OLLSCOIL NA GAILLIMHE
UNIVERSITY OF GALWAY

NUTRIENT LOSS FROM POORLY DRAINED GRASSLAND SOILS AND LAND DRAINAGE SYSTEMS, AND THE POTENTIAL FOR LOSS MITIGATION

by

Daniel Gyamfi Opoku, B.Sc., PGDip., MEnvMgmt

Thesis submitted to the Department of Civil Engineering, College of Science and Engineering, University of Galway, in fulfilment of the requirements for the Degree of Doctor of Philosophy.

September 2025

Research Supervisors:

Prof. Mark G. Healy, Civil Engineering, University of Galway, Co. Galway

Dr. Patrick Tuohy, Teagasc, Moorepark, Co. Cork

Prof. Owen Fenton, Teagasc, Johnstown Castle, Co. Wexford

Declaration

I, the undersigned, hereby declare that this thesis, entitled, '*Nutrient loss from poorly drained soils and land drainage systems, and the potential for loss mitigation',* is entirely my own work. The thesis has not been submitted in whole or in part to any other University or Institution. All sources used have been acknowledged and referenced in the text.

Daniel Gyamfi Opoku

Acknowledgements

First and foremost, I give my utmost thanks to God, whose grace and guidance have sustained me throughout the moments of doubt, exhaustion and uncertainty in this PhD journey. I owe all my success to Him, from favouring me to gain the scholarship to this PhD and to seeing it to a perfect end. I am truly grateful for the unwavering blessings and love He has shown me.

I would like to express my sincere gratitude and deepest appreciation to my University of Galway supervisor, Prof. Mark G. Healy and Teagasc supervisors, Dr. Patrick Tuohy and Prof. Owen Fenton for their continuous expert guidance and advice, constant encouragement and assistance, insightful feedback and invaluable time. These efforts as well as supplying all required resources for the experiment ensured the high quality of this work.

The path to earning a PhD is demanding and challenging, and I would like to express my family, especially my mother, Mrs. Rita Gyekye and my father, Mr. Samuel Opoku Gyamfi for their unwavering love, support, encouragement, care and understanding. My parents, although far away in Ghana, always provided a listening ear and offered advice on how handle the rough situations, contributing to this successful PhD journey. I am grateful to my grandmother who encouraged me, as the first grandson, to achieve high heights to set the pace for the other grandchildren. I would especially like to thank Wendy for her constant support, patience and encouragement during my PhD, and the endless confidence in my ability which urged me on during my rough days. To all my friends, especially in Teagasc and Cork, that kept me going throughout the low moments, I am most grateful for all the support shown. To Dr. Effah, I appreciate all the statistical expertise you provided to support me in this PhD.

Many thanks to Teagasc for providing the Walsh Scholarship and the funding to carry out this work. I extend my sincere appreciation to the Teagasc Heavy Soils project farmers whose farms I conducted this work. Finally, I wish to thank the Teagasc technical staff, Tomas Condon and

Asaf Shnel, for always giving a hand in carrying out this work, and to the Teagasc-Johnstown castle water laboratory members, especially Denis and Jessyca, for their assistance in water quality analysis.

Abstract

The high rainfall and low evapotranspiration of Ireland's temperate climate cause water-saturated root zones that deplete soil oxygen for root growth and promote unhealthy crop growth for pastures on poorly drained soils. Wet soils are also subject to damage by machinery or animal traffic. These factors limit the full potential of such soils in producing optimal grass for animal production, therefore necessitating the need for the installation of drainage systems. Within drainage systems, networks of surface drains (i.e., open artificial ditches and natural drains) are linked to drain excess water from subsurface (in-field) drains and surface runoff, thereby enhancing grass production and reducing adverse field trafficability conditions in poorly drained soils.

Surface drains transport nutrients from varied surface and subsurface hydrological connectivity pathways to receiving water sources. However, to date, assessment of the connectivity risk of surface drains in transporting nutrients to receiving waters has only assessed phosphorus (P) loss, neglecting nitrogen (N), and has not considered the varying risks from connecting hydrological pathways such as surface roadways and subsurface drains, springs, upwelling and seepage. In addition, the identification of surface drains which pose a high-risk to receiving waters, as well as the contributing factors to these risks (landscape nutrient content, vicinity to farmyards, etc.), which vary spatially along the nutrient transfer continuum (NTC), has remained unexplored. Lastly, farm roadway runoff is an identified nutrient and sediment contributor to connecting surface drains. Yet, farmers' willingness to implement recommended mitigation measures such as swales, sediment ponds and bunded drains, among others, is limited as these measures have not been widely tested for efficiency.

To address these knowledge gaps, the aims of this study were to: (1) create an integrated connectivity risk ranking for surface drains considering P and N simultaneously, (2) develop a semi-quantitative risk model to identify high-risk surface drains, and (3) assess the efficiency

of one mitigation measure (sedimentation ponds) for roadway runoff. Across seven dairy farms, surface drains were mapped, assessed for hydrological connectivity pathway nutrient losses and reclassified to create an integrated N and P loss connectivity risk ranking for surface drains. A semi-quantitative risk assessment model identified high-risk surface drains for targeted mitigation. At three locations where farm roadway runoff was connected to surface drains, three different configurations of sediment ponds were designed and operated for a period of 6 months, to remove nutrients and sediments.

Farmyard-connected drains were ranked as the riskiest due to connectivity to point source losses, whereas outlet drains had the highest risk across surface drains with diffuse sources connectivity. In surface drains associated with diffuse sources, nitrate was introduced by subsurface sources (in-field drains and groundwater interactions from springs, seepage, and upwelling) and ammonium was introduced through surface connectivity pathways (runoff from internal roadways). This study classified 23 %, 68 %, 9 % and 0 % of all surface drains across all farms studied as low, medium, high, or very high-risk class, respectively, with high or above requiring a mitigation plan. Two-thirds of high-risk surface drains were connected to farmyards, with a potential for high nutrient loss from point sources, while other factors including hydrological connectivity pathways from farm roadways contributed to the remaining one-third. A combined source management and targeted mitigation approach is recommended for high-risk or above classes. The study showed sediment ponds are efficient for reducing roadway runoff pollution to surface drains especially for removing total suspended solids and particulate nutrients but vary in their effectiveness in removing dissolved nutrients. Sediment ponds designed to incorporate segmentation, considering all site conditions and containing vegetation, may enhance nutrient and sediment removal. This may facilitate uptake from farmers. The study recommends long-term monitoring to inform maintenance procedures and scheduling.

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Nomenclature

APHA	American Public Health Association
CSA	Critical Source Area
DHLGH	Department of Housing, Local Government and Heritage
DON	Dissolved Organic Nitrogen
DRP	Dissolved Reactive Phosphorus
DUP	Dissolved Unreactive Phosphorus
EC	European Commission
EEA	European Economic Area
EEC	European Economic Community
EIP	European Innovation Partnership
EPA	Environmental Protection Agency
EU	European Union
GIS	Geographic Information System
ha	Hectare
kg	Kilogram
LOD	Limit of Detection
N	Nitrogen
N_2O	Nitrous Oxide
NAP	Nitrates Action Programme
NH_4^+-N	Ammonium as Nitrogen
NO	Nitric Oxide
NO_2^--N	Nitrite as Nitrogen
NO_3^--N	Nitrate as Nitrogen
NTC	Nutrient Transfer Continuum
NUE	Nitrogen Use Efficiency
OJEC	Official Journal of the European Communities
OM	Organic matter
P	Phosphorus
PIP	Pollution Impact Potential
PON	Particulate Organic Nitrogen
PP	Particulate Phosphorus
QC	Quality Control
R^2	Coefficient of determination
S.I.	Statutory Instrument
S-M-P-R	Source-Mobilisation-Pathway-Receptor
SRP	Soluble Reactive Phosphorus
TDP	Total Dissolved Phosphorus
TN	Total Nitrogen
TON	Total Oxidised Nitrogen
TP	Total Phosphorus
TRP	Total Reactive Phosphorus
TSS	Total suspended solids
WFD	Water Framework Directive
WSP	Water Soluble Phosphorus
µm	Micrometre

1. Introduction

1.1 Overview

This chapter provides the background context to this research, identifies knowledge gaps, and presents the aims and objectives.

1.2 Background

Ireland's agricultural economy relies on its grassland for dairy production and exportation (Bord Bia, 2019). With about 90 % of its agricultural lands under grassland (O'Mara, 2012), Ireland's temperate climatic conditions support longer grass growth periods (Humphreys et al., 2009), enabling the maximation of grazed grass conversion (the cheapest feed approach) to support its livestock production (Finneran et al., 2012).

The climatic condition in Ireland is characterised by mild winters and cool summers, with a 30-year long term (1991-2020) average annual rainfall of 1288 mm, ranging from 878 mm along the east coast to 2045 mm in the southwest mountainous regions (Coonan et al., 2024). Over this period, winter and autumn have the highest average seasonal rainfall, 380 mm and 369 mm, respectively. December, October, November and January are the wettest months, with average rainfall ranging from 130 to 142 mm while April and May are the driest months receiving 79 - 82 mm rainfall. Annual mean air temperature for Ireland is 9.8 °C, ranging from 8.5 °C to 10.8 °C (Curley et al., 2023). Mean air temperatures are highest in summer (14.6 °C), followed by Autumn and Spring with means of 10.3 °C and 8.8 °C, respectively, and lowest in winter with a mean of 5.4 °C. However, such climatic conditions of high rainfall and low temperatures create wet soils (impeded drainage and high water table) and hinder grass growth for optimum milk conversion efficiency (Shalloo et al., 2004) on marginal grasslands. These marginal grasslands have soils with poor drainage, limiting their agricultural yields.

Of the 3.18 million ha of managed grassland in Ireland, marginal grasslands comprise up to 0.96 million ha (30 %) that are imperfectly or poorly drained (O'Sullivan et al., 2015). Data from 2020 - 2025 across some of these grassland areas indicate total annual rainfall ranging from 1092.5 mm (2021) to 1418.5 (2020), mean air temperatures ranging from 9.5 °C to 10.4 °C and all year-round rainfall excess (rainfall minus evaporation) ranging from 8.2 mm in May to 126 mm in October (unpublished). These grasslands contribute up to 30 % of total milk production (O'Loughlin et al., 2012). They therefore require measures to meet production demands (Dillon et al., 2005). One such measure is the installation of agricultural drainage systems, which improves grass growth and overall trafficability (Teagasc, 2022; Tuohy et al., 2019).

Agricultural drainage systems comprise subsurface and surface systems (Ritzema, 2006; Tuohy et al., 2013). Subsurface drainage systems (i.e., in-field and collector drains) are buried within grasslands and remove excess water into surface drainage systems (open artificial ditches and natural open drains, also referred as main drains) (Skaggs et al., 2012). However, they can facilitate varying outcomes of nutrient losses (Ibrahim et al., 2013; Skaggs et al., 1994; Strock et al., 2010) as drained water interacts with soil hydrology, surface hydrology and soil chemistry (Granger et al., 2010). Surface drains, as the final component within the drainage system, directly connect these potential nutrient losses to receiving waters. Nutrient losses from surface drains depend on connectivity to surface and subsurface hydrological pathways from in-field drains, farmyards and farm roadways, as well as soil nutrient status, farm management, climate and attenuation capacity. Consequently, this poses varying risks from surface drains to larger receiving waters. Understanding nutrient loss connectivity mechanisms, identifying the hydrological connectivity flows and assessing the risk status on the surface drains allows the identification of areas in the surface drainage network that pose a risk for nutrient loss and into which appropriate mitigation measures may be installed.

1.3 Knowledge gaps

The following knowledge gaps in the research have been identified:

- Subsurface and surface hydrological connectivity pathways (as defined in Table 1.1) on surface drains have not yet been included in connectivity risk ranking studies. Identifying such connectivity improves understanding of immobilization and transformation processes of nutrient loss through hydrological pathways into open ditches (Deelstra et al., 2014). Current research considers only phosphorus (P) connectivity risk ranking, and not nitrogen (N) (Moloney et al., 2020).
- Nutrient-loss influencing factors from field management practices, landscape and soil characteristics, and surface and subsurface hydrological connectivity under the source-mobilisation-pathway-receptor (S-M-P-R) of the nutrient transfer continuum (NTC) vary spatially and temporally (Adams et al., 2022; Harrison et al., 2019; Withers and Lord, 2002). Such variations may lead to varying nutrient loss risks posed to directly connecting surface drainage channels. However, knowledge on how these nutrient loss risks are classified on surface drains for targeted mitigation is limited.
- Farm roadways, when connected to surface drains, act as hydrological pathways transporting nutrients and sediments from poached, soiled and disturbed road surfaces to surface drains (Fenton et al., 2021; Rice et al., 2022). Mitigation measures exist in broad terms, but bespoke solutions are needed for specific roadway runoff scenarios. The Nitrate Action Programme (NAP) recommends multiple mitigation measures and highlights a “right measure, right place” approach to address diffuse pollutant sources,

including farm roadways (DHLGH, 2024). However, the implementation of these mitigation measures has generally only occurred on a very small number of farms with no efficiency testing to guide future iterations and improvements of the mitigation measures.

Table 1.1 Criteria for surface and subsurface connectivity pathways on surface drains

Connectivity pathway	Source of connection	Criteria description[1]
In-field drains	Subsurface	Evidence of in-field pipe drains connecting into ditch, usually with less water flow.
Farm roadway	Surface	Evidence of farm roadway and hard surface runoff connectivity with the open ditch network (directly during rainfall or indirect signs such as established rills and breakthrough points).
Groundwater springs	Subsurface	Evidence of natural springs or pipe springs (with high water flow) connecting into ditch.
Groundwater upwelling or seepage	Subsurface	Evidence of groundwater seeping from either base or side of ditch into the ditch.

[1] Criteria description (Teagasc, 2022)

1.4 Research aim and objectives

The overall research aim of the study was to undertake an in-depth analysis on nutrient loss from drainage systems across a broad spectrum of landscape, soil and drainage system types; establish new knowledge and insights on nutrient loss connectivity risk to improve tailored mitigation, and test tailored mitigation options for managing nutrient loss risks.

To address this aim, the objectives were to:

- (Objective 1) Improve understanding of nutrient loss risk dynamics associated with surface drains and review existing classification systems to assess the nutrient loss risks in surface drains.
- (Objective 2) Establish nutrient loss risk from surface and subsurface hydrological connectivity pathways on surface drainage systems to improve existing P-only connectivity risk classification and create an integrated N and P loss connectivity risk classification for surface drains.
- (Objective 3) Assess the nutrient loss influencing factors to identify risky surface drains and establish key influencing factors within the surface drainage network.
- (Objective 4) Develop, implement and monitor a mitigation measure on high-risk surface drains with farm roadway connectivity.

1.5 Structure of dissertation

The dissertation is structured under six chapters as presented in Figure 1.1.

Chapter 2 reviews the current understanding of the dynamics of nutrient loss risks associated with surface drainage systems on grassland farms. It investigates existing assessments classifying nutrient loss risks on surface drainage systems for targeted mitigation on poorly drained grassland farms. This chapter addresses the first objective of this study.

Chapter 3 derives a farm-scale integrated open ditch risk ranking for both P and N loss risk based on connectivity to inform future mitigation management on heavy textured, grassland dairy farms. The chapter validated the presence or absence of pathways for N and P based on a conceptual understanding of hydrological pathways and developed an integrated N and P loss connectivity risk classification of an open ditch (artificial surface drain) network. This chapter addresses the second objective of this study.

Chapter 4 develops a semi-quantitative risk model for heavy textured grassland dairy farms that identifies open (surface) drainage channel network sections that pose a risk of contributing nutrients to the adjoining aquatic water courses. This builds on the theory of the previous chapter and captures all other relevant S-M-P-R factors under the open drainage network nutrient transfer continuum to rank the nutrient loss risk in an open drainage channel network on a farm. This chapter addresses the third objective of this study.

Chapter 5 adopts the “right place, right measure” philosophy to identify surface roadway runoff connectivity to surface (open) drainage channels (i.e., “right place”), co-design and co-implement with farmers suitable mitigation measures at farm roadway critical source area (CSA) locations (i.e., “right measure”), and monitor the efficiency of the mitigation measures by measuring nutrient and sediment removal efficiency over time. Information on efficiency of the mitigation measures enhances their adoption for managing surface hydrological connectivity to connecting surface drains. This chapter addresses the fourth objective of this study.

Chapter 6, the final chapter, details the conclusions and wider implications of the study, along with future recommendations.

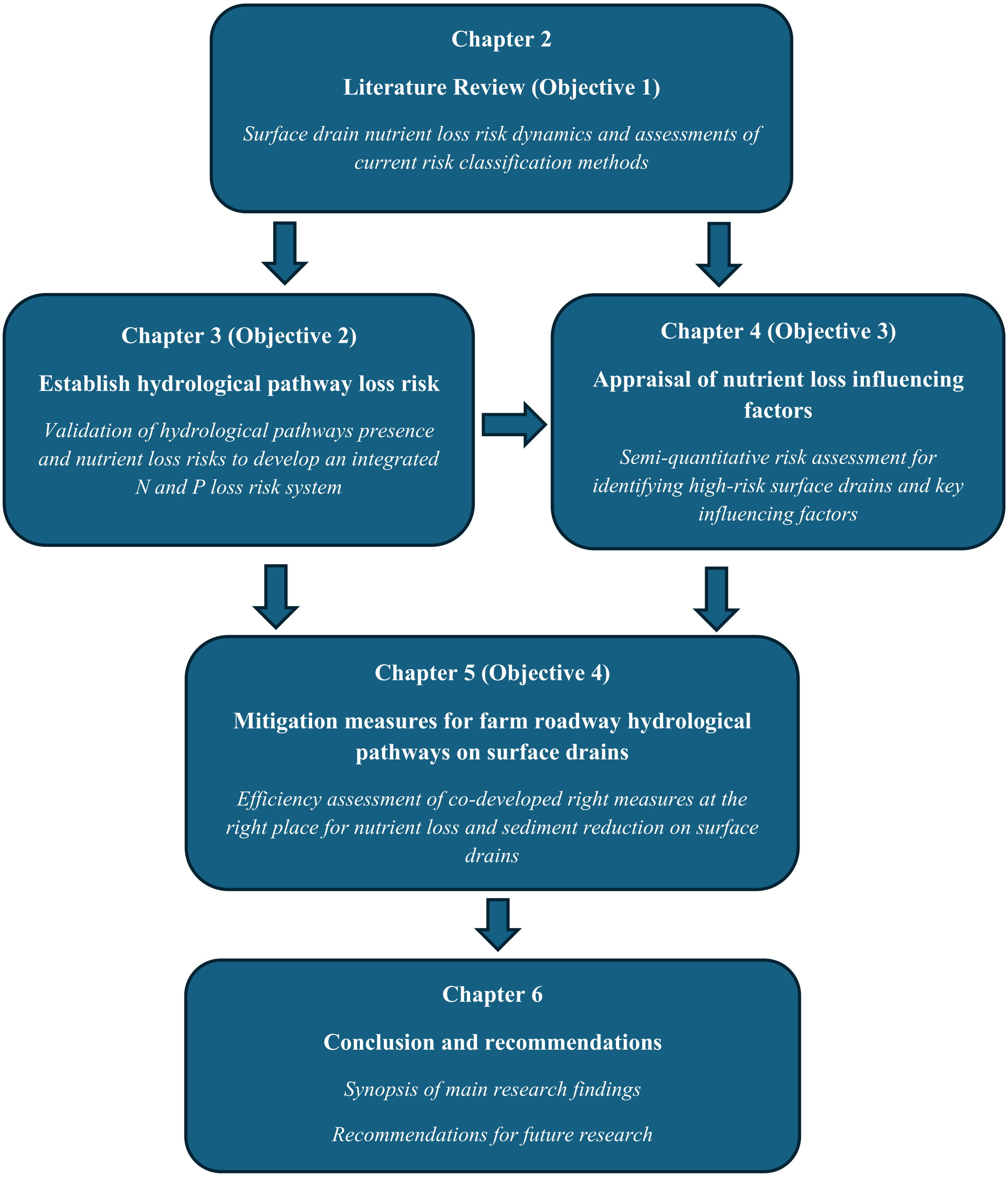


Figure 1.1 Flowchart of thesis structure

1.6 Contribution to Existing Knowledge

1.6.1 Journal Articles (Published)

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T., & Tuohy, P. (2024). An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management. *Frontiers in Environmental Science*, *12*(1337857), 1–16. https://doi.org/10.3389/fenvs.2024.1337857

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T. & Tuohy, P. (2024). A semi-quantitative risk model for dairy farms to pinpoint and break source-pathway connections between nutrient sources and open drainage channel sections. *Frontiers in Environmental Science*, *12*(1435418), 1–13. https://doi.org/10.3389/fenvs.2024.1435418

Opoku, D. G., Healy, M. G., Fenton, O. & Tuohy, P. (2025). Examination of nutrient and sediment loss mitigation for farm roadway runoff on an Irish dairy farm. *Journal of Agricultural Water Management*, 322, 110007. https://doi.org/10.1016/j.agwat.2025.110007

1.6.2 Conference Publications

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T. & Tuohy, P. (2023). An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management. *NCERA-217: Drainage Design and Management Practices to Improve Water Quality*. 4th–6th April 2023, Tidewater Inn, Easton MD, USA. Poster presentation.

