry for application to land at a later date. In temperate climates, with a high rainfall frequency and high soil moisture for much of the year, this will increase the risk of diffuse nutrient losses and therefore may not be the most cost-effective solution in some areas. Roofing of existing manure/slurry storage facilities and improved routing of clean water away from storage would maximize available capacity while respecting technical feasibility and costs. However, further research is required to address this issue and identify the most cost-effective solutions for different farm types and locations. Manure and urine deposited on the hardstanding should be collected in manure or slurry stores promptly to reduce the likelihood of rainfall coinciding with a nutrient loaded surface that is primed for losses via runoff. Efficient handling of livestock, particularly during milking, would reduce time spent in the collection area and, consequently, loading with manure and urine. Achieving safe and efficient handling is contingent upon adequate labor and facilities (e.g., sufficient hardstanding, drafting facilities, etc.).

The potency of silage effluent may not be wholly reflected in its nutrient content but is attributed in part to its high BOD. Steps can be taken at each stage of production to minimize effluent generation. Early harvesting and increased artificial N fertilizer application significantly increase effluent production and nutrient concentration (Keady & O'Kiely, 1998). Based on that study, judicious use of fertilizers and aiming for a June rather than a May harvest date reduced effluent volume by 21 g kg^{-1} ensiled and 25 g kg^{-1} ensiled, respectively. Optimal harvesting dates will vary depending on geographic location. Furthermore, wilting of forage for 24–48 h prior to ensiling can help attain target dry matter in order to minimize effluent and maximize silage quality. Additives to prevent the production of effluent during ensiling (O'Kiely, 1992), to absorb it in situ, or to reduce its potency once produced (Arnold et al., 2000) have been trialed with various levels of efficacy. These will represent an additional cost to the farmer, and, as noted by Arnold et al. (2000), technical feasibility must also be accompanied with financial viability for such interventions to be adopted into practice.

To minimize potential losses to water, collection gutters must be maintained around silage pits in order to route effluent to storage. As with manure/slurry stores, roofing of silage pits will significantly reduce rainfall ingress and hence the volume of effluent created and reducing the pressures on slurry storage facilities. Baled silage may contain produced effluent within the wrap, but this can be released during feed-out. The number of layers in which bales are stacked has been significantly correlated with effluent production (one layer, 24 L t^{-1}; two layers, 41 L t^{-1}; and three layers, 45 L t^{-1}) (Jones & Jones, 1995). Ideally, storage in a single layer should be preferred if space permits. If stacking must occur, the use of three layers seems most efficient if it can be achieved safely. Baled silage may also be stored in-field, allowing a 20-m or 10-m distance from the nearest watercourse in the Republic of Ireland and the United Kingdom, respectively. This effectively removes silage effluent from loading of the farmyard, although it may contribute to leached losses.

3.2 | Mobilization mitigation

Eliminating loading of farmyards entirely is not realistic; however, sources can be managed such that they are not mobilized and entrained in the transport continuum. In essence, mobilization mitigation consists of moving available sources to suitable storage areas or otherwise disposing of them safely. The objective is to prevent coincidence of precipitation with periods in which surfaces are soiled in accordance with the critical source area concept. In the absence of roofing, cleaning of the hard surface is key to minimizing mobilization. This may consist of sweeping, scraping, or hosing, although hosing should be minimized in order to reduce the volumes of soiled water to be collected and thereby reducing the demands on storage tanks. Scraping or sweeping may be less effective from uneven or damaged hard surfaces; therefore, the standard of farmyard facilities may have an indirect effect on the ability of the farmer to prevent mobilization. Similarly, good-quality guttering and efficient routing of clean and dirty water within the farmyard will allow more effective management of mobilization by controlling the hydrologic connectivity element. In some instances, this will require financial investment in the upgrading of facilities; however, ensuring clean, unobstructed channels will in many cases be sufficient to yield some improvements. Magette et al. (2007) highlighted the importance of "fatal flaws" in certain farmyards, such as cracked storage facilities, that may bypass or overwhelm the typical transport stage of the NTC by immediately delivering nutrients to a receptor. In such cases, urgent remediation and structural improvement of facilities are required.