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T. & Tuohy, P. (2022). An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management. *International Symposium on Climate-Resilient Agri-Environmental Systems (ISCRAES)*, 28th August 2022, Talbot Hotel Stillorgan, Dublin. Poster presentation.

1.6.3 Practical/Popular publications

Opoku, D. G**.,** Healy, M. G., Fenton, O., Daly, K., Condon, T., & Tuohy, P. (2025, July 2). *Assessing connectivity risks on surface open drains to minimise nutrient losses. In Irish Dairying – Innovating for the Future – Open Day Proceedings.* Teagasc Moorepark, Animal and Grassland Research Innovation Centre.

Heerey, L., Opoku, D. G., Sifundza, L., Maher, P. J., Condon, T., Fenton, O., Daly, K., Tuohy, P., & Murnane, J. (2024, July 16). *Targeted mitigation: Breaking surface connectivity on farms – drainage ditches and roadways. In Farming for a better future: Resilient and sustainable farming systems – Open Day Proceedings* (p. 115). Teagasc, Johnstown Castle Environment Research Centre.

The journal, conference abstracts and popular publications can be found in Appendix A.

2. Literature Review

2.1 Overview

This chapter reviews nutrient loss risks associated with surface drainage systems on poorly drained grasslands. It presents information on the current understanding of the dynamics of nutrient loss risks associated with surface drainage systems on grassland farms on poorly drained soils, and examines existing assessments that categorises or ranks surface drainage systems based on their nutrient loss risks, aiming to enhance targeted mitigation on poorly drained grassland farms.

2.2 Introduction

In temperate locations, high rainfall and low evapotranspiration on poorly drained grasslands create water-saturated root zones that deplete soil oxygen for root growth and promote unhealthy crop growth (Enciso et al., 2009). This limits animal and machinery traffic (Beukes et al., 2013) and impedes profitability (Shalloo et al., 2004). Cow trafficking on wet grasslands can reduce grass utilisation by 20–40 % and pasture growth by up to 34 % (Herbin et al., 2011). To address these issues, agricultural drainage systems are installed to increase hydraulic conductivity, allow excess water removal and control water table levels, thereby enhancing grass production and reducing adverse field trafficability conditions (Ibrahim et al., 2013; Tuohy et al., 2018).

Agricultural drainage systems comprise subsurface or surface drainage systems (Gaillot et al., 2021; Ritzema, 2006; Tuohy et al., 2013). Subsurface drainage systems are buried within grasslands and remove excess subsoil water and channel into surface drainage systems (Skaggs et al., 2012). These surface drainage systems are open artificial (ditches) and natural main drains which drain into receiving water sources (Schultz et al., 2007) and therefore act as an important and final component for nutrient losses. These nutrients may be mobilised, and during transfer may be adsorbed onto sediment often in the case of P, transformed often in the case of N, and/or released to receiving water sources (Daly et al., 2017; Ezzati et al., 2020; Mattila & Ezzati, 2022). Knowledge on the nutrient loss dynamics of these hydrological pathways connectivity to surface drainage systems is crucial for nutrient loss management on drainage systems. Moloney et al. (2020) developed a system to evaluate and rank the risk of nutrient loss from surface drainage systems, but only considered P loss and did not consider the potential impact hydrological connectivity pathways such as surface roadway, subsurface infield drains, and groundwater springs, upwelling or seepage. The incorporation of N into this risk classification is important in terms of refining surface drain connectivity nutrient loss. It is also important to include spatially varying hydrological connectivity pathways, which are influenced by factors such as soil, weather, farm management and hydrogeochemistry.

It is essential to identify sections of a surface drainage system that pose a risk for nutrient loss to receiving water sources. A risk assessment needs to be cognisant of the complex interactions of the surface drainage system with existing farm management, hydrology, soil and landscape. These complex spatial and temporal interactions influence nutrient dynamics along the NTC, comprising S-M-P-R (Harrison et al., 2019; Mellander et al., 2017). Most risk assessments of nutrient losses from farms or fields only consider surface drainage systems as pathway factors in the NTC (Davison et al., 2008; Mockler et al., 2017; Roberts et al., 2017; Schoumans & Chardon, 2003; Thomas et al., 2016) but do not consider all aspects of the NTC. A limited number of studies have focused on factors influencing the NTC, such as the spatiotemporal variation factors (Chen et al., 2023; Wang et al., 2017). There are also a limited number of studies that directly assess the nutrient loss risk based on factors related to certain aspects of the NTC on surface drainage systems, such as connectivity to source within farm landscape (e.g. farmyards, field and soils) and sediment and nutrient transport potential (Moloney et al., 2020; Shore et al., 2015).

Critically evaluating and synthesising the existing knowledge on the dynamics of nutrients (N and P) loss within surface drainage systems helps improve clarity and identify shortfalls, which inform proper mitigation strategies. Ascertaining such information, especially for high-risk surface drainage systems, enables the appropriate mitigation method selection, development, and management to limit nutrient losses. Post mitigation selection, efficiency assessment of co-developed mitigation measures is required to enhance mitigation uptake among farmers.

The aim of this review is to present the current understanding of the dynamics of nutrient loss risks associated with surface drainage systems and discuss implications for mitigation options on grassland farms. Specifically, this review seeks to investigate the existing risk assessments of nutrient losses on surface drainage systems for targeted mitigation on poorly drained grassland farms.

2.3 Nutrient losses in drainage systems

In agricultural grasslands, nutrient loss is a product of the interactions between farm practices, hydrology, soil and landscape features (Granger et al., 2010). This may be conceptualised as a NTC comprising a S-M-P-R (Haygarth et al., 2005). Nitrogen and P are the major nutrients of concern, and are sourced from applied fertilisers, organic animal waste, and soil legacy nutrient sources. Applied fertiliser type, whether organic or inorganic, and the amount/rate of fertiliser (kg) applied, influence loss from nutrient sources (Richards et al., 2015). Nutrient sources of organic animal waste (faeces and urine) may come from livestock within farmyards (Vero et al., 2020) and on grazing fields (Bilotta et al., 2007). Legacy nutrient sources are existing nutrients in the soil, and factors such as soil P status have been used to classify the risk for potential loss (Moloney et al., 2020). Fenton et al. (2022) showed legacy nutrients can be found in farm roadways, indicating their role as a nutrient source contributing to the NTC. The available nutrients are primarily mobilised from the nutrient sources by desorption, incidental losses, solubilisation and/or detachment (Granger et al., 2010) driven by hydrological flows originating from rainfall (Yao et al., 2021). These processes may also be influenced by other processes including mineralisation, pH and redox-driven nutrient release, which increase nutrient availability before mobilisation. The mobilised nutrients are transported from either surface flows, including farm fields and hard standing areas (farmyard area and roadway), and subsurface flows into receiving water bodies (receptors).

Poorly drained grassland soils requiring drainage systems have infiltration impediments that restrict flows into the subsurface zones, and therefore mobilised nutrients are carried away through surface runoff to receiving water bodies. However, where drainage systems are installed, hydrology, the primary driver of nutrient transfer (Sukias et al., 2003), is affected, which alters infiltration and lateral flow of excess water and nutrients (Gramlich et al., 2018). These nutrients may be transported through excess field water collected from the subsurface zone by subsurface drains (in-field drains) and transferred into the surface drain (main drain) (which also drains surface runoff flows), before exiting through an outlet into receiving waters (Figure 2.1).

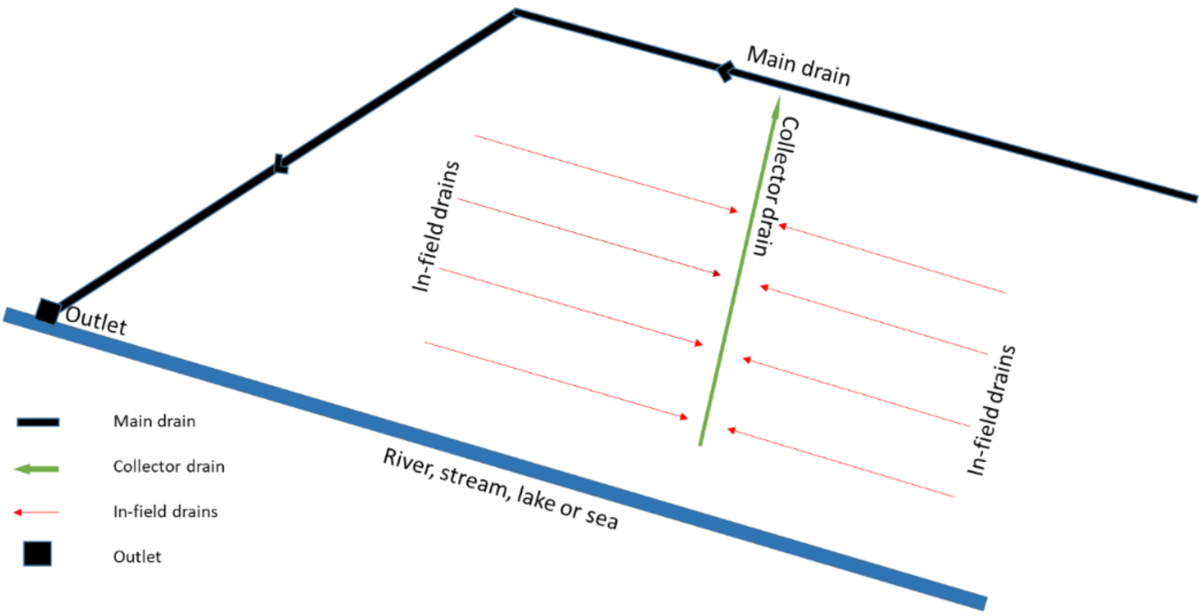


Figure 2.1 Layout of a drainage system. Adapted from Schultz et al. (2007).

During the transport, the nutrient content may be increased by legacy soil nutrients, reduced through adsorption or trapping, and/or transformed within the flow, depending on the hydrology, soil, and drainage characteristics prior to delivery into receiving waters.

2.3.1 Hydrology and soil characteristics

The installation of drainage systems in poorly drained soils creates macropores such as soil cracks, and facilitates preferential flows (Blann et al., 2009; Granger et al., 2010) exceeding the natural matrix flow in soils. Such flows shorten travel (residence) time for interaction, thereby reducing the filtration of nutrients carried from the surface (Kladivko et al., 1999; Zimmer et al., 2016). This limits nutrient attenuation (O'Sullivan et al., 2015) and denitrification potential (Clagnan et al., 2020) and can increase nutrient load transfer across the wider farm area (Heathwaite & Dils, 2000; Manninen et al., 2018).

The efficiency of macropores in facilitating drainage water varies by soil type and texture. Macropores form due to shrink-swell moisture dynamics (Liu et al., 2014; McCarter et al., 2020) of retained water in fine pores in clayey soils (Peng & Horn, 2007), and this may vary depending on clay content (Vogel et al., 2005). They may also develop due to reduced water retention in organic matter and its decomposition in organic soils (Liu et al., 2016), and this impact may be either short or long-term (Gramlich et al., 2018; Holden et al., 2006). Depending on the availability of conditions for macropore formation, differences in flow efficiency may occur, and consequently the nutrient loss potential may differ. Additionally, the inherent soil types possess varied affinity for P that influences P adsorption capacity and loss potential to receiving waters (Roberts et al., 2017).

2.3.2 Surface drainage system type, design and characteristics

Surface runoff, together with subsurface in-field drain runoff from the amended hydrology as described above, transports nutrients into surface drains. Factors including topography (Gramlich et al., 2018; Roberts et al., 2017), subsurface in-field drainage design (Tuohy et al., 2015, 2016; Tuohy et al., 2018), surface (Noij et al., 2013) and in-drain vegetation (Castaldelli et al., 2015; Soana et al., 2017), episodic and low rainfall intensity (Kleinman et al., 2006; Tuohy et al., 2016), and antecedent soil moisture (Adams et al., 2022; Tuohy et al., 2016) influence runoff volume and determine nutrient loss species and types through the surface drainage systems. Sloped landscapes under rapid flows promote particulate nutrient losses, while flat landscapes under base flows promote soluble nutrient losses (Kleinman et al., 2007). In addition, the flow variations defined by the closeness (intensity) and depth designs of both surface and subsurface drainage systems influence the load and type of nutrient discharges from the surface drainage system (Song et al., 2013). The work of Cassidy et al. (2017) showed deterioration of subsurface drainage systems may increase nutrient release to connecting surface drainage systems.

While vegetation within surface drainage systems reduces nutrient losses (Castaldelli et al., 2015; Soana et al., 2017), dredging (removal of benthic biota, sediments and standing biomass) or vegetation cutting can negate this benefit with remobilisation of trapped nutrient-retaining sediment (Blann et al., 2009; Strock et al., 2010), especially during high flows (Powell et al., 2007). Such dredging impacts are characterised as short-term nutrient transfer (Smith, 2009) but replenish and act as sinks for adsorbing nutrients in the long-term (Daly et al., 2017; Ezzati et al., 2020; Smith & Huang, 2010).

2.3.3 Nutrient transfer continuum within surface drainage systems, and implications for nutrient losses

Nutrients enter surface drains through flows from connecting hydrological pathways. In surface drains, connecting hydrological pathways include surface flows such as farmyards and farm roadways (Fenton et al., 2022, 2024b), subsurface flows such as in-field drains, and groundwater flows such as springs, seepage and upwellings (Simpson et al., 2011; Teagasc, 2022). Nutrients contributed by these hydrologically connected flows may vary in dominant nutrient species and concentrations, so there is a need for a thorough understanding of their individual risks for tailored mitigation. Instances of multiple nutrient loss risks have also been reported from individual surface drains (Clagnan et al., 2019; Ezzati et al., 2020).

Existing surface drainage system studies have focused on improving our understanding of numerous aspects, including the organic matter content (Hunting et al., 2016), surface drainage channel management (Dollinger et al., 2015; Hertzberger et al., 2019), changes in dissolved organic carbon (Tiemeyer & Kahle, 2014), sediment attenuation potential (Ezzati et al., 2020; Mattila & Ezzati, 2022), vegetation-enhanced attenuation potential (Soana et al., 2017; Zhang et al., 2020), and greenhouse gas emitted indirectly from the system (Clagnan et al., 2019; Hyvönen et al., 2013). These studies have highlighted that nutrient transfer may be altered within surface drainage systems, subsequently changing the nutrient composition and concentrations/loads delivered to receiving water bodies. These factors interact and complicate nutrient loss impacts. Therefore, isolating the influence of individual factors in assessing the risk of nutrient loss may prove difficult. To enable effective management of nutrient loss from surface drainage systems, similar detailed but integrated studies on nutrient contributions are needed (Granger et al., 2010).

2.4 Nutrient loss risk assessment on surface drainage systems

Assessing the risk of nutrient loss from surface drainage systems is essential for improving water quality (Needelman et al., 2007; Strock et al., 2010). Risk assessments for nutrient loss on surface drains aim to classify and rank surface drainage networks based on their risk potential. On grassland farms, multiple factors (e.g., applied fertilisers, rainfall, soil drainage, field slope, soil and sediment nutrient concentrations; Granger et al., 2010; McDowell et al., 2001) and processes (e.g., chemical and biological processes of nutrient solubilisation and detachment, nutrient transformation, interception and uptake; Bieroza et al., 2020; Granger et al., 2010) influence nutrient losses. Increasing the accuracy of the risk assessment can be achieved by including detailed nutrient loss contributing factors and processes to properly replicate the conditions for nutrient loss. A limited number of studies have ranked surface drainage systems' nutrient loss risks on grassland farms. Examples include catchment-scale (for sediment and associated P transport; Shore et al., 2015) and farm-scale (of landscape connectivity for P loss; Moloney et al., 2020) studies. The implications of the resolution of nutrient contributing factors and processes, and limitations of the risk ranking approaches used in literature are now explored within a NTC framework.

2.4.1 Factors and processes used in risk assessment

Not all studies in the literature use the NTC framework in ranking nutrient losses risk in surface drainage systems. The type and quantity of factors used in predicting nutrient loss risk vary (Table 2.1). For example, only physical factors (e.g. vegetation presence and drain slope) of the NTC pathway were assessed by Shore et al. (2015). This approach by Shore et al. (2015) created a 4-risk ranking system comprising 'Class 1' (streams, with low fine sediment retention potential), 'Class 2' (high slope surface drains of low-to-moderate fine sediment retention potential), 'Class 3' (moderate slope surface drains of moderate-to-high fine sediment retention potential), and 'Class 4' (low slope surface drains that retain fine sediment). While this approach may evaluate some aspects of nutrient loss risk for sediment and potential of associated P loss from surface drains, considering sediment loss metrics alone may be inaccurate (Sharpley et al., 2007; Sherriff et al., 2018), as biogeochemistry is not considered (Cassidy et al., 2017; Granger et al., 2010). Neglecting such important factors assumes all sediment and associated P have the same P source and concentrations. Moloney et al. (2020) addresses this shortcoming and incorporates source and pathway factors (landscape positioning/connectivity to source/water sources) and chemical factors (drain sediment P chemistry) to assess risk of P loss for all surface drains on grassland farms. This assessment developed a 5-risk ranking system: with farmyard connection drain as the highest risk, followed by outlet, outflow, secondary drains, and disconnected drains. However, this assessment neglects potential nutrient loss risk factors related to hydrological connectivity to surface drains along the NTC pathway.

Table 2.1 Review of risk ranking assessment of P and N loss on surface drainage system on grassland farms.

Risk classes (in order of descending risk)	Nutrient transfer continuum (NTC) aspects[1] and factors[2] considered in risk classification	Analysis of parameters in NTC	Risk validation	Pros of risk assessment	Cons of risk assessment	Reference
1. 'Class 1' (streams) - low fine sediment retention potential 2. 'Class 2'- high slope surface drains of low-to-moderate fine sediment retention potential 3. 'Class 3' - moderate slope surface drains of moderate-to-high fine sediment retention potential 4. 'Class 4' - low slope surface drains that retain fine sediment	Pathway -Physical characteristics of surface drain -Presence of sediment cover -Presence of in- and along-drain vegetation cover	Risked by slope, and its tendency to retain/mobilise sediment and associated P. All natural surface drains (streams) are ranked high-risk, regardless of slope.	Net accumulation of fine sediment in the surface drains measured as: -high (≥75 % sediment cover) -moderate (25 % and 75 % sediment cover) -low (≤25 % sediment cover)	Separate risk classes for natural and artificial surface drainage systems, which accommodate varying flow dynamics in sediment and associated P retention/transfer. Risk assessment developed for storm flow conditions, although assessment survey was done under low flow conditions due to the applicability of sediment net accumulation assessment in both flows. Effective at large scale such as catchment Easy and direct observation validation methods	Risk based only on sediment physical characteristics, not chemical characteristics. Excludes factors from other NTC aspects such as source, receptor, and the complete assessment of risk from some pathway factors (e.g. in-drain vegetation and hedgerow types on flow power and velocity) Factors like vegetation may not be present in some drains, so their inclusion in sediment retention/transfer risk ranking principles may be limiting. Risk ranking was only validated for summer	Shore et al. (2015)
1.Farmyard connection drains 2.Outlet drains 3.Outflow drains 4.Secondary drains 5.Disconnected drains	Pathway -Landscape position within farm -In-drain sediment P dynamics.	Uses surface drain connectivity to farmyard (source) and proximity to nearby surface waters (receptor) at farm-scale for P loss risk ranking	Sediment dynamics and drain water hydrochemistry assessment to validate risk classes.	Risk classes and validation from varying landscapes and grassland farm systems enhance inclusion of wide spectrum of risks to improve reliability and applicability. Combining landscape and physicochemical characteristics enables inclusion of risk potential from nutrient load transport (to receptor and from source) and transport dynamics to define the risk classes.	Risk assessment omits N connectivity risk losses. The risk of nutrient potentially introduced from other surface and subsurface hydrological connectivity flows not assessed. Does not distinguish between subsurface drains and surface drains connected to farmyard. Excludes geographically varying risks from other	Moloney et al. (2020)

				Employs GIS tools, ground scoring and sampling which improves risk certainty to develop tailor-made management. Effective at farm-scale, although improvement is required	NTC aspects and assessment of temporal risks. Includes only transport proximity to source for only one surface drain class farmyard and neglects potential diffuse sources from surrounding fields proximity	

[1]Aspects refer to source, mobilisation, pathway, and receptor of the NTC that describes nutrient losses.
[2] Factors refer to nutrient contributing entities that describe every NTC aspects for nutrient losses.

Increasing the resolution of the analysis to include hydrological connectivity increases the variability of nutrient-contributing factors, thereby increasing the effectiveness in determining potential risk of nutrient loss (Hayes et al., 2023). These hydrologically connected flows including farm roadways (Fenton et al., 2021; Rice et al., 2022) springs (Soana et al., 2017), subsurface (in-field) drains (Ibrahim et al., 2013; Valbuena-Parralejo et al., 2019) and groundwater upwelling and seepage (Gold et al., 2002; Williams et al., 2015) potentially introduce nutrients of varied compositions (King et al., 2014; van Esbroeck et al., 2016) from CSAs into surface drains, and confirms Reid et al.'s (2018) postulations that nutrient loss to water sources may enter from different pathways connecting from multiple nutrient sources. Incorporating the risk associated with hydrological connectivity to surface drains in the risk assessment increases an assessment's ability to assess different surface drain conditions and define risks for targeted mitigation.

Hayes et al. (2023) report that P losses through hydrological connectivity vary even under similar management and landscape conditions, and over small distances (Adams et al., 2022). Therefore, assessing factors beyond field-scale may be the best approach for the assessment of nutrient loss risk from hydrological connectivity. This highlights the need to develop a risk ranking system based on comprehensive risk scoring assessment of spatial, physical and chemical factors that influence nutrient loss under all NTC aspects on surface drains and adjacent fields. The incorporation of a high number of risk factors at such detailed resolution in the assessment of the nutrient loss risk increases accuracy in identifying risk hotspots for targeted mitigation relative to the previous approaches, even beyond farm-scale level to field/drain-scale level.

The assessment of physical and chemical factors (Moloney et al., 2020) allows risk evaluations that include variations in nutrient solubility, retention, availability and transport (Haygarth et al., 2014; Jarvie et al., 2013; Kleinman et al., 2011; Sharpley et al., 2013). Incorporation into the nutrient loss assessment of the point source connectivity, sediment equilibrium P concentration along with the soil and legacy P connectivity from adjacent fields (Moloney et al., 2020) will increase the accuracy of a risk ranking system for surface drainage networks.

The inclusion of physicochemical factors such as hydrological connectivity flows, soil properties, farm management practices, and meteorological factors in the NTC S-M-P-R framework will allow a holistic risk assessment of factors that characterise nutrient availability until delivery into water sources. This increases risk prediction accuracy and sensitivity to targeted mitigation measures (Cherry et al., 2008). Primarily, this methodology will allow GIS and high-resolution data assessment of the factors related to nutrient input and intensity, release, transport, and delivery at different spatial locations in developing a quantitative surface drain risk ranking. Subsequently, this improves understanding of the nutrient-contributing factors and processes (Hayes et al., 2023; Niemi et al., 2023; Scott et al., 2024), eliminates subjectivity in the development of the risk ranking, and allows for a quicker identification of the high-score nutrient-contributing factors.

2.4.2 Limitations of the risk assessments

Assessing P loss from sediments in surface drains by solely relying on physical indicators without direct P concentration/load measurements (Shore et al., 2015) may underestimate or overestimate P losses (Vadas et al., 2004). Cassidy et al. (2017) reported that assessing only topography, a physical factor, leads to inaccurate assessment of risk. Ensuring a comprehensive assessment of nutrient-contributing chemical and physical factors during the nutrient transport continuum from S-M-P-R improves risk ranking systems and is important for characterising nutrient loss conditions.

The Shore et al. (2015) risk assessment lacked source–connectivity to receiving water sources. Furthermore, the risk assessment in Shore et al. (2015) and Moloney et al. (2020) excluded an assessment of N loss risk, limiting their application for assessing complete farm nutrient loss risk from surface drains. The risk assessments in both Shore et al. (2015) and Moloney et al. (2020) do not consider nutrients potentially introduced from hydrological connectivity flows. These require data-intensive assessments at field and drain-scales, which may hinder practical application for farmers. Van den Berg et al. (2023) noted these technical and resource demands were some of the bottlenecks for farmers to undertake management and mitigation measures. Other issues in high data-intensive assessment may include the use of experts' subjective opinions in assessing data. The use of experts' opinions may promote bias and skewed results in replicating the risk assessments and limit the reliability in pinpointing high-risk surface drains. Therefore, it is imperative that experts' opinions are critically assessed in a systematic and transparent approach to prevent bias in farm nutrient management (Agarwal et al., 2016), and their use in modelling (Krueger et al., 2012).

The lack of seasonal or temporal assessments in all risk assessments of the published studies limits the accuracy of risk predictions. Seasonal or temporal variations influence nutrient loss contributing factors from farm management, weather and landscape (Zhang et al., 2004). A one-time assessment limits the true representation of nutrient loss risk. Undertaking temporal sampling influences the selection and weighting of contributing factors for each season. For example, weighting for surface runoff may be high during wet periods relative to dry periods, as high saturation in soil reduces soil moisture deficits and encourages surface flow. Following different flow conditions and periods, the composition of P species lost through drain discharges varies. Particulate phosphorus (PP) has been shown to dominate P losses under high rainfall (Simard et al., 2000), whereas dissolved reactive P (DRP) accounted for 66–86 % of P in a study conducted under above-average rainfall conditions (Heckrath et al., 1995). These variations in P species composition are influenced by soil characteristics including soil matrix DRP sorption capacity (Simard et al., 2000), P-rich soil particles availability, stored P pools (Delgado et al., 2006) and subsoil erosion processes (Cooke, 1976; Simard et al., 2000). They are also influenced by rainfall characteristics including intensity driving rapid or slow movement of freshly applied fertiliser and P-rich soil particles and frequency driving PP losses and DRP solubilisation.

The influence from vegetation as a factor in surface drains mitigates nutrient concentrations (Castaldelli et al., 2015; Moeder et al., 2017; Soana et al., 2017; Västilä et al., 2021) prior to entering larger water sources. The influence of vegetation is only partially considered in Shore et al. (2015) risk assessment. Vegetation characteristics vary within surface drains (Bouldin et al., 2004), with vegetation characteristics such as the presence of wood and leaves determining their mitigation potential for nutrients (Kumwimba et al., 2024). Over time, this vegetation may also undergo senescence, decomposing into organic matter and releasing mineralised nutrients back into the surface drains. The conditions of vegetation may vary depending on weather conditions, and may result in differences in their mitigation potential. Defining the influence of vegetation characteristics in risk assessment at a particular time of assessment may be critically important for accurate nutrient loss risk predictions.

Considering these seasonal and temporal variations in risk assessment would further improve adaptation to future changes caused by climate change in risk assessment of nutrient losses from surface drainage systems. Climate change impacts on agricultural water quality are expected to intensify in the coming years (Mellander & Jordan, 2021), and it is imperative that these risk assessments incorporate the impact of these anticipated intensified precipitation and extreme weather conditions for proactive and pragmatic surface drain nutrient loss management. It is important to highlight the need for undertaking long-term validation. This was absent from all the different risk ranking assessments identified, but would improve predictions for temporal risk changes and make these assessments robust for future climate change impacts.

2.5 Summary

In this chapter, the dynamics of nutrient loss connectivity risk on surface drains on poorly drained grasslands were presented. This was followed by an in-depth analysis of the existing nutrient loss connectivity risk classification on surface drains in poorly drained grasslands. Some of the key gaps in knowledge identified were the:

1. Absence of N loss risk in nutrient loss connectivity risk on surface drains in grasslands.
2. Exclusion of surface (hard standing including farmyards and farm roadways) and subsurface (infield drains, groundwater springs, upwelling and seepage) hydrological pathways with connectivity to surface drains in nutrient loss risk ranking.
3. The absence of a risk assessment model that assesses all the spatially and temporally varying nutrient loss contributing factors on surface drains along the NTC on grasslands.
4. Need to develop mitigation measures to reduce nutrient losses from connecting hydrological pathways on surface drains.

3. An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management.

3.1 Overview

The aim of this chapter was to derive a farm-scale integrated open ditch risk ranking for both P and N loss risk based on connectivity, to inform future mitigation management on heavy textured, grassland dairy farms.

Parts of this chapter have been published in Frontiers in Environmental Science (Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T., & Tuohy, P. (2024). An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management. *Frontiers in Environmental Science*, *12*(1337857), 1–16. https://doi.org/10.3389/fenvs.2024.1337857)

To avoid repetition, acronyms that have already been defined in preceding chapters are not defined in this chapter.

3.2 Introduction

Open ditch networks, also referred to as "surface ditch networks", are installed in poorly-drained soils to remove excess water, control the water table, and aid with grass production and utilisation (Tuohy et al., 2016; Hertzberger et al., 2019). These networks comprise a series of connected and unconnected sections that receive nutrients from a variety of surface and subsurface pathways, all of which can then be transported to other sections or associated water bodies (Herzon & Helenius, 2008; Kröger et al., 2007; Moloney et al., 2020). Connectivity is defined as the transfer of energy and matter across two landscape zones, whereas disconnectivity is the isolation of these zones (Chorley & Kennedy, 1971). Identifying the connectivity of these systems enables mitigation strategies to be implemented at optimal locations where nutrients can be reduced or restrained (e.g. intercepting the pathway, slowing the flow, removing some of the nutrients in the water) to minimise the impact on the receiving water body (Fenton et al., 2021). Research continues to help farmers to optimise farm management practices (baseline) and engineering solutions (above baseline) (Carstensen et al., 2020; Moore et al., 2010; Schoumans et al., 2014). Many open ditch studies have focused on nutrient dynamics (Sukias et al., 2003), sediment attenuation capacity (Ezzati et al., 2020; Mattila & Ezzati, 2022), nutrient loss attenuation potential by vegetation (Soana et al., 2017; Zhang et al., 2020), dissolved organic carbon dynamics (Tiemeyer & Kahle, 2014), organic matter composition (Hunting et al., 2016), ditch management (Dollinger et al., 2015; Hertzberger et al., 2019), and indirect greenhouse gas emissions (Clagnan et al., 2019; Hyvönen et al., 2013). However, few studies have investigated the role that open ditch connectivity plays in the transfer of nutrients from source to receptor. Such studies may provide vital information to ascertain the positioning of an engineered ditch mitigation option and the dominant nutrient species it is required to target. Moreover, there is a poor understanding of processes leading to the immobilisation and transformation of nutrients within soil and drainage systems along the hydrological pathways into ditches (Deelstra et al., 2014). For efficient mitigation of nutrient loss from open ditch networks, a conceptual understanding of how nutrient sources and their pathways connect to the open ditch system must be established.

The general trend and pathways of agricultural pollutants have been well documented and are summarised in Figure 3.1. In summary, nutrient entry into ditches is predominantly from diffuse sources, and often through complex surface and subsurface pathways determined by soil type, climate, landscape position, farm management, and nutrient input sources (fertiliser type) (Gramlich et al., 2018; Granger et al., 2010; Monaghan et al., 2016). These factors regulate the hydrology, the primary driver of nutrient transfer, and the terrestrial and aquatic biogeochemistry that defines the type and form/species of nutrients entering open ditches and subsequently discharging to associated water bodies (Sukias et al., 2003). Processes such as redox processes may also influence the ammonium concentrations (Pett-ridge et al., 2006) and nitrate concentrations (Bohrerova et al., 2004). Conceptually, P, either as PP or DRP, and N, as ammonium (NH_4^+) or nitrate (NO_3^-), are transported from fields or hard surfaces like roadways through surface flow pathways into open ditches (Figure 3.1).

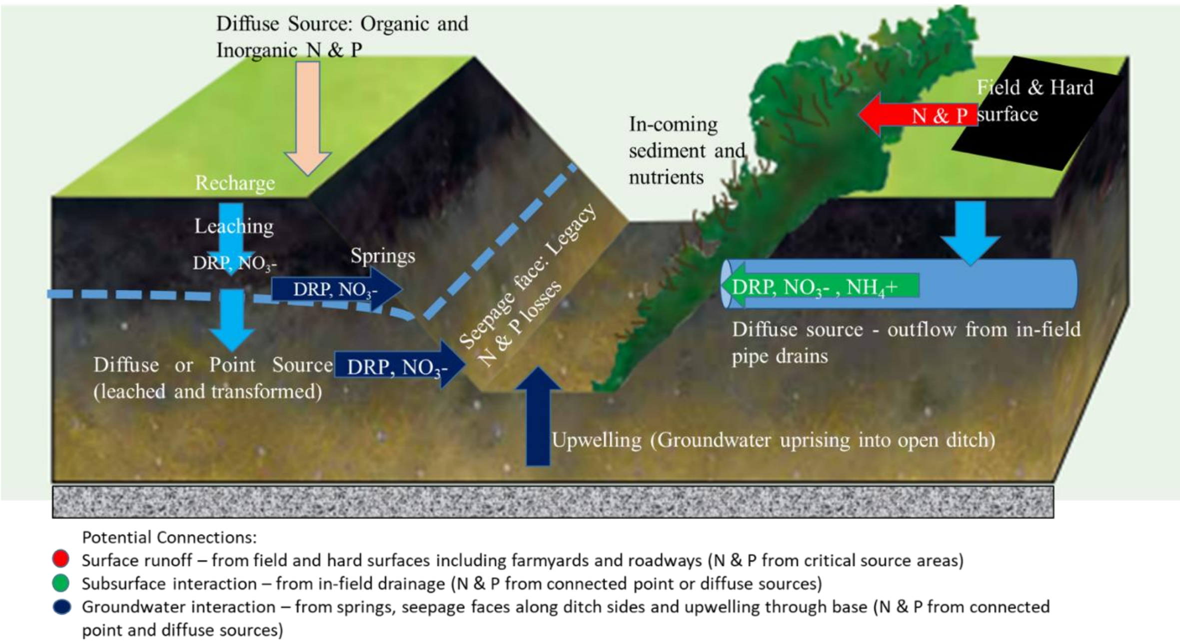


Figure 3.1 Conceptual figure of an open ditch showing all potential nitrogen and phosphorus sources (point and diffuse), pathways, and discharge connections [modified from Teagasc (2022) and Simpson et al. (2011)].

In Figure 3.1, any groundwater-to-open ditch water connection represents a subsurface interaction distinct from in-field drain connections. In this scenario, typically P is in the form of DRP and NO_3^- represents mineralised N that has become mobilised due to infiltrating water. This N is primarily lost from diffuse sources in fields due to fertilisation and grazing of animals.

Clagnan et al. (2018a) have shown N conversion to NH_4^+ in poorly drained soils, which can be discharged in waters from in-field drains within the groundwater-to-open ditch water connections (Needleman et al., 2007; Valbuena-Parralejo et al., 2019). The presence of NO_3^- in open ditch networks suggests more permeable connectivity pathways that eventually seep into open ditches along seepage faces or upwell as the water table rises, whereas NH_4^+ suggests less permeable routes before discharge occurs. Groundwater springs represent a distinct groundwater storage component that protrudes onto fields, which are often drained by the installation of an intersecting pipe into an open ditch below the spring. This creates a direct discharge point within the open ditch (Figure 3.1). The presence of this discharge may change during dry periods, as the water level falls below the base of the open ditch.

Moloney et al. (2020) used this concept to rank connectivity risk (from highest to lowest) for P along agricultural open ditches. The riskiest open ditches were those directly connected to farmyards (farmyard connection ditches) and watercourses (outlet ditches), while the least risky open ditches included secondary and outflow ditches (disconnected ditches did not pose any risk of connectivity). The system devised by Moloney et al. (2020) conceptualised P sources and pathways with the aim of disconnecting P losses before discharge to associated water bodies. The current study takes the same approach but creates an integrated connectivity risk ranking that considers both N, which discharges into the open ditch network via surface and subsurface pathways (Figure 3.1), and P. Such integration necessitates a thorough understanding of N and P biogeochemical cycles and an understanding of how sources are connected along different surface and subsurface pathways to the open ditch network, and how this network is connected and delivered to the adjoining aquatic system e.g. river. Accounting for attenuation along the pathway and within the open ditch network is a constraint within the current conceptual framework. Therefore, there is a need to integrate N into the connectivity risk ranking, so that a more holistic mitigation management strategy may be designed (i.e., source protection on the farm and "right measure, right place" in the open ditch).

The objective of this study was to derive a farm-scale integrated open ditch risk ranking for both P and N loss risk based on connectivity, to inform future mitigation management on heavy textured, grassland dairy farms. To fulfil this objective, seven farms were selected with open ditch networks on heavy textured soils. A conceptual figure illustrating trends and pathways of agricultural pollutants for an open ditch is presented (Figure 3.1). The open ditch networks were mapped during a ground survey, and a qualitative water sampling campaign was conducted (based on the conceptual figure) to validate the presence or absence of pathways for N and P. This enabled an integrated classification of an open ditch network ranking to be developed. Mitigation options for each ditch class are presented.

3.3 Materials and Methods

3.3.1 Site selection and characteristics

Seven grassland dairy farms on poorly drained soils geographically located across the SW and NE of Ireland were selected to represent a variety of agronomic dairy production systems and bio-physical settings (Table 3.1). As per the EPA soils and subsoils maps (Fealy et al., 2009), the soil types on these farms varied from organic to mineral soils. The majority of these farm fields were imperfectly or poorly drained, necessitating an ad-hoc network of artificial drainage installations on the farms. The grazing area of each farm ranged from 28 to 45 ha. Intensive dairy farm management practices were observed on all farms. Morgan's extractable soil P test (Morgan, 1941) was used to determine the agronomic excesses and deficiencies in plant available P for fields of each farm. Farms in this study were located in high rainfall areas with an average of 1092.5 mm. The average farm slope was measured on all seven farms, as it could influence open ditch connectivity.

Table 3.1 Summary of agronomic and soil data and associated in-field drainage percentages across case study farms.

Farm #	Farm size (ha)	Annual N stocking rate per ha (kg N /ha)	NUE[1] (kg N /ha)	% of number of fields with high P index[2]	Soil OM[3] (%)	Annual rainfall (mm)	Farm topography slope angle range (°)	Dominant Soil type	Drainage classes[4] (%)				Major soil type[4] (%)			% Fields with in-field drains[5]
									Poor	Imperfect	Moderate	Well	Mineral	Humic	Organic	
1	43	232.9	27	16.3	16.2	1086.3	2-3	Humic Surface Water Gley	30.9	52.9	16.2	0	69.1	30.9	0	48.4
2	40	263.5	23	40.0	16.7	1283.7	3-11	Humic Surface Water Gley	8.8	39.7	35.1	16.4	68.4	31.6	0	34.1
3	45	210.0	24	19.6	30.6	1002.4	0	Groundwater Gley	50.1	38.5	11.4	0	46.2	31.0	22.8	72.5
4	37	254.2	32	10.3	18.0	1320.2	4-8	Humic Brown Podzolic	45.1	0.9	54	0	58.4	41.6	0	13.6
5	41	291.6	35	59.4	8.4	900.0	0.6-0.9	Surface Water Gley	57.5	17.2	2.1	23.1	88.2	11.8	0	78.4
6	39	222.7	45	21.5	14.8	1035.6	1-8	Typical Surface Water Gley	42.1	3.5	25.1	29.3	84.3	10.9	4.9	25.2
7	28	327.3	42	41.7	12.1	1019.6	5-7	Typical Surface Water Gley	50.2	5.1	42.5	2.2	97.1	1.7	1.2	69.6

[1] Nitrogen use efficiency [2] High P index (Index 4) fields have soils with excess P concentration (above 8 mg L^{-1}, measured as Morgan's P, on grassland) [3] OM, organic matter (Corbett et al. 2022[a]; Corbett et al. 2022[b]) [4] Data from Tuohy et al. (2018, 2021) [5] % Field with in-field drain = (size of drained field / total farm size) × 100 %

3.3.2 Ground survey and mapping connectivity pathways for N into P connectivity risk ditch categories

A ground survey was carried out on all the farms during winter (November 2021 to March 2022) to characterise the field boundaries, surface and subsurface networks on each farm. This period was selected following multiple field visits carried out across all seasons in the previous year. This period was identified as the best hydrological period when connectivity pathways were active for grab sampling. Drainage network features such as open ditches connected to the farmyard, and the proximity of the open ditch to water bodies were noted on each farm during the ground survey. In addition, the connectivity pathways for N into open ditches from in-field drains, farm roadways, groundwater springs, seepage and upwelling as per the conceptual figure (Figure 3.1) throughout the drainage network were noted during this time. During the ground survey, all drainage network data such as drain locations, flows and connections, and sampling locations, were recorded using an electronic device with ESRI ArcGIS Field Maps mobile software (ESRI, 2024).

Open ditches were identified as man-made open drains usually sited along the field edges to carry excess water from the field and farm. Surface water bodies (1^{st} and 2^{nd} order streams) in and around each farm, defined as those appearing on the national ordnance survey maps (6-inch maps) (osi.ie), were mapped onto each farm map before each ground survey.

Information from the ground survey observations and qualitative interviews with farmers on drainage networks were used to digitise and map farm and field boundaries, and the open ditch network (open ditches, sub-surface in-field drains and drainage outlets) and associated connectivity pathways for N (Figure 3.2). For the open ditch network within each farm, each ditch was assigned a ditch category using their connection to a farmyard, watercourse, neighbouring farm, other ditches on the same farm and also their non-connection to any other part of the open ditch network after Moloney et al. (2020) (Table 3.2). These categories are: (1)

farmyard connection ditch (2) outlet ditch (3) outflow ditch (4) secondary ditch, and (5) disconnected ditch (Figure 3.2) using ArcMap GIS software (version 10.5).