3.3 | Delivery mitigation

Delivery to the watercourse allows opportunities for intervention and mitigation depending on the location of farmyards in the landscape. In farm drainage networks, attenuation or mobilization of entrained nutrients is in part controlled by geochemical parameters (Al/P ratio and binding energy [Moloney et al., 2019]). Although the attenuation capacity of drainage channel can be difficult to modify, the addition of additives to increase the retention capacity of ditches has been explored (Penn et al., 2007, 2017). The geometry of ditches can also be used to control the speed of flow and to encourage deposition of sediment and attenuation of nutrients. Slopes ≥5% offer low P retention potential, 2–5% offer moderate potential, and ≤2% offer high fine-sediment retention potential (Shore et al., 2015). Retention of sediment will reduce delivery of attenuated P to watercourses. Existing ditches could be modified to conform to conditions that encourage deposition of sediment and nutrients before they reach the watercourse. Sediments can then be simply excavated and applied to field areas, where nutrients may be effectively used by the growing crop. Dollinger et al. (2015) reported wide variability in nutrient (and other contaminant) mitigation potential in drainage ditches, with 3–92% reduction in loads exiting channels, depending on nutrient species and ditch characteristics. However, the studies reviewed in that paper were from the United States, and evaluation of potential in Ireland and the United Kingdom is a knowledge gap. In-ditch mitigation infrastructure offers potential win-win solutions in moderating nutrient losses en route to receptors with minimal demands on land availability. Various filter and attenuation materials have been trialed at laboratory and field-scale (Arenas-Montaño et al., 2021), many of which may be recycled from waste from other industries and help to attain circularity goals.

Sediment traps and settlement ponds have been primarily used for mitigation on non-point source sediments and associated nutrients (Mekonnen et al., 2015). However, they may offer potential for curtailing entrainment of farmyard nutrient losses within the drainage network. Although use of this technology for mitigation of diffuse sources is optimally placed at a low position in the landscape, where it can capture the entire subcatchment delivering to that point, to be effective in a farmyard context positioning may need to be near the yard–drain junction unless engineered solutions to help redirect flows prove to be cost effective. In theory, Category 1 ditches, as described by Moloney et al. (2019), provide the best location for the installation of mitigation measures.

In contrast, integrated constructed wetlands (ICWs) have received perhaps the most investigation as mitigation measures to offset or reduce delivery of nutrients lost from the farmyard. Nutrient loss is modified via physical (settlement and attenuation in substrate) and biochemical (plant uptake, nitrification, denitrification, and anammox) processes occurring in ICWs subject to optimal design and sufficient hydraulic residence times. Dunne et al. (2005) documented the performance of a 4,265 m^2 ICW (three cells) receiving discharge from a 42-cow dairy unit (surface area, 2,031 m^2). During spring, summer, and autumn, retention rates for soluble reactive P within the wetland were similar (81–84%), though performance was vastly depressed with retention of only 5%. The authors of that study attributed this to high precipitation leading to reduced retention times. Conversely, in another study also in the southeast of Ireland, a similar ICW was reported to achieve a >90% removal efficiency for both molybdate reactive P and ammonia-N in wastewater from a 77-cow dairy unit (Mustafa et al., 2009). Forbes et al. (2011) report reductions of 95 and 93% in P and N, respectively, from a five-pond ICW located on 1.2 ha of land. Although the reduction observed in these studies is significant, the uptake of ICW as a cost-effective measure has been slow due to the removal of agricultural land from production and regulatory issues (Everard et al., 2012). There may also be opportunities for bioremediation using crops such as willow (Curneen & Gill, 2014; Forbes et al., 2017) irrigated with soiled water. This offers potential both to offset farmyard nutrient impacts and to simultaneously contribute to meeting our sequestration goals and providing an alternative farm income, where suitable supply routes to the consumer are in place (Styles et al., 2008).

4 | KNOWLEDGE GAPS AND FUTURE NEEDS

This review has identified several knowledge gaps, chief of which is the overall contribution of farmyards to P and N losses to watercourses at catchment scale. Quantification of this will allow improved the cost-effective targeting of mitigation measures and allocation of resources. At present, the farmyard component of nutrient export models is coarsely defined and often treated as a black box. Although upstream and downstream comparisons are useful, they are influenced by dilution and other inputs that can obscure farmyard contributions. Measurement of flow from yards is particularly difficult and has added a degree of uncertainty to estimation of loads (Edwards et al., 2008) and partitioning of sources. This is logistically challenging but is essential in order to capture the variability in farmyard design, use, and management and the effects of different precipitation regimes. On-farm monitoring at high temporal resolution at multiple sites would allow this limitation to be overcome.

The impact of farmyard runoff on watercourses is not restricted to eutrophication driven by N and P content. High BOD as a result of dissolved organic carbon (DOC) is commonly responsible for acute oxygen depletion in watercourses (Penn et al., 2009) and consequent fish kills and reduced macroinvertebrate populations (Harrison et al., 2019). Indeed, elevated BOD has been referred to as "the most obvious sign of pollution" (Hooda, Moynagh, et al., 2000) observed downstream of farmyards discharging organic material. To date, DOC has not been explicitly framed within the NTC conceptual model, although the framework could be valid considering that losses of DOC occur through the same hydrologic pathways (Lambert et al., 2011) and share many of the same sources as N and P (Edwards & Hooda, 2008; Edwards et al., 2008). Although considerable attention has focused on export from peat soils or catchments dominated by forestry, longitudinal patterns of organic matter across stream networks in lowland catchments have been observed to reflect land use and connectivity (Yates et al., 2016). Application of the NTC framework to DOC should be explored in future research. With reference to farmyard export, DOC in hardstanding runoff has been observed to range between 82 and 671 mg L^{-1} ($n = 4$) (Edwards et al., 2008) and downstream BOD to increase significantly (Edwards & Hooda, 2008; Harrison et al., 2019). Hooda et al. (2000) reported significant to severe impacts of farmyards on stream ecology, with a depletion of all but the most pollution-tolerant macroinvertebrates in response to discharge of silage effluent. These effects were attributed primarily to high BOD leading to oxygen depletion. Although the focus of the present study is on N and P from a WFD reporting perspective, the potential impacts of other stressors (oxygen depletion [Calapez et al., 2017], sediment [Davis et al., 2018], etc.) on stream ecology may be pronounced.

Further work is needed to evaluate the cost-effectiveness and suitability of mitigation measures, particularly regarding trade-offs and optimization of land use (McDowell & Nash, 2012). Although significant research has been conducted into individual measures, a comparative analysis of their relative efficacies and suitability for distinct scenarios would allow farmers and advisors to identify which solutions will be most successful. The development of a physically based decision support tool to help inform farmyard management would be a valuable resource but requires quantification of the relative contribution of different sources and pathways of nutrient loss. Such a tool could be designed to address multiple contaminants (e.g., nutrients, pesticides, pharmaceuticals) and account for the timing and type of contaminant loss, the most viable stage of the NTC at which to intervene, the efficacy of the measure (in terms of load reduction), the cost of the measure, and the land use requirements. These measures will include both infrastructural and management approaches (Vero et al., 2020).

Based upon modeling of farmyard losses at national scale followed by validation based upon on-farm monitoring and assessment of mitigation needs, estimates of financial requirements for improved farmyard infrastructure and effluent management facilities could be conducted. This will help to inform future policy decisions and identify needs for further industry and/or government investment to reduce the impact of farmyards on waterbodies.

5 | CONCLUDING REMARKS

This paper has presented the nutrient transfer continuum in the context of livestock farmyards. Other contaminants, including fecal indicator organisms, N, high BOD effluents, biocides, and pharmaceutical residues, have also been attributed to farmyards, and a similar framework could be described in each instance. Identifying mechanisms of nutrient loss from farmyards will improve the cost-effective targeting of mitigation measures and provide a structure for decision-making. Such analyses and decision support systems are an area for future research.

ACKNOWLEDGMENTS

This work was funded by the CatchmentCARE project (project reference IVA5058; www.catchmentcare.eu). The CatchmentCARE project is supported by the European Union's INTERREG VA Programme, managed by the Special EU Programmes Body (SEUPB). The views and opinions expressed in this document do not necessarily reflect those of the European Commission or the SEUPB. The authors thank Russell McCune for providing the graphical abstract illustration.

AUTHOR CONTRIBUTIONS

Sara E. Vero: Conceptualization; Investigation; Methodology; Writing-original draft; Writing-review & editing. Donnacha Doody: Conceptualization; Investigation; Methodology; Project administration; Resources; Supervision; Writing-review & editing.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

ORCID

Sara E. Vero https://orcid.org/0000-0001-6562-5173
Donnacha Doody https://orcid.org/0000-0003-2361-2504

REFERENCES

Aiping, T., Jinbao, W., Rong, W., Shuang, L., & Hongyan, S. (2015). Importance of pH, dissolved oxygen and light to phosphorus release from ditch sediments. *Nature Environment and Pollution Technology*, *14*(3), 475–484.