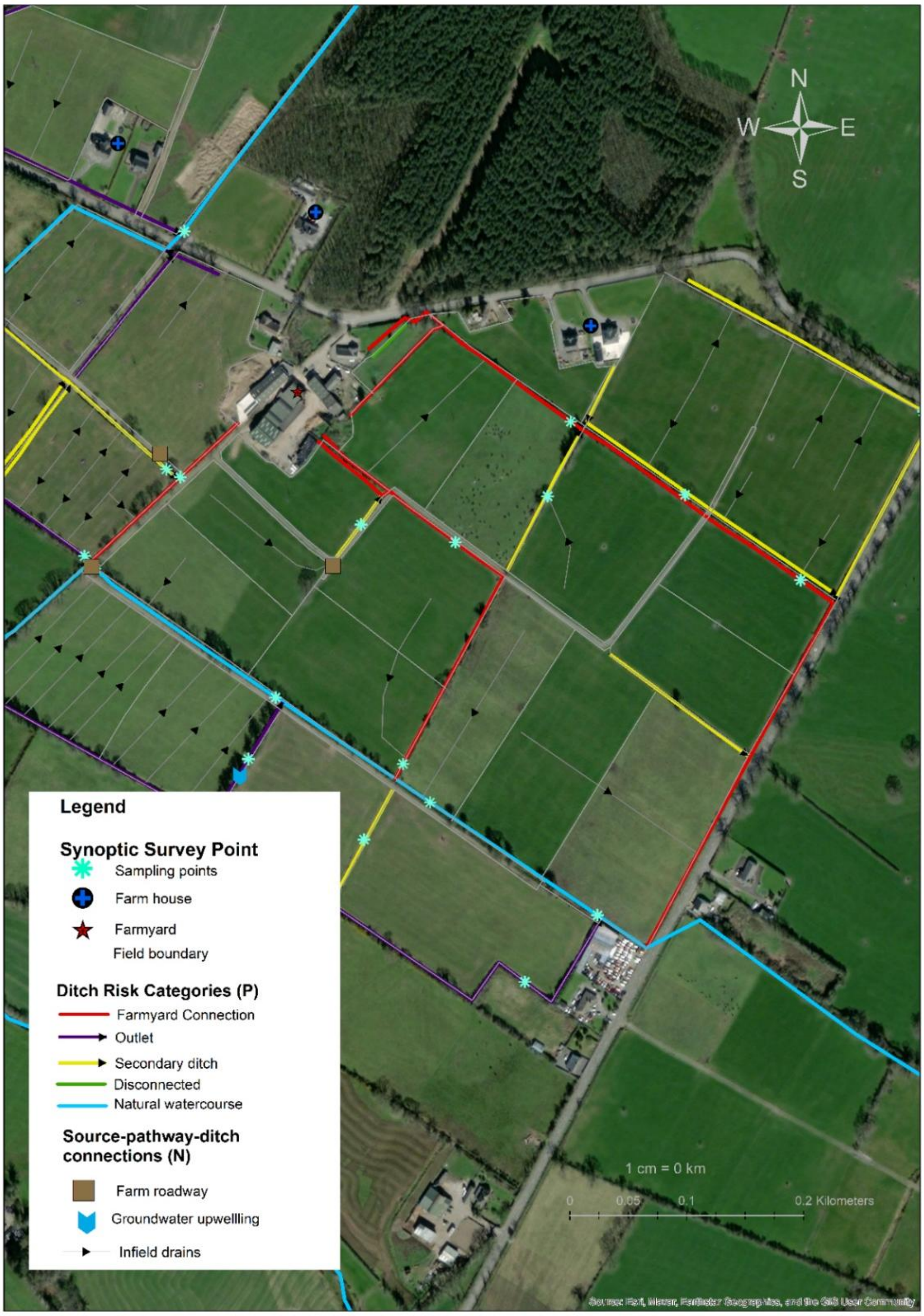


Figure 3.2 Example of a farm output map (for Farm 5) showing the ranked classification risk along the open ditch network for P (colour coded into categories of connectivity risk) and all conceptualised N open ditch connectivity pathways to individual open ditch sections. For in-field drains, arrows indicate fall and flow direction towards open ditch sections, with a particular P risk indicated by the existing colour coding scheme of Moloney et al. (2020).

Table 3.2 Definition and description of open ditch categories for the P classification system of Moloney et al. (2020).

Ditch category	Description
1. Farmyard	A ditch/pipe that connects a farmyard to the drainage connection network or directly to a surface water body.
2. Outlet	A ditch that connects the drainage network to a surface water body.
3. Outflow/transfer	A ditch that carries drainage water across the farm boundary onto neighbouring land.
4. Secondary	A ditch that typically flows perpendicular to the slope of the land connecting two larger open ditches or running through a field for excess water removal.
5. Disconnected	A ditch that is not connected to the overall drainage network but may have groundwater connectivity potential.

On each assigned ditch category, the connectivity pathways for N (Table 1.1), where present, were mapped within this open ditch network using the conceptual figure (Figure 3.1) as a guide during fieldwork to integrate N connectivity pathway risk into the P connectivity risk open ditch categories. To identify the connectivity pathways, landscape position was taken into account, especially for assessing groundwater interaction with an open ditch section. Groundwater seeping through open ditch bank sides and groundwater uprising through the base of the open ditch were identified as groundwater seepage and upwelling, respectively (Table 1.1), and were classified together as one connectivity pathway. Roadways were identified as a connectivity pathway when there were site observations of water flow and eroded/gully surface (due to continuous past water flows) from the farm roads into a nearby open ditch. Groundwater springs were identified as high-flow groundwater purging out into open ditches either over the surface or through pipes. Subsurface in-field drains were all piped drains directed into ditches but were differentiated from piped springs with their low and intermittent flows into the open ditches.

The length of the open ditches, and farm and field boundaries were measured in ArcGIS and compared for each farm in Table 3.3. In addition, the occurrence of a particular N connectivity pathway was calculated as a percentage of the total number of N connectivity pathways observed for each farm, and for each open ditch category.

Table 3.3 Summary of open ditch data including the proportion of the open ditch network accounted for by different P open ditch categories for each case-study farm.

Farm Number	Field perimeter (m)	% perimeter as ditch	Total ditch length (m)	Proportion of total ditch length (%)				
				1.Farmyard connection	2. Outlet	3. Outflow	4. Secondary	5. Disconnected
1	16471.5	44.3	7290.4	10.7	0	18.4	70.2	0.7
2	21524.1	9.0	1935.1	6.8	59.4	33.8	0	0
3	19737.9	35.4	6990.7	5.7	22.6	9.4	62.4	0
4	16572.3	17.2	2847.4	28.4	23.3	4.6	10.5	33.2
5	13085.9	43.5	5692.4	25.5	39.5	0	34.3	0.7
6	16966.5	52.6	8916.3	8.5	22.4	7.2	60.9	0.9
7	9607.5	28.9	2773.3	34.2	11.7	15.8	38.3	0
Average	16280.8	33.0	5206.5	17.1	25.6	12.7	39.5	5.1

3.3.3 Grab water sampling campaign to assess integrated nutrient connectivity pathways

Water quality parameters change over time, depending on the local climatic conditions and farming practices (Huebsch et al., 2013). In the present study, the objective was to establish a link or connection (see Figure 3.1) between the source and pathway to the open ditch network.

Therefore, “snapshot” sampling in spring (March) presented a good opportunity to collect qualitative data.

In spring (March) 2022, a total of 210 water samples were collected directly from 105 sampling sites in open ditches throughout the drainage network across all farms during a one-time sampling event following the procedure of Moloney et al. (2020). These sampling sites reflected connectivity pathways presented in Figure 3.1. March was selected for sampling because the period is hydrologically-active in Ireland and all pathways interact with the open ditch network (e.g. groundwater upwelling, seepage and springs) as observed from the previous year's field visits. As this study aimed to validate established connectivity risk (water and the presence or absence of N and P) between open ditch types and adjoining surface waterbodies, and did not aim to elucidate the load or impact of this connection, a temporal water sampling survey was not required. It is acknowledged that the connectivity level at the time of sampling water is influenced by the precipitation level (both antecedent and current). Therefore, sampling was undertaken when both surface and subsurface pathways were most active, and such data were used to validate source and hydrologic connectivity with the open ditch network.

The number of samples collected was dictated mainly by the observations of connectivity pathways on open ditches during the initial fieldwork campaign. As such, open ditches that had surface or subsurface connectivity pathways (Table 1.1) noted in the earlier survey were prioritised for sampling. These observations were used to validate surface, subsurface and groundwater flows that entered open ditches on the case study farms. However, some sampling points had no N connectivity pathways. Therefore, only four ditch categories from Table 3.2 (farmyard connection, outlet, outflow, and secondary ditches) were sampled for water across the seven case study farms. Shallow disconnected ditches (category 5 in Table 3.2) were dry, which indicated no N connectivity with perched or true water tables at the time of sampling.

These acted as storage and recharge areas for groundwater during rainfall periods. At each water sample location, two 50 ml samples (filtered on-site using 0.45 μm filter paper and unfiltered) were collected for dissolved and total P analyses, respectively. Grab sampling was carried out in the mapped ditch categories on each farm, provided water was present in the open ditch. The grab water sampling taken directly from an open ditch was conducted within 1 m downstream of in-field drain outlets, farm roadways, groundwater springs, and groundwater seepage/upwelling, where present, in the open ditch categories. All water samples were kept in an ice-box during sampling and transportation and then tested within one day of sample collection.

Filtered water samples were analysed for DRP and total dissolved phosphorus (TDP) using a Gallery discrete analyser (Gallery reference manual, 2016) and a Hach Ganimede P analyser, respectively. Total dissolved phosphorus (TDP) was measured by acid persulphate oxidation, under high temperature and pressure. The unfiltered water samples were analysed for nitrite (NO_2-N), NH_4-N, total oxidised nitrogen (TON), and total reactive phosphorus (TRP) using the Gallery analyser. Total phosphorus (TP) and total nitrogen (TN) was analysed using the Ganimede P analyser and Ganimede N analyser, respectively. Phosphorus was measured colourimetrically by the ascorbic acid reduction method (Askew and Smith, 2005), where the 12-molybdophosphoric acid complex is formed by the reaction of orthophosphate ion with ammonium molybdate and antimony potassium tartrate (catalyst) and reduced ascorbic acid. All samples, reagent blanks, and check standards were analysed at Teagasc Johnstown laboratory following the Standard Methods (APHA, 2005). All quality control (QC) samples/check standards are made from certified stock standards from a different source than calibration standards. Quality control samples were analysed at the beginning and end of every batch, and every 10 samples within a batch, and if the QC fell outside limits, samples were repeated back to the last correct QC. Blanks were included in every batch and approximately

10 % of samples were repeated. Tolerances range up to a maximum of ±7.5 % of nominal value. All instruments used were calibrated in line with manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, particulate phosphorus (PP) was the difference between TP and TDP, and dissolved unreactive phosphorus (DUP) was the difference between TDP and DRP.

3.3.4 Data Analysis

To validate the link between the conceptualised connectivity sources-pathways and their introduction of N and P into the open ditch system, data from the spring season synoptic survey were analysed statistically to differentiate the nutrient concentrations for the various open ditch categories and also for the various connectivity to ascertain if they varied from each other. As the data for each water quality parameter were not normally distributed, Kruskal Wallis analysis was undertaken to find out the significant differences between farmyard connection, outlet, outflow and secondary ditch categories, and also between the conceptualised N connectivity pathways (in-field drains, internal roadways, springs, and seepage/upwelling) within and across the outlet, outflow and secondary ditch categories for all the water quality parameters (NH_4-N, NO_3-N, TN, DRP, DUP, TP and PP). Organic N is transformed into inorganic N forms which are the readily available forms that impact water quality, and therefore organic N forms were not assessed in the study. Data were analysed using R studio software version 4.0.2 (2020). Where significant differences were observed using alpha level of 0.05 (95 % confidence level), the pairwise Wilcoxon Rank Sum test was further used to find the differences between the means of the pairs. Microsoft Excel software version 16.0 (2016) was used to find a correlation between the number of occurrences of in-field drains and the percentage of drained fields on poorly draining soil farms.

3.4 Results

3.4.1 Analysis of the open ditch networks

All five ditch categories, classified by Moloney et al. (2020), were identified using the criteria outlined in that work. Expressed as an average percentage of the total ditch network in all farms, 17.1 %, 25.6 %, 12.7 %, 39.5 %, and 5.1 % were farmyard connection, outlet, outflow, secondary, and disconnected ditches, respectively (Table 3.3). Farm 2 contained the fewest drainage categories (3 out of 5).

3.4.2 Observations relating to conceptualised N connections within the open ditch networks

Based on the criteria for identifying N connectivity pathways (Table 1.1), 52 % of all the open ditch network sampling points were observed to have N connectivity pathways interacting with them. The N connectivity pathways to open ditches considered in this study were mainly connected to secondary ditches, followed by farmyard connection, outflow, and outlet ditches, with no N connectivity pathway to disconnected ditches (Appendix B, Table B1). For each ditch category (Table 3.2) sampled in this study, the percentages of the different N connectivity pathways occurrence are shown in Figure 3.3. Among these N connectivity pathways across all ditch categories, in-field drains were the most common (representing 64 %), followed by groundwater springs, internal roadways, and groundwater upwelling/seepage, respectively, representing 20 %, 11 %, and 5 % of the sampling points (Appendix B, Table B1). The occurrence of observed in-field drains was positively correlated to the percentage of drained fields on case study farms (R^2=0.35).

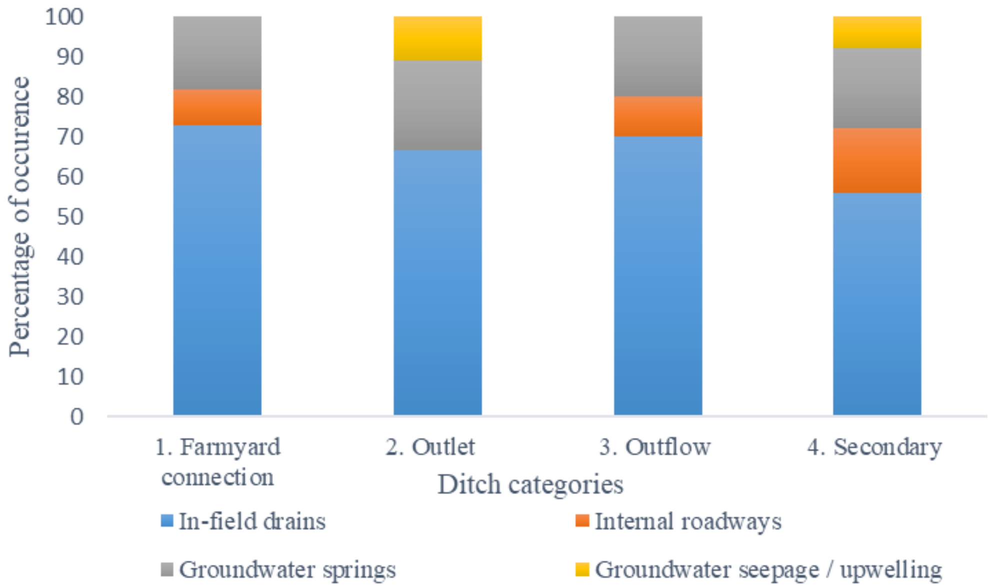
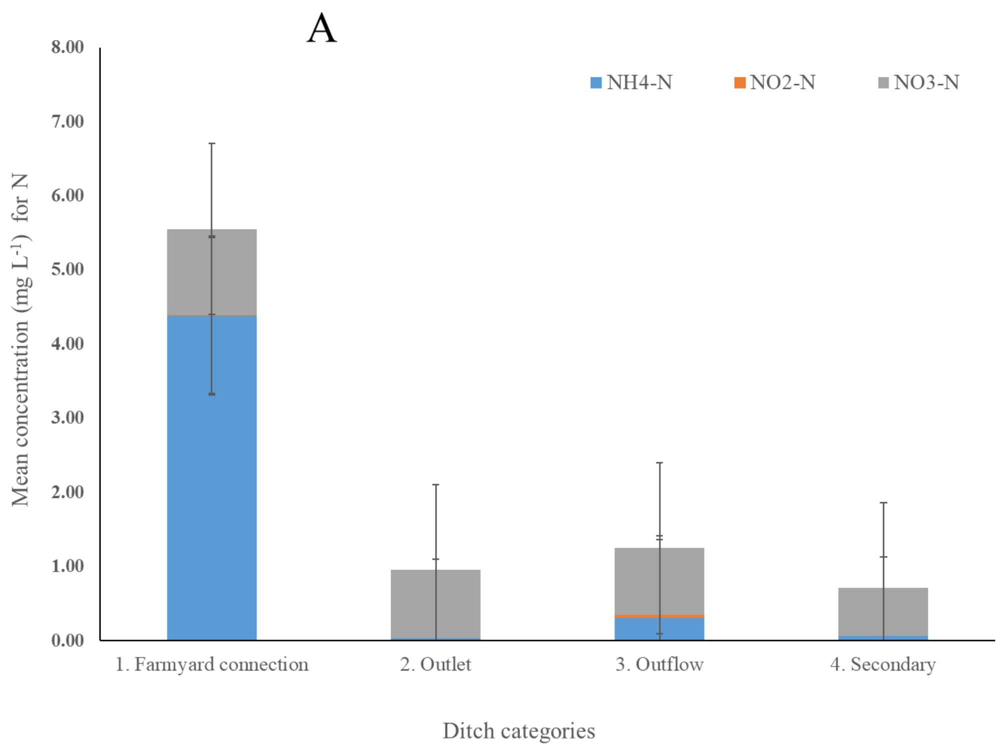


Figure 3.3 The percentages of the occurred N connectivity pathways for the ditch categories.

Farms 2 and 4, which had the lowest percentage of in-field drained fields (Table 3.1), had relatively high connectivity of groundwater springs to open ditches (Appendix B, Table B1). Aside from farm roadway connectivity pathways to open ditches on Farm 2, roadway connectivity pathway to open ditches was highest on farms with a flat topography, particularly Farms 3 and 5. Groundwater upwelling/seepage connectivity to ditches was uncommon. There was an absence of groundwater upwelling and seepage connectivity pathways on outflow and farmyard connection ditches, and roadway connectivity pathways on outlet ditches across all farms. In addition, there was evidence of multiple N connectivity pathways to individual ditches on some farms.

3.4.3 Validation of N connectivity pathway using synoptic survey

The average TN and TP concentrations were significantly higher in farmyard connection ditches (Figure 3.4) than in outlet, outflow and secondary ditches ($P < 0.01$). Across the outlet, outflow and secondary ditch categories, NO_3-N was the dominant N species, contributing on average to 44.7 % of TN at sampling points near N connectivity. Only 10.6 % of TN comprised NH_4-N within these ditch categories. The highest average NO_3-N across these ditch categories was observed in groundwater springs (1.90 mg L^{-1}), followed by in-field drains (0.75 mg L^{-1}), groundwater upwelling (0.65 mg L^{-1}), and roadways (0.17 mg L^{-1}) (Appendix B, Table B1). In addition, NO_3-N at groundwater springs were dissimilar ($P < 0.05$) to NO_3-N at roadways and in-field drains (Figure 3.5a). High concentrations of NO_3-N were also measured on roadways (Figure 3.5a), where NH_4-N is conceptualised as being dominant (Figure 3.1) on secondary ditches. However, NH_4-N dominated TN across these ditches at sample points near roadways, with 25.3 % composition as opposed to 6.9 % of NO_3-N. Ammonium-N concentrations across these ditch categories were not statistically significant ($P > 0.05$).



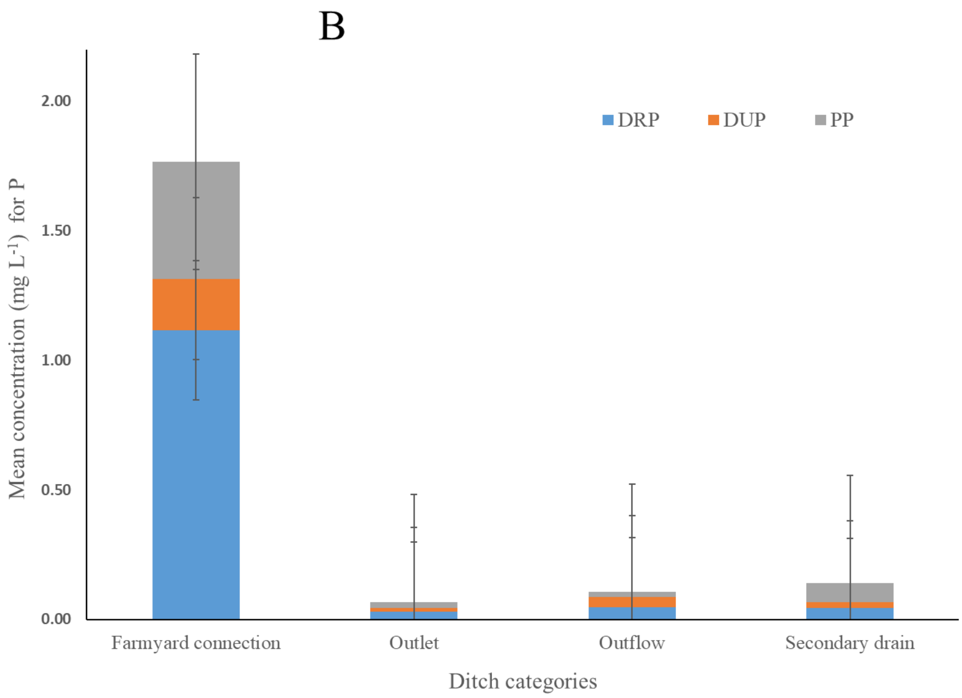
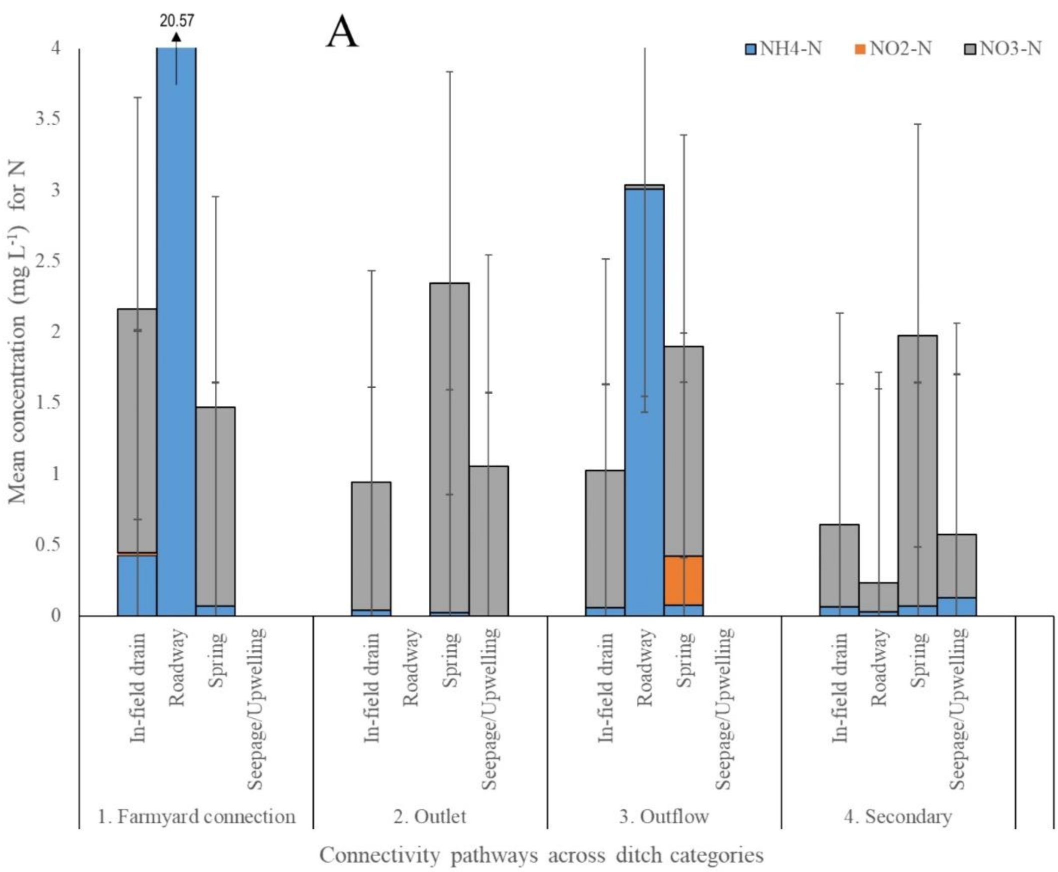


Figure 3.4 (A) Nitrogen (N) and (B) Phosphorus (P) mean ± standard errors (SE) concentrations within the open ditch categories across case study farms.