Aitken, M. N. (2003). Impact of agricultural practices and river catchment characteristics on river and bathing water quality. *Water Science and Technology*, *10*, 217–224. https://doi.org/10.2166/wst.2003.0577

Arenas-Montaño, V., Fenton, O., Moore, B., & Healy, M. G. (2021). Evaluation of the fertiliser replacement value of phosphate-saturated filter media. *Journal of Cleaner Production*, *291*, 125943. https://doi.org/10.1016/j.jclepro.2021.125943

Arnold, J. L., Knapp, J. S., & Johnson, C. L. (2000). The use of yeasts to reduce the polluting potential of silage effluent. *Water Resources*, *34*(15), 3699–3708.

Barnes, A. P., Willock, J., Hall, C., & Toma, L. (2009). Farmer perspectives and practices regarding water pollution control programmes in Scotland. *Agricultural Water Management*, *96*(12), 1715–1722. https://doi.org/10.1016/j.agwat.2009.07.002

Bracken, L. J., Wainwright, J., Ali, G. A., Tetzlaff, D., Smith, M. W., Reaney, S. M., & Roy, A. G. (2013). Concepts of hydrologic connectivity: Research approaches, pathways and future agendas. *Earth-Science Reviews*, *119*, 17–34. https://doi.org/10.1016/j.earscirev.2013.02.001

Brewer, A. J., Cumby, T. R., & Dimmock, S. J. (1999). Dirty water from dairy farms, II: Treatment and disposal options. *Bioresource Technology*, *67*, 161–169. https://doi.org/10.1016/S0960-8524(98)00105-9

Buckley, C., Moran, B., & Donnellan, T. (2020). *Teagasc National Farm Survey: A report on bovine manure management, application and storage practices in Ireland*. Teagasc.

Cai, K., Phillips, D. H., Elliott, C. T., Muller, M., Scippo, M.-L., & Connolly, L. (2013). Removal of natural hormones in dairy farm wastewater using reactive and sorptive materials. *Science of the Total Environment*, *461–462*, 1–9. https://doi.org/10.1016/j.scitotenv.2013.04.088

Calapez, A. R., Branco, P., Santos, J. M., Ferreira, T., Hein, T., Brito, A. G., & Feio, M. J. (2017). Macroinvertebrate short-term responses to flow variation and oxygen depletion: A mesocosm approach. *Science of the Total Environment*, *599–600*, 1202–1212. https://doi.org/10.1016/j.scitotenv.2017.05.056

Crump, A., Jenkins, K., Bethell, E. J., Ferris, C. P., & Arnott, G. (2019). Pasture access affects behavioural indicators of wellbeing in dairy cows. *Animals*, *9*, 902. https://doi.org/10.3390/ani9110902

Cumby, T. R., Brewer, A. J., & Dimmock, S. J. (1999). Dirty water on dairy farms, I: Biochemical characteristics. *Bioresource Technology*, *67*, 155–160. https://doi.org/10.1016/S0960-8524(98)00104-7

Curneen, S. J., & Gill, L. W. (2014). A comparison of the suitability of different willow varieties to treat on-site wastewater effluent in an Irish climate. *Journal of Environmental Management*, *133*, 153–161. https://doi.org/10.1016/j.jenvman.2013.12.004

Department of Agriculture, Food and the Marine (DAFM). (2016). *Targeted farm modernisation scheme*. https://www.gov.ie/en/publication/9e8f1-tams-ii-support-documents/

Department of Agriculture, Food and the Marine (DAFM). (2019). *Nitrates exploratory handbook*. DAFM.

Davis, S. J., Ó Huallacháin, D., Mellander, P.-E., Kelly, A.-M., Matthaei, C. D., Piggott, J. J., & Kelly-Quinn, M. (2018). Multiple-stressor effects of sediment, phosphorus and nitrogen on stream macroinvertebrate communities. *Science of the Total Environment*, *337–638*, 577–587. https://doi.org/10.1016/j.scitotenv.2018.05.052

De Klein, J. J. M., & Koelmans, A. A. (2011). Quantifying seasonal export and retention of nutrients in West European lowland rivers at catchment scale. *Hydrological Processes*, *25*(13), 2102–2111. https://doi.org/10.1002/hyp.7964

Department of Environment, Food and Rural Affairs. (2009). *Protecting our water, soil and air. Code of good agricultural practice for farmers, growers, and land managers*. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/268691/pb13558-cogap-131223.pdf

Dollinger, J., Dagès, C., Bailly, J.-S., Lagacherie, P., & Voltz, M. (2015). Managing ditches for agroecological engineering of landscape: A review. *Agronomy for Sustainable Development*, *35*, 999–1020. https://doi.org/10.1007/s13593-015-0301-6

Doody, D. G., Bailey, J. S., & Watson, C. J. (2013). Evaluating the evidence-base for the Nitrate Directive regulations controlling the storage of manure in field heaps. *Environmental Science and Policy*, *29*, 137–146. https://doi.org/10.1016/j.envsci.2013.01.009

Dunlea, A. P., & Dodd, V. A. (1988). The application of membrane filtration to silage effluent. *Canadian Agricultural Engineering*.

Dunlea, A. P., Dodd, V. A., & McKenna, B. M. (1989). The Application of membrane filtration to silage effluent. In R. P. Singh & A. G. Medina (Eds.), *Food properties and computer-aided engineering of food processing systems* (NATO ASI Series, Series E: Applied Sciences, Vol. 168). Springer. https://doi.org/10.1007/978-94-009-2370-6_35

Dunne, E. J., Culleton, N., O'Donovan, G., Harrington, R., & Olsen, A. E. (2005). An integrated constructed wetland to treat contaminants and nutrients from farmyard dirty water. *Ecological Engineering*, *24*, 221–234. https://doi.org/10.1016/j.ecoleng.2004.11.010

Edwards, A. C., & Hooda, P. S. (2008). Farmyard point discharges and their influence on nutrient and labile carbon dynamics in a second order stream draining through a dairy unit. *Journal of Environmental Management*, *87*, 591–599. https://doi.org/10.1016/j.jenvman.2007.02.013

Edwards, A. C., & Withers, P. J. A. (2008). Transport and delivery of suspended solids, nitrogen and phosphorus from various sources to freshwaters in the UK. *Journal of Hydrology*, *350*, 144–153. https://doi.org/10.1016/j.jhydrol.2007.10.053

Edwards, A. C., Kay, D., Mcdonald, A. T., Francis, C., Watkins, J., Wilkinson, J. R., & Wyer, M. D. (2008). Farmyards, an overlooked source for highly contaminated runoff. *Journal of Environmental*

Management, *87*, 551–559. https://doi.org/10.1016/j.jenvman.2006.06.027

Edwards, A. C., Watson, H. A., & Cook, Y. E. M. (2012). Source strengths, transport pathways and delivery mechanisms of nutrients, suspended solids and coliforms within a small agricultural headwater catchment. *Science of the Total Environment*, *434*, 123–129. https://doi.org/10.1016/j.scitotenv.2011.10.055

Everard, M., Harrington, R., & Mcinnes, R. J., (2012). Facilitating implementation of landscape-scale water management: The integrated constructed wetland concept. *Ecosystem Services*, *2*, 27–37. https://doi.org/10.1016/j.ecoser.2012.08.001

European Commission. (1991). *Council Directive (1991/676/EC). EC Directive of the European Parliament and of the Council of 12 December 1991 on the protection of waters against pollution caused by nitrates from agricultural sources*. European Commission.

Ezzati, G., Fenton, O., Healy, M. G., Christianson, L., Feyereisen, G. W., Thornton, S., Chen, Q., Fan, B., Ding, J., & Daly, K. (2020). Impact of P inputs on source-sink P dynamics of sediment along an agricultural ditch network. *Journal of Environmental Management*, *257*, 109988. https://doi.org/10.1016/j.jenvman.2019.109988

Forbes, E. G. A., Foy, R. H., Mulholland, M. V., & Brettell, J. L., (2011). Performance of a constructed wetland for treating farm-yard dirty water. *Water Science and Technology*, *64*(1), 22–28. https://doi.org/10.2166/wst.2011.584

Forbes, E. G. A., Johnston, C. R., Archer, J. E., & Mccracken, A. R. (2017). SRC willow as a bioremediation medium for a dairy farm effluent with a high pollution potential. *Biomass and Bioenergy*, *105*(2), 174–179. https://doi.org/10.1016/j.biombioe.2017.06.019

Friberg, N., Skriver, J., Larsen, S. E., Pedersen, M. L., & Buffagni, A. (2010). Stream macroinvertebrate occurrence along gradients in organic pollution and eutrophication. *Freshwater Biology*, *55*, 1405–1419. https://doi.org/10.1111/j.1365-2427.2008.02164.x

Gebrehanna, M. M., Gordon, R. J., Madani, A., Vanderzaag, A. C., & Wood, J. D. (2014). Silage effluent management: A review. *Journal of Environmental Management*, *143*, 113–122. https://doi.org/10.1016/j.jenvman.2014.04.012

Haigh, P. M. (1998). Effect of additives on grass silage fermentation and effluent production, and on intake and liveweight change of young cattle. *Journal of Agricultural Engineering Research*, *69*(2), 141–148. https://doi.org/10.1006/jaer.1997.0240