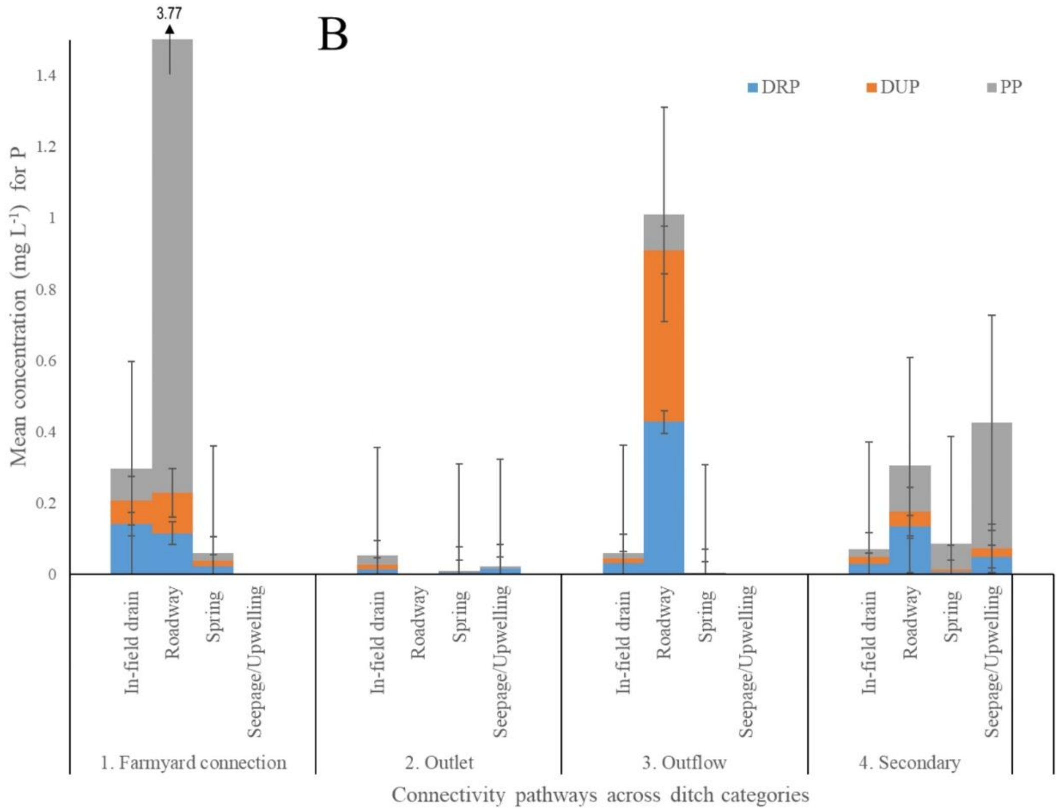


Figure 3.5 (A) Nitrogen (N) and (B) Phosphorus (P) mean ± standard errors (SE) concentrations within associated connectivity pathways in sampled open ditch categories across case study farms.

No consistent trends in species of TP were observed across the outlet, outflow and secondary ditch categories. Among these ditch categories, TP concentrations were relatively high in secondary ditches, in which PP was predominant (Figure 3.5b). Across the outlet, outflow and secondary ditch categories, PP was statistically significant ($P > 0.05$), particularly between in-field drain and roadway connectivity pathways, and DRP was statistically significant ($P > 0.01$), particularly between roadways and groundwater springs. Comparing P species for each N connectivity pathway, average PP concentrations were highest in groundwater upwelling/seepage (0.24 mg L^{-1}), followed by roadways (0.12 mg L^{-1}), groundwater springs (0.04 mg L^{-1}), and in-field drains (0.02 mg L^{-1}) connectivity pathways, whereas average DRP concentrations were highest in roadways (0.19 mg L^{-1}), followed by groundwater upwelling/seepage (0.04 mg L^{-1}), in-field drains (0.03 mg L^{-1}), and groundwater springs (0.01 mg L^{-1}).

3.5 Discussion

3.5.1 Observations on ditch categories and associated N connectivity pathways

Of the seven farms surveyed, disconnected and secondary ditches comprised the lowest and highest average percentage of the total ditch length, respectively. This result is consistent with Moloney et al. (2020), who recorded similarly low and high average percentages for total ditch length on varying soil grasslands in Ireland. Disconnected ditches are ineffective for excess field water removal within the drainage system, and exist either as blocked normal ditches or as created disconnecting ditches that remove field runoff or precipitation water by infiltration or evaporation. Disconnected ditches, when wet, may hold water with vegetation and potentially provide denitrification or create pollution swapping by the release of nitrous oxide (N_2O) or nitric oxide (NO) greenhouse gases.

Secondary ditches, as the most prevalent connectivity pathway, had multiple N connectivity pathways of which in-field drains were the most prevalent (Figure 3.3). Secondary ditches connect to other ditch categories from the central farm fields, and due to farm slopes, frequently have a shallow water table (Clagnan et al., 2018b). As the majority of the farms in this study contained poorly drained soils (Table 3.1), a positive, albeit weak, correlation (R^2=0.35) between the number of occurrences of in-field drains (Appendix B, Table B1) and the percentage of drained fields (Table 3.1) on poorly draining soil farms was observed. Both the number of occurrences of in-field drains and the percentage of drained fields help in regulating water table levels and supporting grass growth functionality, so they were positively correlated.

3.5.2 Hydrochemistry across P ditch categories and consideration of N connectivity pathways

Higher TN and TP average concentrations were measured in farmyard connection ditches relative to the other ditch categories, which was similar to the findings of Moloney et al. (2020), Harrison et al. (2019) and Ezzati et al. (2020). In the farmyard connection ditches, the TN and TP concentrations were nearly three times higher than the TN standard limits of 2.5 mg L^{-1} in the European Union for estuarine waters (Wuijts et al., 2022) and fifteen times higher for TP standards such as 0.1 mg L^{-1} as proposed by Wetzel (2001). While both Edwards et al. (2008) and Mockler et al. (2017) identified farmyards as point sources for high nutrient loss, the former argued runoff from farmyards has been overlooked and not duly considered as a major nutrient loss hotspot. Such runoff may lead to high nutrient-concentrated fields near the farmyard relative to fields further away (Fu et al., 2010), and these potentially may enter open ditches near the farmyard to create major downstream water quality problems. Unlike ditches (associated with point sources), the lower TP and TN concentrations in outlet, outflow and secondary ditch categories may be associated with diffuse nutrient sources. Studies have shown diffuse sources, relative to point sources, have lower TN and TP concentrations (Edwards & Withers, 2008; Pieterse et al., 2003). Management of some of these diffuse sources is problematic as they are difficult to locate in a landscape (Harrison et al., 2019). However, their impact on the deterioration of receiving water bodies is substantial and therefore needs to be managed (Andersen et al., 2014; Bradley et al., 2015). Diffuse sources depend on landscape and other management factors, which influence diffuse N and P mobilisation, transformation and delivery into the ditches (Granger et al., 2010; Schoumans et al., 2014). However, notable among these factors are the hydrological conditions, on which diffuse nutrient release strongly depends (Chen et al., 2013; Edwards & Withers, 2008). This, coupled with biogeochemical factors, which may vary within a landscape, influences the spatial and temporal distribution patterns of diffuse N and P, including the pathways by which they enter and leave farms (Clagnan et al., 2019; Grenon et al., 2021). Nutrient losses from the diffuse sources are delivered into open ditches along surface and subsurface pathways, creating hotspots of nutrient loss in certain open ditch categories, which need to be characterised and potentially mitigated. Climatic, landscape and management factors all have a role to play in when and where impacts occur. These could have contributed to the higher TN concentrations in water samples that were measured near N connectivity pathways than at locations with no N connectivity pathways within the outlet, outflow and secondary ditch categories, and also for TP in the outflow ditch category. This observation aligns with the reported works of Ibrahim et al. (2013) and Valbuena-Parralejo et al. (2019) on in-field drains, Fenton et al. (2021) and Rice et al. (2022) on roadways, Soana et al. (2017) on groundwater springs, and O'Callaghan et al. (2018) on groundwater upwelling/seepage.

Nitrate was the dominating N species in in-field drains, groundwater springs, and upwelling connectivity pathways in outlet, outflow and secondary ditch categories (Figure 3.5a). This may be attributed to their connection to a subsurface N source, which comprises leached N from animal excreta and fertiliser that may have been nitrified to NO_3-N (Necpalova et al., 2012). In poorly drained grasslands, nitrification may have been elevated by the high in-field drainage density (Table 3.1), which enhanced N preferential flow (Van Der Grift et al., 2016) and limited potential N attenuation (Clagnan et al., 2019; Valbuena-Parralejo et al., 2019). The average NO_3-N concentration was highest in groundwater springs and in-field drains. Factors such as the presence of these N connectivity pathways within the shallow subsurface region, nearness to the soil surface (where farm management mostly occurs), and exposure to N sources at the groundwater-ground surface intersection spots (particularly for groundwater springs; Infusino et al., 2022), could have contributed to the high NO_3-N concentrations in these locations. In contrast, NH_4-N was the most dominating N species measured for roadway connectivity pathways across the outlet, outflow and secondary ditch categories, especially where physical animal excreta were observed (Table B2). This observation aligns with Fenton et al. (2021), who observed that roadways draw surface nutrient sources, high in NH_4-N, as runoff from soil-bound and animal excreta into nearby ditches and streams. Although important, redox reactions were not considered in the present study.

For TP concentrations across outlet, outflow and secondary ditch categories, P concentrations were relatively low compared to the farmyard connection ditch category. However, such TP concentrations in the outlet, outflow and secondary ditch categories were still high enough to cause eutrophication downstream if undiluted. High TP concentrations measured in secondary ditches may be related to the impacts of farm management activities including grazing and farm machinery movement, which is intense within the central fields of most farms where secondary ditches lie as connecting ditch links. These contribute to the erosion of ditch sides and associated deposition of soils in the secondary ditches, as reflected in the higher PP concentrations observed. High TP concentrations measured near roadways on outflow ditches may be due to animal excreta and poached surfaces (personal observations), run-on deposits from farmyards and fields, as a result of animal and machinery movement (Fenton et al., 2021). Both PP and DRP can trigger eutrophication in waterbodies and may pose risk to downstream water bodies. However, this depends on their closeness, connection, and mitigation along the pathway to water sources within agricultural landscapes.

Such information from the study provides additional insight into the source, connection and presence (and transformation process) of N in ditch categories from a previous study by Moloney et al. (2020), who observed high NH_4^+ and NO_3^- concentrations in all ditch categories except for the outlet ditch, where high NO_3^- and low NH_4^+ were measured, and disconnected ditches where NO_3^- dominated. The risk ranking of connectivity along the open ditch for N and P does not determine the impact of the nutrients being lost to the associated water body; it simply establishes the N connectivity pathway if it is present.

3.5.3 Deriving a connectivity risk for N into P agricultural open ditch categories

The evidence of N concentrations in the ditch water chemistry from Moloney et al. (2020) and the current study informs an improved ditch connectivity risk category system (Table 3.4). This is a valuable information tool for environmental sustainability officers to enhance water quality management and mitigation options for N and P losses on dairy grassland farms with heavy textured soils in high rainfall areas. It considers both the connectivity pathways, through which N can be introduced to a ditch network, and their associated N species.

In the current study, all of the conceptualised N connectivity pathways (Figure 3.1) established from the literature were present, but not in all of the sampled P risk ditch categories developed by Moloney et al. (2020) (Appendix B, Table B1). For instance, the established general trends and connectivity pathways of groundwater seepage and upwelling were not present on farmyard connection and outflow ditches. Moreover, the grab water data results validated all the conceptualised N connectivity pathways present in ditches (Figure 3.5a), except groundwater seepage and upwelling. The dominance of high NO_3-N concentrations at in-field drains and springs, and high NH_4-N concentrations at roadways within farmyard connection ditches, indicated a point pollution source arising from their connection to the farmyard aside from the hydrology-induced N concentrations. Farmyards pose the greatest nutrient loss risk on farms due to high nutrient concentration within discharges (Vedder, 2020) and like other point sources, they are independent of hydrology (Edwards & Withers, 2008). As such, primarily managing the farmyard wastewater before discharge into connecting ditches for mitigating nutrient connectivity to water sources is essential (NFGWS, 2020) before deployment along/within ditches interventions.

For the other sampled outlet, outflow and secondary ditch categories, all N conceptualised pathways were observed, except for internal farm roadway on outlet ditches, and groundwater seepage and upwelling on outflow ditches (Appendix B, Table B1). In outlet, outflow and secondary ditch categories, the ditch water synoptic data validated the conceptualised NO_3-N and NH_4-N for all the observed N connectivity pathways, except farm roadway connection on secondary ditches (which was invalid with NO_3-N dominance over conceptualised NH_4-N from hard field surface flow pathways). Nitrate dominated in-field drains, groundwater springs, upwelling and seepage connectivity pathways, and NH_4-N-dominated farm roadways across the outlet, outflow and secondary ditch categories, as conceptualised in Figure 3.1.

Assessment of N connectivity pathway within ditch category 5 could not be included in the study due to the unavailability of water samples in this ditch for validating conceptualised N connectivity pathways. Moloney et al. (2020) showed that disconnected ditches were the least risky ditch class for nutrient loss and therefore merit less focus during nutrient loss mitigation for surface water. However, such low nutrient concentrations could be leached into groundwater and therefore may require mitigation interventions to prevent leaching.

To apply this research in practice, once open ditches are investigated and mapped, a category should be assigned for an individual open ditch, after which the available N connections for that ditch are noted. All of these connections in combination will aid in the future mitigation management strategy. It is unlikely, for example, that more than one mitigation option will be installed in a single open ditch. Therefore, the information gathered from Table 3.4 can be used to ensure that the correct nutrients and their speciation are targeted for mitigation in the open ditch. Mitigation options may be a combination of those that limit diffuse and point sources. For example, with respect to diffuse sources, strict adherence to action programmes to reduce losses is important (e.g., Good Agricultural Practice Regulations, in line with the Nitrates Directive (91/676/EEC)). With respect to roadway runoff, NH_4^+ mitigation options are available and have been outlined in Fenton et al. (2021) and Rice et al. (2022) (e.g., diversion bars to move runoff to a buffer area of at least 1.5 m, cambering farm roadways, and directing flow onto adjacent fields). Adopting a two-stage ditch design may reduce high PP concentrations (Faust et al., 2018; Hodaj et al., 2017; King et al., 2015). With respect to the subsurface N connectivity pathways (in-field drains, groundwater springs, upwelling and seepage), in-ditch management practices may control the flow and the nutrient content leaving the open ditch. These may include sediment traps (Wilkinson et al., 2014), vegetated ditches (Faust et al., 2018; Kröger et al., 2008; Soana et al., 2017) or in-ditch filters or bioreactors (Goeller et al., 2020; King et al., 2015; Liu et al., 2020). Nutrient filtering through vegetation (Moeder et al., 2017) or use of media (Ezzati et al., 2020) can only aim to mitigate a small amount of overall nutrients leaving the ditch due to hydraulic retention times needed and by-pass flow during high storm events. Furthermore, mitigation practices including the construction of wetlands (Tanner et al., 2005), vegetated buffer zones (Faust et al., 2018) and low-grade weirs (Baker et al., 2016; Kröger et al., 2012; Littlejohn et al., 2014) that may be placed at the end of ditches after the connectivity pathways, especially for farmyard connection and outlet ditch categories, would help to limit nutrient loss from these farms. Therefore, all measures need to be considered as a package and not in isolation when trying to minimise nutrient and sediment loads leaving an open ditch system. It is worth noting that co-operation at the local level is needed to prevent other mitigation-related problems (such as the polluter pays principle regarding outflow ditches between neighbouring farmers) to ensure mitigation occurs before waters are impacted.

Table 3.4 An updated integrated ditch connectivity ranking that considers both phosphorus and nitrogen coupled with suggested strategies to reduce nutrients from ditches on dairy farms.