Harrison, S., Mcaree, C., Mulville, W., & Sullivan, T. (2019). The problem of agricultural 'diffuse' pollution. *Science of the Total Environment*, *677*, 700–717. https://doi.org/10.1016/j.scitotenv.2019.04.169

Haygarth, P. M., Condron, L. M., Heathwaite, A. L., Turner, B. L., & Harris, G. P. (2005). The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. *Science of the Total Environment*, *344*, 5–14. https://doi.org/10.1016/j.scitotenv.2005.02.001

Hennessy, T., Buckley, C., Cushion, M., Kinsella, A., & Moran, B. (2011). *National farm survey of manure application and storage practices on Irish farms*. Teagasc.

Holden, J., Haygarth, P. M., Dunn, N., Harris, J., Harris, R. C., Humble, A., Jenkins, A., Macdonald, J., Mcgonigle, D. F., Meacham, T., Orr, H. G., Pearson, P. L., Ross, M., Sapiets, A., & Benton, T. (2017). Water quality and UK agriculture: Challenges and opportunities. *WIREs Water*, *4*, e1201. https://doi.org/10.1002/wat2.1201

Hooda, P. S., Edwards, A. C., Anderson, H. A., & Miller, A., (2000). A review of water quality concerns in livestock farming areas. *Science of the Total Environment*, *250*(1–3), 143–167. https://doi.org/10.1016/S0048-9697(00)00373-9

Hooda, P. S., Moynagh, M., Svoboda, I. F., & Miller, A. (2009). Macroinvertebrates as bioindicators of water pollution in streams draining dairy farming catchments. *Chemistry and Ecology*, *17*(1), 17–30. https://doi.org/10.1080/02757540008037658

Hughes, K. J., Magette, W. L., & Kurz, I. (2005). Identifying critical source areas for phosphorus loss in Ireland using field and catchment scale ranking schemes. *Journal of Hydrology*, *304*(1–4), 430–445. https://doi.org/10.1016/j.jhydrol.2004.07.042

Humphreys, J. (2008). Nutrient issues on Irish farms and solutions to lower losses. *International Journal of Dairy Technology*, *61*(1), 36–42. https://doi.org/10.1111/j.1471-0307.2008.00372.x

Jones, D. I. H., & Jones, R. (1995). The effect of crop characteristics and ensiling methodology on grass silage effluent production. *Journal of Agricultural Engineering Research*, *60*, 73–81. https://doi.org/10.1006/jaer.1995.1001

Lambert, T., Pierson-Wickmann, A.-C., Gruau, G., Thibault, J.-N., & Jaffrezic, A. (2011). Carbon isotopes as tracers of dissolved organic carbon sources and water pathways in headwater catchments. *Journal of Hydrology*, *402*(3–4), 228–238. https://doi.org/10.1016/j.jhydrol.2011.03.014

Lennox, S. D., Foy, R. H., Smith, R. V., Unsworth, E. F., & Smyth, D. R. (1998). A comparison of agricultural water pollution incidents in Northern Ireland with those in England and Wales. *Water Resources*, *32*(3), 649–656.

Keady, & O'Kiely. (1998). An evaluation of potassium and nitrogen fertilization of grassland, and date of harvest, on fermentation, effluent production, dry-matter recovery and predicted feeding value of silage. *Grass and Forage Science*, *53*, 326–337. https://doi.org/10.1046/j.1365-2494.1998.00136.x

Magette, W. L., Hallissey, R., Hughes, K., & Cosgrove, E. (2007). *Eutrophication from agricultural sources: Field- and catchment-scale risk assessment*. Environmental Protection Agency.

Martinez-Suller, L., Provolo, G., Carton, O. T., Brennan, D., Kirwan, L., & Richards, K. G. (2010). The composition of dirty water on dairy farms in Ireland. *Irish Journal of Agri-Food Research*, *49*, 67–80.

McDowell, R. W., & Nash, D. (2012). A review of the cost-effectiveness and suitability of mitigation strategies to prevent phosphorus loss from dairy farms in New Zealand and Australia. *Journal of Environmental Quality*, *41*(3), 680–693. https://doi.org/10.2134/jeq2011.0041

Mekonnen, M., Keesstra, S. D., Stroosnijder, L., Baartman, J. E. M., & Maroulis, J. (2015). Soil conservation through sediment trapping: A review. *Land Degradation and Development*, *26*, 544–556. https://doi.org/10.1002/ldr.2308

Mellander, P.-E., & Jordan, P. (2021). Charting a perfect storm of water quality pressures. *Science of the Total Environment*, *787*, 147576. https://doi.org/10.1016/j.scitotenv.2021.147576

Ministry for Agriculture, Food and Fisheries (MAFF). (2000). *Environmental impacts of baled silage* (Final project report, WA 0111). MAFF.