P Ditch Category	Description	Validated N Connection with Category	Associated Source	Future Mitigation Management
1.Farmyard Connection	A ditch/pipe that connects a farmyard to the drainage network or directly to a surface water body. These connections pose the highest risk and should be prioritised in terms of future management.	Subsurface interaction	In-field drains (pipes; moles; gravel moles; older variation) bring P and N from fields to the open ditch. All forms of P and N are potentially lost through this pathway to the ditch, with NO_3^- and DRP dominating.	Management practices that disconnect sub-surface drainage system discharges into the open ditch: • These may include adherence to correct land drainage design, installation guidelines and maintenance. • Use of end-of-pipe land drainage mitigation options including low grade weirs (Baker et al., 2016), filter cells, cartridges, and structures (Goeller et al., 2020; King et al., 2015; Liu et al., 2020) (see discussion for details). Strict adherence to good farming practices to minimise diffuse losses and leaching of nutrients to sub-surface drainage system that are connected to the open ditch: • These may include in-ditch measures such as sediment traps, bioreactors, and filters to slow the flow and control nutrient loads (Fenton et al., 2021).
		Surface runoff	Farmyards and hard surfaces including farm internal roadways bring P and N forms, dominated by NH_4^+ and PP from raw organic waste, loss to the ditch	Management practices that disconnect the farmyard from the open drainage ditch and internal farm roadway network are needed specifically within 100 m of the farmyard in this category: • These may include measures that prevent roadway runoff from entering the open ditch using low-cost diversion bars or surface modifications (Fenton et al., 2021). There must be a buffer of at least 3 m (EPA, 2020) to reduce runoff impacts surface waters.
		Groundwater interaction	Natural springs bring shallow groundwater P and N, dominated by NO_3^-, into open ditches through piped drains.	Strict adherence to good farming practices to minimise diffuse losses: • These may include end-of-pipe mitigation measure where spring has been piped e.g. vegetated buffer spots (Faust et al., 2018) and filter cells, cartridges, and structures using various materials (Ibrahim et al., 2015; King et al., 2015; Penn et al. 2020) (see discussion for details). Full list of materials is reviewed in Ezzati et al. (2020).
2. Outlet	A ditch that connects the drainage network to a surface water body.	Subsurface interaction	In-field drains (pipes; moles; gravel moles; older variation) bring P and N forms, dominated by NO_3^-, from fields to the open ditch.	Management practices that disconnect sub-surface drainage system discharges into the open ditch: • These may include adherence to correct land drainage design, installation guidelines and maintenance. • Use of end-of-pipe land drainage mitigation options such as constructed wetlands (King et al., 2015; Tanner et al., 2005) (see discussion for details)

				Strict adherence to good farming practices to minimise diffuse losses and leaching of nutrients to sub-surface drainage system that are connected to the open ditch: • These may include in-ditch measures such as sediment traps, bioreactors, and filters to slow the flow and control nutrient loads (Fenton et al., 2021).
		Groundwater interaction	Natural springs bring shallow groundwater, dominated by NO_3^- concentration, into ditches through piped drains.	Strict adherence to good farming practices to minimise diffuse losses: • These may include end-of-pipe mitigation measures where spring has been piped e.g. vegetated buffers (Faust et al., 2018) and filter cells, cartridges, and structures using various materials (Ibrahim et al., 2015; King et al., 2015; Penn et al., 2020) beneath piped springs location on ditch. Full list of materials is reviewed in Ezzati et al. (2020).
		Groundwater interaction	Seeping and upwelling deep groundwater, dominated by NO_3^-, enters into ditches.	Strict adherence to good farming practices to minimise diffuse losses: • In terms of groundwater up-welling or spring connectivity in-ditch intervention that slows the flow and mitigates nutrients using bioreactors, two-stage ditch, filters and vegetated ditches (Faust et al., 2018; King et al., 2015) may be introduced after spring connectivity and before the outlet to reduce dissolved and particulate nutrients entering waters.
3. Outflow/transfer	A ditch that carries drainage water across the farm boundary through neighbouring land.	Subsurface interaction	In-field drains (pipes; moles; gravel moles; older variation) bring P and N, dominated by NO_3^-, from fields to the open ditch.	This drainage water will pass to an adjoining farm and will be mitigated as another landowners Farm Management Plan. Some mitigation can occur in Outflow ditches using mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across the farm landscape.
		Surface runoff	Farm internal roadways introduce NH_4^+ and DRP-dominated hard surface water to the ditch	This drainage water will pass to an adjoining farm and will be mitigated as another landowners Farm Management Plan. Some mitigation can occur in Outflow ditches using mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across the farm landscape.
		Groundwater interaction	Natural springs connect shallow groundwater, dominated by NO_3^- concentration, into ditches	This drainage water will pass to an adjoining farm and will be mitigated as another landowners Farm Management Plan. Some mitigation can occur in Outflow ditches using mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across the farm landscape.
4. Secondary	A ditch that typically flows perpendicular to the slope of the land connecting two larger ditches. Can also	Subsurface interaction	In-field drains (pipes; moles; gravel moles; older variation) bring P and N, dominated by NO_3^- from fields to the open ditch.	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in Secondary ditches using in-ditch mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm.

	occur as an open ditch running through a field to collect and remove large excesses of surface water	Surface runoff	Farm internal roadways introduce PP, DRP and NO_3^- dominated within the water from hard surface to the ditch	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in Secondary ditches using in-ditch mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm.
		Groundwater interaction	Natural springs bring shallow groundwater, dominated by NO_3^- concentration, through piped drains over ditch sides to introduce both PP and NO_3^- into the ditch	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in Secondary ditches using in-ditch mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm.
		Groundwater interaction	Deep groundwater, dominated by NO_3^-, seeps through ditch side surfaces and/or upwells through ditch base to introduce PP and NO_3^- into ditches	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in Secondary ditches using in-ditch mitigation management practices provided for Farmyard Connection and Outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm.
5. Disconnected	A ditch that is not connected to the overall ditch network. May be connected with groundwater.	Surface and Groundwater interaction	Diffuse source of NO_3^- interacts with open ditch. Runoff may interact with the open ditch.	Connectivity is not present to surface water within the open network but there may be a groundwater connection which subsequently discharges to surface water. Precautionary practices should be taken at these locations to minimise recharge to groundwater by provision of a soil buffer.

3.6. Conclusion

Distinctly different from a P-only classification system, the integrated connectivity risk classification system for N and P showed that not all source-pathway interactions within open ditches are active. This is a valuable information tool that enables a much more specific and targeted nutrient-specific mitigation approach to be implemented on open ditches in heavy textured grassland dairy farm in high rainfall areas. The new system avoids the pitfalls of a P-only classification system (i.e. mitigating for P but allowing N to affect water quality unabated). The findings of this study are limited to these field sites, and may (or may not) differ in other geographic areas with different soils, climates, agricultural practices, etc. However, the same methodology may be applied to other areas to develop a bespoke integrated connectivity risk ranking for P and N along agricultural open ditches to inform targeted and specific mitigation strategies on those farms. Further assessment of the temporal and spatial variability of soil, weather, drainage system, and general hydrogeochemistry, which influences nutrient connectivity, may be needed to rank the N and P risk in each ditch category.

3.7 Summary

This chapter developed a farm-scale, integrated risk ranking for P and N losses through open ditches, based on hydrological connectivity for enhanced tailored mitigation. The study confirmed the presence or absence of N and P transport pathways using a conceptual understanding of hydrological flows. Nutrient loss risks vary for every open ditch within a ditch network and may be influenced by spatially varying factors. Using this information, all spatially varying nutrient loss contributing factors along the NTC will be risk-assessed to identify high-risk drains and improve nutrient loss risk categorisation for open ditches with the ditch network in Chapter 4.

4. A semi-quantitative risk model for dairy farms to pinpoint and break source-pathway connections between nutrient sources and open drainage channel sections.

4.1 Overview

The aim of this chapter is to assess the nutrient loss influencing factors to identify risky surface drains and establish key influencing factors within surface drainage network.

Parts of this chapter have been published in Frontiers in Environmental Science (Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T. & Tuohy, P. (2024). A semi-quantitative risk model for dairy farms to pinpoint and break source-pathway connections between nutrient sources and open drainage channel sections. *Frontiers in Environmental Science*, *12*(1435418), 1–13. https://doi.org/10.3389/fenvs.2024.1435418)

To avoid repetition, acronyms that have already been defined in preceding chapters are not defined in this chapter. Citations to papers published by the author as part of this thesis are referred to by Chapter number.

4.2 Introduction

Agricultural landscapes in areas of high annual precipitation and heavy textured soils are characterised by high densities of open drainage channels, which provide outfalls for in-field drainage systems (Shore et al., 2015; Tuohy et al., 2018). Open drainage channels, comprising drainage ditches and smaller streams, are networked to collect and drain away excess water from different parts of a farm to larger water courses (Kröger et al., 2007). Within the open drainage channel network, streams exist as intermittent or perennial natural channels, whereas drainage ditches exist as man-made channels that may be intermittent or perennial, depending on their landscape position and their interplay with subsurface water and groundwater. These open drainage channels perform many functions (Daly et al., 2017; Ezzati et al., 2020) including storage and release of nutrients by sediments, transportation and interception of farm surface and subsurface runoff which may carry nutrients to the larger water courses.

It is important to minimise the source of nutrients and intercept instantaneous and legacy nutrients from farms in high rainfall areas (Fenton et al., 2021; Peyton et al., 2016; Valbuena-Parralejo et al., 2019). In these areas, open drainage channels form an integral part of the S-M-P-R component of the nutrient transfer continuum (Haygarth et al., 2005) (defined as the framework that captures the nutrient-loss influencing factors from source to receptor). Water drained in both natural and man-made open drainage channels may be nutrient-rich from different nutrient sources that are mobilised through point (e.g. farmyard (Martínez-Suller et al., 2010; Vero et al., 2020), farm roadway (Fenton et al., 2021; Rice et al., 2022) and diffuse (Daly et al., 2017; Roberts et al., 2017)) sources. Where hydrological connectivity exists with the surrounding environment, nutrients from these sources travel through different pathways (Wall et al., 2011) to enter open drainage channels. The nutrients are either transformed or remain unchanged along the pathway to the open drainage channel, before being transported to the adjoining waterways (Clagnan et al., 2018a). Aside from nutrient transformation, these nutrients can be buffered and/or retained to prevent connectivity losses as they go through the processes and pathways (Deelstra et al., 2014). Understanding the nutrient dynamics and loss risks occurring within an open drainage channel system is critical to assessing, managing and mitigating nutrient losses from farms (Collins et al., 2016; Herzon and Helenius, 2008).

Moloney et al. (2020) ranked connectivity risk for P loss along man-made open drainage channels and showed that varying levels of connectivity to nutrient source, depending on their geographical position, exist between man-made open drainage channels and surface waters. The highest to lowest connectivity for P loss was as follows: farmyard connection ditch, outlet (a ditch that connects the drainage network to a surface water body), outflow (a ditch that carries drainage water across the farm boundary through neighbouring land), secondary, or disconnected ditch. Chapter 3 further developed this concept by creating an integrated (i.e. P and N) ranked connectivity risk incorporating nutrient loss from sources within open drainage channels. That study showed that other factors i.e. farm management practices, landscape characteristics, and surface and subsurface hydrological connectivity of directly connecting areas, described the risk of P and N loss in categories of man-made open drainage channels. These factors vary spatially and temporally (Harrison et al., 2019; Mellander et al., 2017; Withers and Lord, 2002), even in a very small distance (Adams et al., 2022), and therefore may vary in the nutrient loss risk they pose for individual open drainage channels at different geographic locations on farm. Characterising these factors for individual open drainage channels is essential to assess the risk of connectivity for nutrient losses from an open drainage channel network, but is not well studied. In previous nutrient loss risk studies, open drainage channels were risk assessed largely as a (transport) pathway factor for nutrient loss based on either their presence, density, connectivity to high-risk fields or sloping conditions (Buczko & Kuchenbuch, 2007; Magette et al., 2007; Roberts et al., 2017; Schoumans & Chardon, 2003), thereby limiting a holistic assessment (Granger et al., 2010). Furthermore, in studies where these factors have been used in assessing farm nutrient loss connectivity (Deelstra et al., 2014; Gramlich et al., 2018), their influences on connectivity to open drainage channels under their respective nutrient transfer continuum sections to enable complete understanding of their nutrient loss risks (Haygarth et al., 2005; Murphy et al., 2015) and improve regulations (Wall et al., 2011) have not been evaluated. Such an evaluation could be achieved by exploring a risk assessment of the factors under the nutrient transfer continuum of open drainage channels and may allow mitigation efforts to be optimised to prevent nutrient losses to open drainage channels and transfer to adjoining water bodies.

Risk assessment provides an overall appraisal of the connectivity components for each element (S-M-P-R) of the nutrient transfer continuum to inform their combined implications and relationships for nutrient loss to open drainage channels on farms (Jordan et al., 2005). Risk can be assessed quantitatively (where data are sufficient; Adkin et al., 2014), qualitatively (where data are insufficient; Nag et al., 2020), and semi-quantitatively (a blend of the two e.g. Rice et al., 2022)). Subjective expert judgment may be used to approximate risk values to inform decision-making (Redmill, 2002; Rice et al., 2022). Different assessment approaches to identify and characterise landscape hotspots for nutrient losses have been documented. These include direct nutrient concentration measurements in open drainage channels (Ezzati et al., 2020; Mattila and Ezzati, 2022), a combination of some nutrient transfer continuum parameters (Alder et al., 2015; Fenton et al., 2022; Hayes et al., 2023), or predictive models (Radcliffe et al., 2015; Vadas et al., 2007; Vadas et al., 2015). A risk assessment to identify open drainage channel sections associated with high-risk nutrient runoff connectivity using all possible field management data, and landscape and hydrological connectivity data across the nutrient transfer continuum for heavy textured farms has not been developed to date. Undertaking an appraisal incorporating these elements will help identify and rank high-risk areas (also known as critical source areas; McDowell et al., 2024) on the open drainage channel network for heavy textured grassland dairy farms for targeted mitigation.

The objective of this study was to develop a semi-quantitative risk model for heavy textured grassland dairy farms that identifies open drainage channel network sections that pose a risk of contributing nutrients to the adjoining aquatic water courses and which require mitigation. Instead of considering only nutrient source connectivity to classify open drainage channel risks for nutrient losses (see Chapter 3), the current study builds on this theory and captures all relevant S-M-P-R factors under the open drainage network nutrient transfer continuum to rank the nutrient loss risk in the open drainage channel network on a farm. To conduct this research, data were collected during field and desk-based studies across seven heavy textured grassland farms in Ireland. These farms are considered representative of heavy textured, poorly draining soils in Ireland, all receive high rainfall and were subjected to high-resolution data collection on a vast range of static and dynamic variables related to farm management.

4.3 Materials and methods

4.3.1 Nutrient transfer continuum framework

A semi-quantitative risk assessment model was developed based on seven intensive grassland heavy textured dairy farms. Using expert opinion and the literature, various parameters that best describe the nutrient transfer continuum between a source and an open drainage channel network (Dollinger et al., 2015; Kleinman et al., 2011; Needelman et al., 2007) were collated and categorised into S-M-P-R components as in Table 4.1.

Table 4.1 Nutrient transfer continuum element, parameter description, units, type, relative magnitude score, relative impact score, and denotation.

Nutrient Transfer Continuum element	Parameter Description	Parameter unit	Parameter type	Relative Magnitude (M) score [1]	Denotation	Relative Impact (I) score[2]
Source (Point)	Connection to farmyard		Categorical	0 3	No Yes (e.g. pipe discharge, seepage from leaking tanks)	10
Source (Diffuse)	Soil P	mg/l	Categorical	1 3	Adequate (<8.0 m/l) Excessive (≥ 8.0 mg/l)	5
Source (Diffuse)	N Fertiliser (kg) applied	kg N ha^{-1}	Continuous	Weighted to 0 – 3		8
Source (Diffuse)	P Fertiliser (kg) applied	kg P ha^{-1}	Continuous	Weighted to 0 – 3		8
Source (Diffuse)	Nutrient deposition associated with grazing (e.g. urine, dung pats)	Grazed or non-grazed field × grazing frequency	Continuous	Weighted to 0 – 3 *(Based on grazing field (1 = not grazed, 3 = grazed) × grazing frequency)*		6
Source (Diffuse)	Fertiliser application count	# *per* field	Continuous	Weighted to 0 – 3		3
Mobilisation	Rainfall	mm	Continuous	1 2 3	Low (<1000 mm) Moderate (1000-1300 mm) High (>1300 mm)	10
Pathway	Farm roadway runoff		Categorical	0 1 2 3	No[3] Yes – flat slope Yes - moderate slope Yes – steep slope	4
Pathway	Farmyard surface runoff		Categorical	0 1 2 3	No[3] Yes – flat slope Yes - moderate slope Yes – steep slope	3
Pathway	Field surface runoff		Categorical	0 1 2 3	No[3] Yes – flat slope Yes - moderate slope Yes – steep slope	6
Pathway	Subsurface connection from infield drains		Categorical	0 3	No Yes (e.g. low flow discharge from pipes)	4

Pathway	Groundwater connection to ditch		Categorical	0 3	No Yes (e.g. springs, upwelling and seepage)	3
Receptor	Connection to watercourse		Categorical	0 3	No Yes	7

[1] Relative Magnitude score (M) = the relative magnitude of contributing nutrients to an open drainage channel network.
[2] Relative Impact score (I) = subjective evaluation of relative relevance (on a 1 – 10 scale) for nutrient contribution to an open drainage channel network.
[3] A barrier e.g. buffer prevents connectivity of this runoff according to EPA (2020) and USDA (2001) with the surface water (man-made or natural) body.

4.3.1.1 Justification to S-M-P-R parameters

a. Source

In a nutrient loss risk assessment, identifying potential sources and their characteristics is critical (Carton et al., 2008; McDowell et al., 2024). Farmyards are largely associated with potential nutrient sources, and connection to them imposes high-risk of direct or indirect discharges of point source nutrients into the open drainage channel network (Moloney et al., 2020; Vero et al., 2020). Soil P status of fields directly connected to open drainage channels offers a potential source contribution of soil nutrients that can be readily lost, and dictates the amount of P that can be applied in a mineral or organic soil (Moloney et al., 2020), and is therefore essential as a source parameter. The organic matter proportion in mineral and organic soils determines the adsorption or repulsion of dissolved nutrients unto soil particles (Roberts et al., 2017; Tejada & Gonzalez, 2008) and therefore influences the soil P status. Soil P Indices of 1, 2 and 3 are defined as low risk, while index 4 is defined as high-risk, with all organic soils categorised as index 4 by default (Daly, 2005; Wall & Plunkett, 2016). The amount of P and N fertiliser (kg) applied is one of the major nutrient sources that influences surface and subsurface nutrient losses in open drainage channels (Hart et al., 2004; Ibrahim et al., 2013; Richards et al., 2015; Watson & Foy, 2001). The rate of fertiliser application increases soluble reactive P (SRP) and TP concentrations in overland flow and drainage water (Watson et al., 2007). On these connecting fields, fertiliser application count is another source parameter that contributes nutrient loss to open drainage channels and may increase nutrient losses especially under wet soil conditions. The grazing status of a field connecting to open drainage channel specifies the risk of another major nutrient source that determines probability of livestock wastes (faeces and urine) being deposited near an open drainage channel (Bilotta et al., 2007; Gary et al., 1983; Hubbard et al., 2004) and damage to soils (that may be high nutrient rich) by trafficking and poaching to runoff into open drainage channels (Cassidy et al., 2017; Doody et al., 2014; Pietola et al., 2005). Its impact varies with grazing frequency (the number of times a grazing field is accessed by animals for grazing), with frequently grazed fields more susceptible to increase nutrient losses (Cassidy et al., 2017; Doody et al., 2014; Hubbard et al., 2004).

b. Mobilisation

Rainfall is the prime mobilising parameter that controls the transfer of nutrients within and around the open drainage channel (Pérez-Gutiérrez et al., 2020; Vadas et al., 2011; Yao et al., 2021).

c. Pathway

Farm roadways that are connected to open drainage channels under the nutrient transfer continuum acts as pathway by which runoff, carrying nutrients, is transferred into the open drain (Maher et al., 2023; Rice et al., 2022). Along the farm roadway network, nutrients may be contributed from the road surface (Davison et al., 2008; Edwards & Withers, 2008; Fenton et al., 2022). The farmyard is another pathway, which comprises hard standing areas that collect rainfall that becomes runoff to the adjacent open drainage channels (Edwards et al., 2008; Vero et al., 2020). The field surface influences runoff to connecting open drainage channels. Field surface is dependent of the soil drainage class (well, moderate, imperfect, and poorly-draining soils) and this dictates the runoff pathway between surface and subsurface pathways (Houlbrooke & Monaghan, 2009). There is high P loss risk from overland flow in poorly drained soils, moderate P loss risk from imperfectly drained soils, low P loss risk from both moderate and well-drained soils (Magette et al., 2007). The subsurface in-field drain pathway influences soil drainage capacity and subsequently the surface and subsurface pathways

(Houlbrooke & Monaghan, 2009). Subsurface in-field drains enhance infiltration and other processes in soils. Groundwater upwelling or seeping pathways introduces nitrate (NO_3-N) and P into open drainage channels, but depends on many factors such as landscape position and soil type. Groundwater composition may be high in nitrate concentrations, especially if the soil processes are modified by drainage (Edwards & Withers, 2008).

d. Receptor

The receptor is associated with the final direct impact on a watercourse (Wall et al., 2011). Watercourse in this regard is defined as any natural river, stream, or lake (but not a man-made drainage channel) (Department of Agriculture Food and the Marine, 2018) identifiable on an Ordnance Survey Ireland 6-inch map (www.osi.ie). In this study, all natural open drainage channels were assumed to have a final connection to a watercourse, with or without any proximity observed during the ground survey.

4.3.2 Scoring continuous and categorical parameters

The parameters were assigned individual risk scores that were scored arithmetically in a magnitude-impact matrix (Teunis & Schijven, 2019). For each open drainage channel, the risk score for every parameter was calculated by multiplying the score for magnitude (M) for contributing nutrients to an open drainage channel by the score for its relative impact (I) (Table 4.1) (after Shariff & Zaini, 2013).

Within the risk assessment, data for some parameters were measured quantitatively as *continuous data* (e.g. N fertiliser (kg) rate applied; Table 4.1), while others were assessed qualitatively as *categorical data* during field observation (e.g. connection to a farmyard; Table 4.1). As such, the M value for each parameter differed depending on the parameter type.

For continuous parameters, the M value was weighted between 0 and 3 using the formula:

$$(X_i - X_{min}) \times 3 \;/\; (X_{max} - X_{min}) \qquad \text{Eqn. 1}$$

where X_i is the on-farm observed data value; X_{min} and X_{max} are the minimum and maximum values observed across all farms.

For categorical parameters, the value was based on literature and/or expert judgement. Either "0" or "1" was scored as the "lowest" and "3" as the "highest" values (Table 4.1). For each open drainage channel, a total risk score was calculated by summing up all the risk scores for each continuous and/or categorical nutrient transfer continuum parameter for that open drainage channel. A total risk score represents the degree of risk (i.e. the scale of likelihood or propensity at which an open drainage channel contributes nutrients to a watercourse) associated with the blend of complex parameters (Table 4.1) for nutrient loss across all the open drainage channels on a given farm. Although the risk assessment takes into account the influence of the contributing area to an open drainage channel, the approach of weighting the contributions over the area rather than adding their impacts ensured an unbiased assessment where a larger area of fields surrounding the stretch of an open drainage channel could have led to high-risk. The risk assessment is simple to use, relying on easily accessible farm data, and can be used to assess the relative risk agricultural open drainage channels pose to water quality, without quantifying the nutrient loss.

4.3.3 Fieldwork to collect nutrient transfer continuum parameter data

Seven farms, dominated by heavy textured soils of a wide variety of bio-physical settings, were selected. These farms represented varying open drainage channel network density and connectivity risk compositions. During winter (November 2021 to March 2022), a field survey was conducted in which all open drainage channel networks were mapped as per Chapter 3 and Moloney et al. (2020). Open drainage channel network features such as connection to the farmyard, field slope, the proximity to water bodies, and connectivity pathways for nutrients into the open drainage channel network from in-field drains, farm roadways, groundwater springs, seepage and upwelling throughout the open drainage channel network, were noted on each farm. All the information characterising the open drainage channel network was recorded using an electronic device with ESRI ArcGIS Field Maps mobile software (version 21.4.0) (ESRI, 2024) during the field survey. This information was transferred to 'geographic information system' (GIS) mapping software, ArcMap GIS software (version 10.5). Data on other parameters for the nutrient transfer continuum elements was obtained from previous studies (Corbett et al., 2022a; Corbett et al., 2022b; Tuohy et al., 2021) and ongoing data collection by participating farmers and field agents. The data were downloaded and collated with data from the field survey, and the parameters in Table 4.1 were assigned an M score for every open drainage channel network across the farms.

In applying nutrient loss risk magnitude to areas that have never been calibrated, errors may prevail due to the unknowns in parameter settings and adjustments, and reliance on experts' opinions to set model parameters without calibration (Sharpley et al., 2017). However, the adoption of systems that are assessed and approved (as suggested by Bhandari et al. (2017) and Nelson et al. (2017)) enhanced the robust calibration of the parameters for the risk assessment.

4.3.4 Formation of Risk Classification System

Total risk score values for every open drainage channel for all seven farms were split into four categories of equal intervals to produce four potential risk classes (i.e. low risk, moderate risk, high-risk, and very high-risk). The range was determined by the possible highest and lowest total risk score that could occur as per the risk assessment scoring system developed. The risk classes were developed by:

$$(TRS_{high} - TRS_{low}) / 4 = I_e \qquad \text{Eqn. 2}$$

where TRS_{high} and TRS_{low} are the highest and lowest total risk score values recorded across the seven farms, and I_e is the interval between the four risk classes. These were colour-coded as green, yellow, orange, and red, respectively, on farm maps. Such maps provide information on the open drainage channels that are potential critical hotspots for nutrient losses on heavy-textured dairy farms. Risk classes in high and very high-risk ranges are identified as hotspots that may require mitigation measures.

4.3.5 Synoptic water sampling across dairy farms

Water quality parameters change over time, depending on the local climatic conditions and farming practices (Huebsch et al., 2013). At 105 sampling points throughout the drainage network across all farms, a total of 210 water samples (a pair of filtered and unfiltered at each sampling point) were collected during each season (sampling event) for 4 seasons (Spring (March) 2022 to Winter (January) 2023). The sampling was carried out across all 4 seasons to capture hydrological fluctuations and conditions, including surface and subsurface connectivity as per Chapter 3. As this study aimed to assess the risk of the open drainage channels, the water

N and P chemistry only validated the potential nutrient losses from the open drainage channel network surroundings and did not aim to elucidate the load or impact of this connection. Except for disconnected ditches (which were mostly dry), all man-made open drainage channels (farmyard connection, outlet, outflow, and secondary ditches; Moloney et al., 2020) and natural open drainage channels were sampled. At each water sample location, two 50 ml samples (filtered on-site using 0.45 µm filter paper and unfiltered) were collected for dissolved and total P analyses, respectively. All water samples were kept in an ice box during sampling and transportation, and then tested within one day of sample collection.

Filtered water samples were analysed for DRP and TDP using a Gallery discrete analyser (Gallery reference manual, 2016) and a Hach Ganimede P analyser, respectively. Total dissolved phosphorus was measured by acid persulphate oxidation, under high temperature and pressure. The unfiltered water samples were analysed for nitrite (NO_2-N), NH_4-N, total oxidised nitrogen (TON), and TRP using a Gallery analyser. Total phosphorus was analysed using the Ganimede P analyser. Phosphorus was measured colourimetrically by the ascorbic acid reduction method (Askew & Smith, 2005), where the 12-molybdophosphoric acid complex is formed by the reaction of orthophosphate ion with ammonium molybdate and antimony potassium tartrate (catalyst) and reduced ascorbic acid. All samples, reagent blanks, and check standards were analysed following the Standard Methods (APHA, 2005). All QC samples/check standards are made from certified stock standards from a different source than calibration standards. Quality control samples were analysed at the beginning and end of every batch, and every 10 samples within a batch, and if the QC fell outside limits, samples were repeated back to the last correct QC. Blanks were included in every batch and approximately 10 % of samples were repeated. Tolerances range up to a maximum of ±7.5 % of nominal value. All instruments used were calibrated in line with manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, PP was the difference between TP and TDP, and dissolved unreactive phosphorus (DUP) was the difference between TDP and DRP.

4.4 Results and Discussion

4.4.1 Open drainage channel characteristics

The total length and the number of open drainage channels in the farms are shown in Table 4.2. The length of an open drainage channel characterised the field area of contribution influencing the connectivity and potential risk of nutrient loss to an open drainage channel. Chapter 3 reported that multiple connectivity pathways may exist on a single open drainage channel. Although the relationship between the presence of connectivity pathways in open drainage channels and the length of the open drainage channels was not assessed in that study, longer open drainage channel lengths may have high connectivity, resulting in a potentially higher risk of nutrient loss. However, other parameters such as soil chemistry (Daly et al., 2017; Ezzati et al., 2020), slope, design (Hodaj et al., 2017), and vegetation (Soana et al., 2017) may also influence nutrient loss.

Table 4.2 The characteristics (length (m) and number) of open drainage channels per farm.

Farm #	Number of open drainage channels *per* farm	Length of all open drainage channels *per* farm (m) Average length	Total length	Natural open drainage channel average length	Man-made open drainage channel average length
1	25	291.50	7290	n/a	203
2	9	271.38	3799	382	188
3	40	509.23	25971	1898	170
4	16	397.44	14308	716	142
5	19	372.71	14163	1030	197
6	49	134.95	10526	322	122
7	13	204.27	4494	860	139

4.4.2 Risk classification system

Table 4.3 presents the risk classification system ranges based on the minimum and maximum possible total risk score from the risk assessment scoring system. These risk classification ranges were the basis on which risk class output maps for open drainage channel networks on each farm were developed (Figure 4.1).

Table 4.3 Risk classification system (risk class and score ranges) for risk assessment model for open drainage channels on heavy textured dairy farms.

Risk class	Risk score classification ranges	
Low	14.0	60.7
Moderate	60.8	107.5
High	107.6	154.3
Very high	154.4	201.0

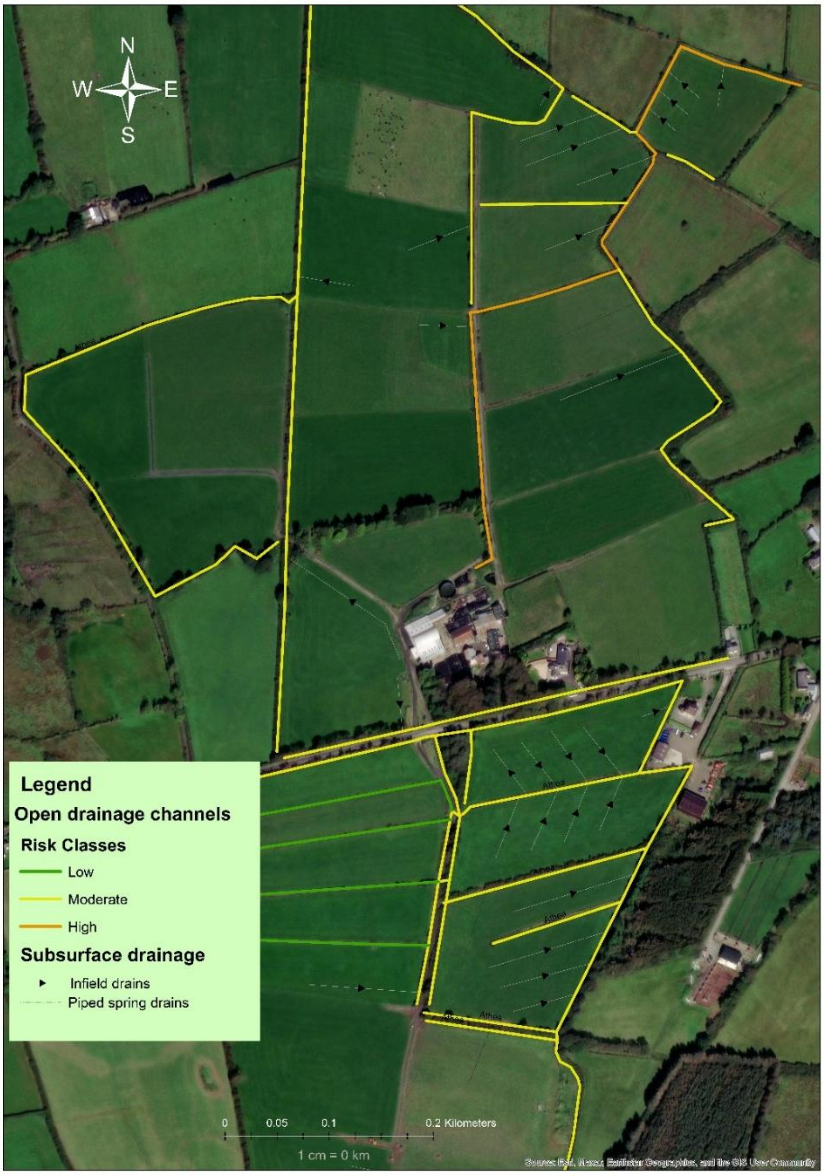


Figure 4.1 A map of a heavy textured grassland dairy farm (Farm #1 from Table 4.2) showing the risk classes of the open drainage channel network.

Although the possible lowest and highest total risk score are 14.0 and 201.0 according to the risk assessment scoring system (Table 4.3), the actual lowest and highest total risk scores recorded for the open drainage channels for the farms studied were 35.9 (Farm 4) and 144.4 (Farm 4), respectively. This indicates the highest total risk score across the farms reached only about 72 % of the potential maximum total risk score. Of the 171 open drainage channels on all seven farms, 23 %, 68 %, 9 %, and 0 % were ranked as low, moderate, high, and very high-risk classes, respectively (Figure 4.2). Data from individual farms were similar to the overall trend (Figure 4.2), except for Farm 6, where the majority (57 %) of the open drainage channels ranked as low-risk.

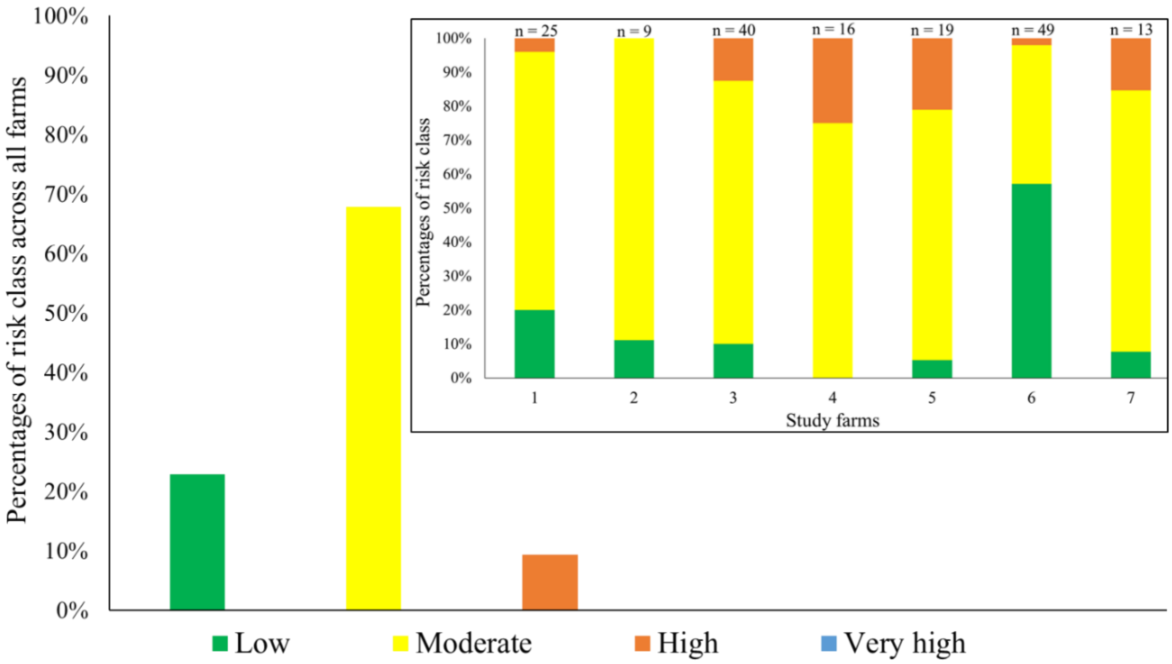


Figure 4.2 Percentages of risk classes for open drainage channels across all farms and within farms (inset).

Across the high-risk open drainage channels, the total risk score varied, with 144.4 being the highest recorded (a farmyard connection ditch) on Farm 4 and 109.9 being the lowest (a farmyard connection ditch) on Farm 7. The 9 % high-risk open drainage channels across the study farms were mostly on farmyard connection and outlet ditches (Table 4.4). This result is similar to Chapter 3 and Moloney et al. (2020), who found that farmyard connection ditches were potentially the riskiest.

Table 4.4 Number of high-risk channels (indicated by a 'X') by open drainage channel category.

Farm #	Natural open drainage channel	Farmyard connection ditch	Outlet ditch	Outflow ditch	Secondary ditch	Disconnected ditch
1		X				
2						
3		X	XXX		X	
4	X	XX	X			
5		XXX				
6		X				
7		XX				

Agricultural pressures on waterbodies in Ireland are associated with excess nutrients, mainly present as NO_3-N or DRP (EPA, 2023a). Phosphorus dominates in poorly drained soils, such as those included in this study, while N loss is more likely to vary depending on other specific site conditions (EPA, 2023a). In Ireland, the EPA considers good water in rivers to have NO_3-N concentrations of less than 1.8 mg L^{-1} and DRP concentrations of less than 0.035 mg P L^{-1} (EPA, 2023b). While open drainage channels assessed in these study farms are different water bodies from rivers as defined on national ordnance survey maps (6-inch maps) (www.osi.ie), comparisons of NO_3-N and DRP concentrations on the open drainage channels with the water quality standards for rivers act as a guide to show if a water sample is high or low.

The annual mean DRP concentrations in the open drainage channels, which ranged from 0.09 mg L^{-1} in moderate-risk class to 0.40 mg L^{-1} in high-risk class (Figure 4.3), were higher than the surface water standard of 0.035 mg L^{-1}. The annual mean NO_3-N concentrations on the open drainage channels were lower across the risk classes, with ranges of 0.59 mg L^{-1} in low-risk class to 1.18 mg L^{-1} in moderate-risk class (Figure 4.3) relative to the standard of

1.8 mg NO_3-N L^{-1}. This is consistent with the poorly draining conceptual model of the EPA in Ireland, as P losses dominate nutrients relative to N losses. While this may be beyond the scope of the present study, 32 % of sampling locations had high NO_3-N concentrations, indicating the N connectivity pathways that may be introducing NO_3-N into these open drainage channels (Chapter 3). Average P and N concentrations per risk class increased as the risk of the open drainage channels increased, except for average P concentrations for the moderate-risk class (Figure 4.3). This could be due to the anthropogenic and natural characteristics that create hydrochemical variation in the farm landscapes that contribute nutrients to the open drainage channels. With this caveat, this showed that the water quality seasonal grab samples validated the total risk score.

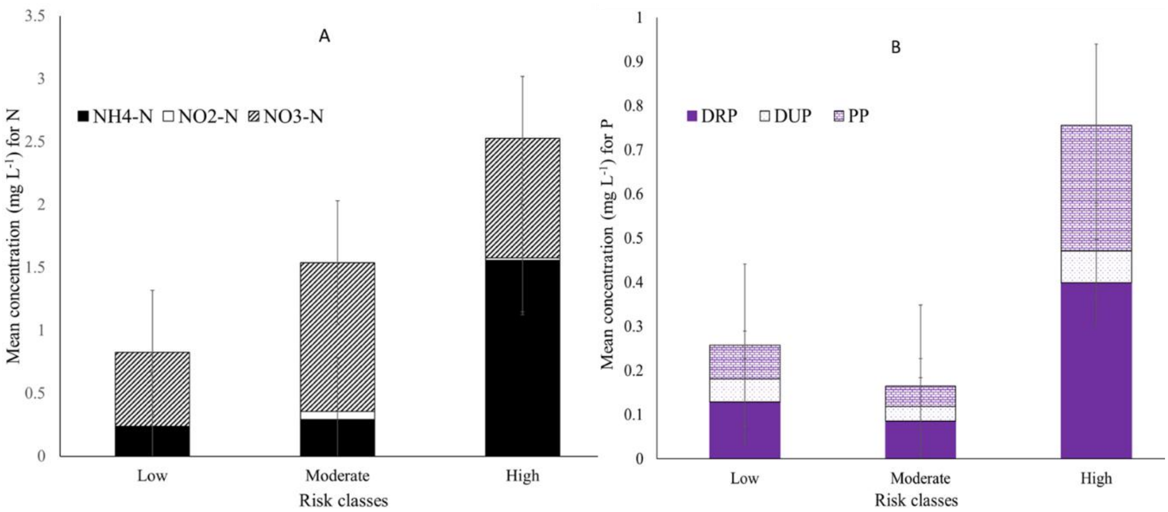


Figure 4.3 (A) Nitrogen and (B) phosphorus mean plus standard error concentrations from seasonal water sampling from within open drainage channels as per the risk classes across the case study farms.

4.4.3 Assessment of the nutrient transfer continuum elements on the open drainage channels

The contribution of the source to the average total risk score of open drainage channels per farm ranged from 44.2 % (Farm 2) to 63.5 % (Farm 5) (Figure 4.4). Similarly, the contribution of the source to the total risk score of each of the high-risk open drainage channels ranged from 40.3 – 70.2 % (Figure 4.5).

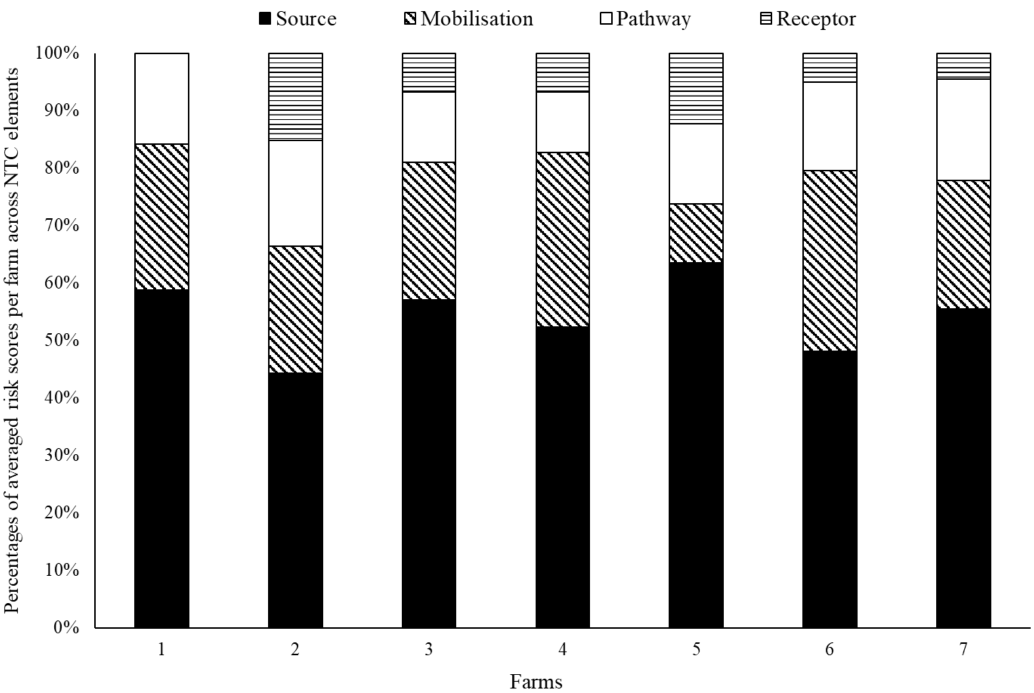


Figure 4.4 Percentages of averaged risk scores per farm across nutrient transfer continuum elements.