Minogue, D., French, P., Bolger, T., & Murphy, P. N. C. (2015). Characterisation of dairy soiled water in a survey of 60 Irish dairy farms. *Irish Journal of Agricultural and Food Research*, *54*(1), 1–15. https://doi.org/10.1515/ijafr-2015-0001

Misselbrook, T. H., Webb, J., Chadwick, D. R., Ellis, S., & Pain, B. F. (2001). Gaseous emissions from outdoor concrete yards used by livestock. *Atmospheric Environment*, *35*, 5331–5338. https://doi.org/10.1016/S1352-2310(01)00289-8

Moloney, T., Fenton, O., & Daly, K. (2019). Ranking connectivity risk for phosphorus loss along agricultural drainage ditches. *Science of the Total Environment*, *703*, 134556.

Murphy, P. N. C., Mellander, P.-E., Melland, A. R., Buckley, C., Shore, M., Shortle, G., Wall, D. P., Treacy, M., Shine, O., Mechan, S., & Jordan, P. (2015). Variable response to phosphorus mitigation measures across the nutrient transfer continuum in a dairy grassland catchment. *Agriculture, Ecosystems and Environment*, *207*, 192–202. https://doi.org/10.1016/j.agee.2015.04.008

Mustafa, A., Scholz, M., Harrington, R., & Carroll, P. (2009). Long-term performance of a representative integrated constructed wetland treating farmyard runoff. *Ecological Engineering*, *35*, 779–790. https://doi.org/10.1016/j.ecoleng.2008.12.008

Novotny, V. (1999). Diffuse pollution from agriculture: A worldwide outlook. *Water Science Technology*, *39*(3), 1–13. https://doi.org/10.2166/wst.1999.0124

Office of the Attorney General. (2017). *S.I. 605. European Union good agricultural practice for protection of waters regulations 2017*. http://www.irishstatutebook.ie/eli/2017/si/605/made/en/print

O'Hara, R. (2020). *The application of Earth Observation for mapping soil saturation and the extent and distribution of artificial drainage on Irish farms* [Master's thesis]. Maynooth University.

O'Kiely, P. (1992). The effects of ensiling sugarbeet pulp with grass on silage composition, effluent production and animal performance. *Irish Journal of Agricultural and Food Research*, *31*(2), 115–128.

Penn, C. J., Bryant, R. B., Kleinman, P. J., & Allen, A. L., (2007). Removing dissolved phosphorus from drainage ditch water with phosphorus sorbing materials. *Journal of Soil and Water Conservation*, *62*(4), 269–276.

Penn, C., Chagas, I., Klimeski, A., & Lyngsie, G. (2017). A review of phosphorus removal structures: How to assess and compare their performance. *Water*, *9*(8), 583. https://doi.org/10.3390/w9080583

Penn, M. R., Pauer, J. J., & Mihelcic, J. R. (2009). *Biochemical oxygen demand.* In *Environmental and ecological chemistry: Vol. II. Encyclopedia of life support systems* (pp. 278–297). EOLSS Publishers.

Piggott, J. J., Lange, K., Townsend, C. R., & Matthaei, C. D. (2012). Multiple stressors on agricultural streams: A mesocosm study of interactions among raised water temperature, sediment addition and nutrient enrichment. *PLOS ONE*, *7*(11), e49873. https://doi.org/10.1371/journal.pone.0049873

Schoumans, O. F., Chardon, W. J., Bechmann, M. E., Gascuel-Odoux, C., Hofman, G., Kronvang, B., Rubæk, G. H., Ulén, B., & Dorioz, J.-M., (2014). Mitigation options to reduce phosphorus losses from the agricultural sector and improve surface water quality: A review. *Science of the Total Environment*, *468*, 1255–1266. https://doi.org/10.1016/j.scitotenv.2013.08.061

Schulte, R. P. O., Richards, K., Daly, K., Kurz, I., McDonald, E. J., & Holden, N. M., August. (2006). Agriculture, meteorology and water quality in Ireland: A regional evaluation of pressures and pathways of nutrient loss to water. In *Biology and environment: Proceedings of the Royal Irish Academy* (pp. 117–133). Royal Irish Academy.

Shore, M., Jordan, P., Mellander, P.-E., Kelly-Quinn, M., & Melland, A. R. (2015). An agricultural drainage channel classification system for phosphorus management. *Agriculture, Ecosystems and Environment*, *199*, 207–215. https://doi.org/10.1016/j.agee.2014.09.003

Simpson, Z. P., Mcdowell, R. W., Condron, L. M., Mcdaniel, M. D., Jarvie, H. P., & Abell, J. M. (2021). Sediment phosphorus buffering in streams at baseflow: A meta-analysis. *Journal of Environmental Quality*, *50*(2), 287–311. https://doi.org/10.1002/jeq2.20202

Sinclair, C. J., Ramwell, C. T., Turnbull, G., Bryning, G., Lynn, R., Franey-Gardner, M., Pepper, T., Fogg, L., & Boxall, A. B. (2007). *Assessment and management of inputs of veterinary medicines from the farmyard*. Central Science Laboratory Project M5EU.

Sommer, S. G. (2001). Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. *European Journal of Agronomy*, *14*, 123–133. https://doi.org/10.1016/S1161-0301(00)00087-3

Stefanowska, J., Ipema, A. H., & Hendriks, M. M. W. B. (1999). The behaviour of dairy cows in an automatic milking system where selection for milking takes place in the milking stalls. *Applied Animal Behaviour Science*, *62*, 99–114. https://doi.org/10.1016/S0168-1591(98)00229-9

Styles, D., Thorne, F., & Jones, M. B. (2008). Energy crops in Ireland: An economic comparison of willow and Miscanthus production with conventional farming systems. *Biomass and Bioenergy*, *32*(5), 407–421. https://doi.org/10.1016/j.biombioe.2007.10.012

Tunney, H., Coulter, B., Daly, K., Kurz, I., Coxon, C., Jeffrey, D., Mills, P., Kiely, G., & Morgan, G. (2000). *Quantification of phosphorus loss from soil to water*. End of Project Report. ARMIS 5365.

Van Den Pol-Van Dasselaar, A., Hennessy, D., & Isselstein, J. (2020). Grazing of dairy cows in Europe: An in-depth analysis based on the perception of grassland experts. *Sustainability*, *12*, 1098. https://doi.org/10.3390/su12031098

Vero, S. E., Daly, K., Mcdonald, N. T., Leach, S., Sherriff, S. C., & Mellander, P.-E. (2019). Sources and mechanisms of low-flow river phosphorus elevations: A repeated synoptic survey approach. *Water*, 11, 1497. https://doi.org/10.3390/w11071497

Vero, S. E., Sherry, E., & Doody, D. (2020). Evidence and perception of phosphorus loss risk factors in farmyards. *Environmental Science and Policy*, *114*, 542–548. https://doi.org/10.1016/j.envsci.2020.09.025

Wall, D., Jordan, P., Melland, A. R., Mellander, P.-E., Buckley, C., Reaney, S. M., & Shortle, G. (2011). Using the nutrient transfer continuum concept to evaluate the European Union Nitrates Directive National Action Programme. *Environmental Science and Policy*, *14*(6), 664–674. https://doi.org/10.1016/j.envsci.2011.05.003

White, S. L., Sheffield, R. E., Washburn, S. P., King, L. D., & Green, J. T. (2001). Spatial and time distribution of dairy cattle excreta in an intensive pasture system. *Journal of Environmental Quality*, *30*, 2180–2187. https://doi.org/10.2134/jeq2001.2180

Withers, P. J. A., & Hodgkinson, R. A. (2009). The effect of farming practices on phosphorus transfer to a headwater stream in England. *Agriculture, Ecosystems and Environment*, *131*(3–4), 347–355. https://doi.org/10.1016/j.agee.2009.02.009

Withers, P. J. A., & Jarvie, H. P. (2008). Delivery and cycling of phosphorus in rivers: A review. *Science of the Total Environment*, *400*, 379–395. https://doi.org/10.1016/j.scitotenv.2008.08.002

Yates, C. A., Johnes, P. J., & Spencer, R. G. M. (2016). Assessing the drivers of dissolved organic matter export from two contrasting lowland catchments, U.K. *Science of the Total Environment*, *569–570*, 1330–1340. https://doi.org/10.1016/j.scitotenv.2016.06.211

How to cite this article: Vero, S. E., & Doody, D. (2021). Applying the nutrient transfer continuum framework to phosphorus and nitrogen losses from livestock farmyards to watercourses. *J Environ Qual*. *50:*1290–1302. https://doi.org/10.1002/jeq2.20285