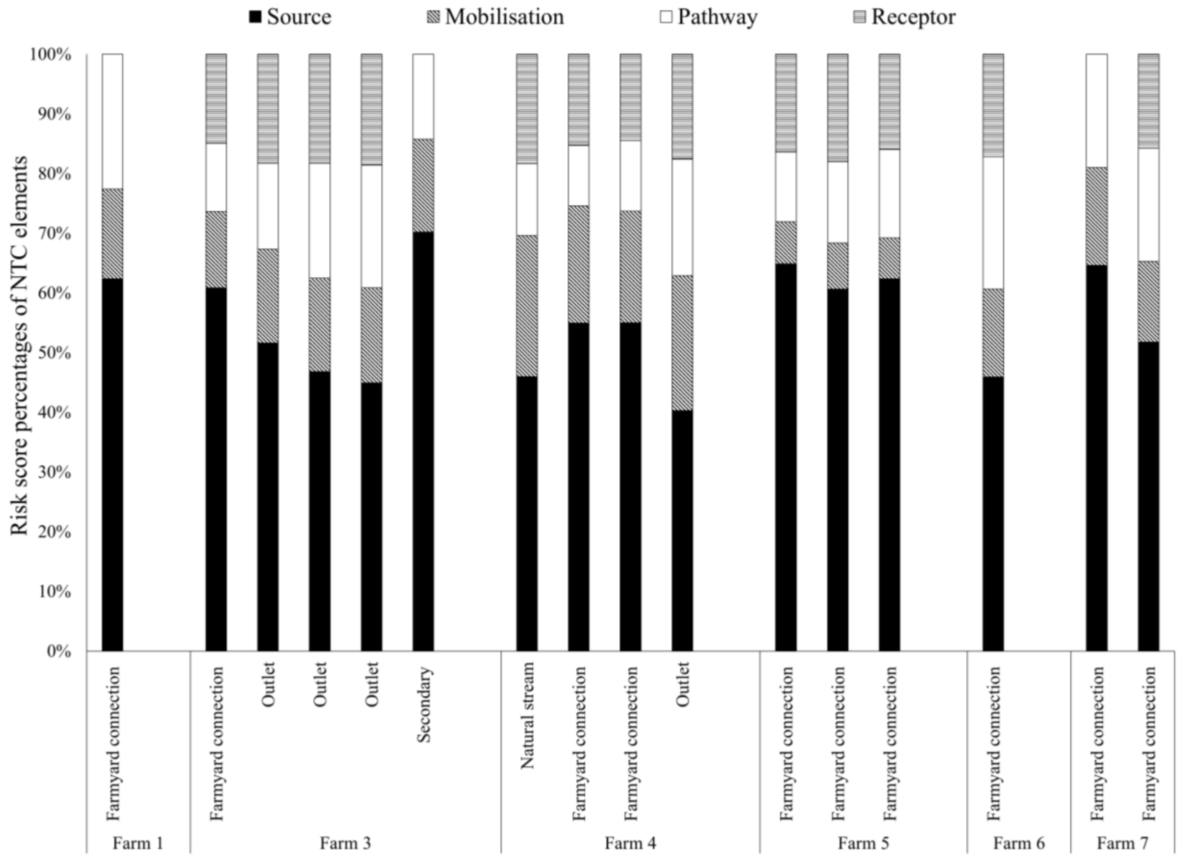


Figure 4.5 Risk score percentages of nutrient transfer continuum elements for farms with high-risk open drainage channels, excluding Farm 2 which had no high-risk open drainage channels.

The high proportion for source total risk score indicates that the multiple sources of nutrients, either from connection to farmyard, legacy soil P, fertiliser application, and grazing input parameters, primarily influenced the risk of nutrient losses (Cassidy et al., 2017; Moloney et al., 2020) to these open drainage channels. Under source contribution, the majority of the high-risk open drainage channels were connected to farmyards (point sources), accounting for 62.5 %, implying the remaining 37.5 % were connected to diffuse sources (Table 4.4). The highest source contribution to a total risk score recorded on high-risk open drainage channel was 70.2 %, and this occurred on secondary ditch with no farmyard connection (Figure 4.5). This could be attributed to the open drainage channel's connectivity with high soil P-status fields, which received high fertiliser application for the duration of this study. This, together with surface and subsurface sources, may have led to the high total risk score on the other 37.5 % (Table 4.4) of the whole high-risk open drainage channels with no connection to farmyards.

Along a connected pathway to the open drainage channel, the mobilisation of nutrients from the source was integral in most of the open drainage channels. The percentage of mobilisation contribution to the average total risk score of the open drainage channels per farm ranged from 10.2 % to 31.5 % (Figure 4.4). Rainfall is the primary factor by which mobilisation occurs for nutrient losses (Wang et al., 2020). Rainfall characteristics, including the intensity, duration and frequency, may influence the hydrological conditions that are critical to the surface and subsurface nutrient movement (Pérez-Gutiérrez et al. 2020). This necessitates the need to break the pathway to prevent the mobilised nutrient from the source to the receptor.

Nutrients enter the open drainage channels through multiple (surface, shallow subsurface and groundwater) pathways. The pathway contribution to the average total risk score per farm ranged from 10.5 % to 18.4 % (Figure 4.4). Heavy textured farms have multiple subsurface and surface connectivity pathways through which nutrients are lost (Clagnan et al., 2019; Granger et al., 2010), and these may have contributed to the high-risk open drainage channels. Eighteen-point-six percent and 18.6 % of the high-risk open drainage channels received risk scores from roadway and farmyard runoff surface connectivity pathways to the total risk score, respectively, while 87.5 % and 31.3 % of the high-risk open drainage channels received risk scores from in-field drains and groundwater subsurface connectivity pathways, respectively. Although the pathway percentage contribution to the total risk score of the high-risk open drainage channels ranged from 10.1 – 22.6 %, the highest pathway contribution to total risk score for an open drainage channel was 44.9 % which was a moderate-risk open drainage channel on Farm 6.

The connection to the receptor was not present on all high-risk open drainage channels. However, contributions from 14.5 to 18.6 % of the total risk score of high-risk open drainage channels with connection to receptor for the study farms (Figure 4.5). This informs the importance of considering the delivery of the final nutrient loss through the open drainage channels and may inform the mitigation type.

4.5 Mitigation of the high-risk open drainage channels

In Ireland, the EU Nitrates Directive is implemented through the NAP, which applies to all farms in the country. This programme of measures outlines best farming practices to achieve good water quality outcomes for different farm enterprises. The EPA in Ireland identifies “breaking the pathway” on poorly draining soils, such as those in the present study, as an effective way to break the connectivity of surface or near-surface runoff between sources and waters. Chapter 3 classified the open drainage channel network into different ditch categories. Building on this work, the present study identifies open drainage channel sections within these large networks to be of higher risk and which may need mitigation. A combination of targeted measures is therefore necessary to improve water quality. This may include (1) source management (2) breaking the pathway (stopping runoff or near-runoff being delivered to waters), and (3) installation of in-channel filters (to slow the flow and attenuate a proportion of nutrients in dissolved and particulate forms from discharging through that open drainage channel section). On poorly draining soils this combined treatment train (Bourke et al., 2022) may prevent high nutrient-content water discharging from high-risk open drainage channel sections to the broader aquatic environment. Scrutiny of individual high-risk total risk score for different open drainage channel sections enables an advisor and farmer to identify specific sources, pathways, and in-channel actions as required. These may differ due to site-specific factors and cannot therefore be generic. Farmers are more inclined to accept less costly measures (van den Berg et al., 2023), and therefore these should be considered during the selection of mitigation measures (McDowell & Nash, 2012; King et al., 2015).

Chapter 3 and Fenton et al. (2021) detailed potential mitigation measures and costs available in terms of "break the pathway" mitigation options and costs. A few examples include: re-directing runoff away from internal roadways and the farmyard to collection or buffer areas with low-cost diversion bars or water bars (Fenton et al., 2021); installation of riparian (spatially targeted and linear) buffers along natural streams (Stutter et al., 2021) to control nutrient losses from the upslope field and connected internal farm roadways (Palmer, 2012; Yuan et al., 2009); targeted engineered mitigation measures including low-grade weirs (Faust et al., 2018), bunded drains, filter cells (Teagasc, 2022); and management of in-channel sediments through maintenance or characterisation of soil/sub-soil layer chemistry (Shore et al., 2015), which is both a sink and source of nutrients (Daly et al. 2017).

4.6 Conclusion

Assessments of nutrient loss from open drainage channels on poorly draining (heavy textured) soils are largely associated with predictions of surface runoff from critical hotspots. The risk assessment developed in this study combines potential water quality impacts from surface, subsurface, and groundwater characteristics of connecting fields to produce a colour-coded model of different potential water quality risk levels by which open drainage channels can be risk assessed. This risk assessment enables the production of risk maps that identify potential high- or very-high risk open drainage channels on dairy farms with heavy textured soils and assesses the nutrient transfer continuum elements to inform mitigation. Unlike previous open drainage channel risk assessment studies of Moloney et al. (2020) and Chapter 3, this study critically assesses all the source-mobilisation-pathway-receptor multi-parameters of the open drainage channel nutrient transfer continuum framework, provides in-depth information regarding high-risk open drainage channels to elucidate which parameters require attention during mitigation. The findings of this study apply to dairy farms on heavy textured soils in high rainfall areas, and may (or may not) differ in other geographic areas with different soils, climates and agricultural practices. However, it should be noted that the same methodology can be applied anywhere to develop a semi-quantitative risk assessment that will inform mitigation management. Future work incorporating varying risks encountered over time across wider farm characteristics will improve the risk scoring system to produce a more robust model that can be applied more generally on farms.

4.7 Summary

This chapter identified high-risk surface drains on grasslands and their contributing factors. Farm roadway surface runoff connectivity to surface drains is among the key contributing factors to nutrient losses from diffuse sources. Numerous measures have been proposed to mitigate farm roadway runoff, but to date, uptake by farmers has been limited. Chapter 5 examines the efficacy of such systems on a farm in Ireland.

5. Examination of nutrient and sediment loss mitigation for farm roadway runoff on an Irish dairy farm.

5.1 Overview

The aim of this chapter was to develop mitigation strategy to identify nutrient loss pathway on the surface drains, co-implement with farmers and assess the efficiency to enable mitigation uptake on farms.

Parts of this chapter have been published in Journal of Agricultural Water Management (Opoku, D. G., Healy, M. G., Fenton, O. & Tuohy, P. (2025). Examination of nutrient and sediment loss mitigation for farm roadway runoff on an Irish dairy farm. *Agricultural Water Management, 322*, 110007. https://doi.org/10.1016/j.agwat.2025.110007)

To avoid repetition, acronyms that have already been defined in preceding chapters are not defined in this chapter. Citations to papers published by the author as part of this thesis are referred to by Chapter number.

5.2 Introduction

Agricultural pollution from nutrient and sediment losses remains a concern for water quality degradation globally (McDowell et al., 2020; Shortle & Horan, 2017). In the European Union (EU), pollution from agriculture contributes to 22 % of surface water and 28 % of groundwater pollution (EEA, 2021). To alleviate this environmental concern, multiple international, regional and local policies and regulations for managing agricultural pollution have been developed and continue to be adapted for practical implementation. In 2000, the EU developed the Water Framework Directive (WFD) (2000/60/EC; OJEC, 2000) for member states to adopt an integrated approach for managing waterbodies to reduce pollution and improve water quality to a "good status" by set deadlines. As part of the WFD integrated approach on water quality management, the Nitrates Directive (91/676/EEC) targets reducing agricultural pollution to waterbodies through good agricultural practices (OJEC, 1991) and requires EU member states to develop a NAP in reaching this goal.

In Ireland, programmes of measures to fulfil the goals of the WFD are set out and revised within the NAP (DHLGH, 2021a) to minimise both diffuse and point agricultural pollution potential. The NAP measures include, but are not limited to, limits on farm stocking rates and nutrient application rates, prohibitions on organic and chemical fertiliser application at environmentally-sensitive periods, minimum storage capacity for livestock manures and minimum set-back distances from waters (DHLGH, 2021b).

Recent iterations of the Nitrates Directive (S.I. No. 605 of 2017) acknowledge the risk of pollution from farm roadway runoff into connected open drainage channels and stipulate that "there shall be no direct runoff of soiled water from farm roadways to waters from 1 January 2021", alongside mitigation guidance options under the NAP to manage farm-scale agricultural pollution. Recent research on roadway runoff shows nutrient losses occurring both on open and closed periods on grassland farms (Fenton et al., 2024b; Sifundza et al., 2024). It has been found that 8.4 % (Rice et al., 2022) to 11.6 % (Maher et al., 2023) of roadway sections are connected to open drainage channels, while farm roadway and open drainage channel densities are highest on heavy textured soils. During rainfall events, nutrients within and on farm roadway sections connected to open drainage channels form CSAs (Chapters 3 and 4) and are a sub-component of the nutrient transfer continuum (Fenton et al., 2022). After identification of CSAs, breaking the pathway before delivery of nutrient-rich roadway runoff to open channels is advised on farms (Fenton et al., 2021; Lucci et al., 2010).

Mitigation measures exist in broad terms, but bespoke solutions are needed for specific runoff problems. Primarily, approaches for preventing roadway runoff connectivity focus on breaking the pathway with on-roadway flow diversion structures and retention mitigation systems to reduce the transfer of nutrient and sediment losses to open drainage channels (Fenton et al., 2021; Tanner et al., 2023). The NAP recommends multiple mitigation measures and highlights a “right measure, right place” approach in their use to address diffuse pollutant sources, including farm roadways (DHLGH, 2024). However, the implementation of these recommended mitigation measures has generally only occurred on EIP participant farms with no efficiency testing to guide future iterations and improvements of the mitigation measures. This limits knowledge of the efficiency of these mitigation measures, especially as they have tailored designs.

The efficiency of mitigation measures likely varies depending on the geo-positioning and design of the measure (Tanner et al., 2020; Thomas et al., 2016) and on the CSA characteristics (Tanner et al., 2020) such as farm management (e.g. grazing, stocking rate), rainfall, landscape characteristics (e.g. slope, soil) and contributing roadway area (e.g. size, composition, length and slope). These factors influence the impact on the hydrological and biogeochemical processes that determine the efficiency of mitigation measures (Persson & Wittgren, 2003). Furthermore, their efficiency may be influenced by available farmland sizes, which is often a constraint due to farmers’ inability to release farm areas (Lastra-Bravo et al., 2015; Wilcock et al., 2012) for environmental measures. Ryan et al. (2025) observed that farmers are inclined to undertake evidence-based measures and those that require high-level knowledge or understanding for effective implementation. Assessing the efficiency of NAP-recommended mitigation measures in breaking the pathway and slowing farm roadway runoff to reduce agricultural nutrient and sediment transfer to connecting open drainage channels will provide a thorough understanding of the context under which these mitigation measures may be effective work. Such an understanding of their efficiency will improve knowledge of the mitigation measures in managing farm roadway runoff on Irish farms.

This study selects an Irish dairy farm with a high density of farm roadways and open drainage channels. The study aims to (1) use existing tools to examine and identify farm roadway CSAs where connectivity runoff enters open drainage channels (2) co-develop and implement bespoke mitigation measures for these identified locations with the farmer, and (3) monitor the efficiency of the implemented mitigation measures at these locations under practical conditions.

5.3 Materials and Methods

5.3.1 Study site

A dairy farm (45 ha) situated in the south-west of Ireland was selected (Figure 5.1) following a previous semi-quantitative risk assessment on open drainage channels where farm roadway runoff connectivity was a prevalent issue and locations with a high risk of roadway runoff were identified (Chapter 4). The location has a 10-year average annual rainfall of 1541 mm. The annual agronomic soil testing for phosphorus (P) using Morgan's reagent (Peech & English, 1944) carried out on the fields showed that 10.3 % had high soil P index 4 (> 8.0 mg L^{-1} P).

The site has undulating topography with steep slopes (4 – 8 °) and soils classified as "heavy-textured". The soils vary from mineral to humic (Fealy et al., 2009), and are mainly moderately drained (55 %) or poorly drained (45 %), with 13.6 % of the fields having in-field drains installed. The nature of the soils and the topography enable overland flow and potential runoff from CSAs of sediment, N and P into open drainage channels. The fields in the central parts of the farm have mostly moderately draining soils and therefore have a potential for infiltration (leaching) of nutrients, which complicates the task of isolating pollutant loss pathways on the farm.

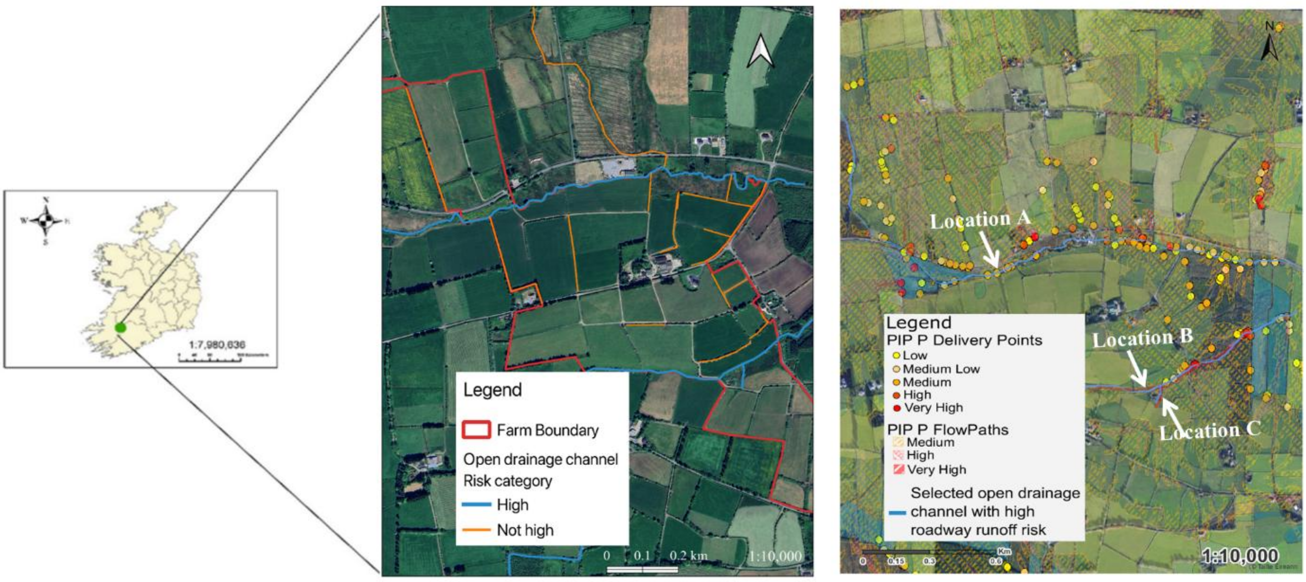


Figure 5.1 Map of Ireland indicating farm location, "high-risk" open drainage channels with roadway runoff connectivity (Chapter 4), and the farm roadway CSA locations on the "high-risk" open drainage channels (https://gis.epa.ie/EPAMaps/Water).

5.3.2 Identifying runoff connectivity and critical source areas

Using the semi-quantitative risk assessment of Chapter 4, three locations on the farm (A, B and C) with "high-risk" open drainage channels with roadway runoff connectivity as a major contributor were identified (Figure 5.1). These locations were cross-checked with the national EPA nutrient loss pathway map (https://gis.epa.ie/EPAMaps/Water) for risky pathways and delivery points to identify roadway runoff CSAs with a high likelihood of nutrient and sediment loss. The identified points of surface runoff delivery to open drainage channels from the national EPA nutrient loss pathway map were noted for further assessment. Following this, a ground survey assessment and visual assessment (Fenton et al., 2021) was conducted during hydrologically-active periods to fine-tune these farm roadway CSAs and to identify the optimum locations for mitigation measures.

5.3.3 Co-developing and co-implementation of mitigation measures

Several farm visits were undertaken to determine possible mitigation solutions for the three identified farm roadway CSAs in consultation with the farmer. For all three locations, a treatment train mitigation measure of diversion-sediment pond-vegetated riparian buffer was proposed because it combines multiple measures with diverse functions to complement one another's limitations (Nicholson et al., 2012; Quinn et al., 2007). The diversion bar/cambered roadway diverts runoff to the sediment pond for primary treatment (sedimentation) and subsequently to the riparian buffer for secondary treatment (removal of dissolved pollutants). At Locations A and B, on-roadway concrete-based diversion bars extending 0.3 m beyond the edge of the roadway were installed to direct runoff to the sediment ponds. For Location C, the farm roadway was resurfaced using gravel and cambered to divert roadway runoff towards the sediment ponds. A constant groundwater spring flow from an adjacent field through the cambered section into the sediment pond was observed at this location. The process of co-designing mitigation measures with landowners or advisors typically involves compromises associated with many factors that affect the final mitigation designs. These may for example, limit the size of the installed sediment pond from optimal e.g., this decision could be based on land availability at the delivery point or an unwillingness of the landowner to use that land.

The optimal sediment pond volume was calculated based on the hydraulic loading rate of the site to ensure optimal pollutant removal through sediment retention (Smith & Muirhead, 2023; Robotham et al., 2021). For each location, the sediment pond volume, V (m^3), was calculated using:

$$V = R \times T \qquad \text{Eq. 1}$$

where R is the peak discharge rate (m^3s^{-1}), and T is the residence time (s). The peak discharge rate, R, in Eq. 1 was calculated using (Barber, 2013):

$$R = C \times A \times I \qquad \text{Eq. 2}$$

where C is a dimensionless runoff coefficient dependent on hydrological factors (the soil type, land use, degree of imperviousness, slope, surface roughness, antecedent moisture condition, duration and intensity of rainfall, recurrence interval of rainfall, interception and surface storage variables); A is the contributing farm roadway area (m^2), and I is the average intensity of rainfall ($m\ s^{-1}$). A value of 0.5 was assigned to C, which was estimated for forest roadways (Jordán & Martínez-Zavala, 2008) of similar gravel and unpaved characteristics. Using local meteorological records, rainfall intensity, I, for a 6-hour duration, 1-in-5-year return period, storm event was used – 5.67 mm hr^{-1} (1.57×10^{-6} m s^{-1}). Contributing farm roadway areas of 429.3 m^2 over an 8.4 ° slope (Location A), 106.8 m^2 over a 6.7 ° slope (Location B) and 249.5 m^2 over a 7.3 ° slope (Location C) were used.

The residence time, T, in Eq. 1 was calculated using:

$$T = s/Vs \qquad \text{Eq. 3}$$

where s (m) is the travel distance set for sediments to fully settle in the sediment pond (using s at 1 m) and Vs (ms^{-1}) is the velocity of sediment settling for clay sediment, calculated using Stokes' Law:

$$Vs = \frac{d^2 g\,(Dp - Df)}{18\mu} \qquad \text{Eq. 4}$$

where, d is the diameter of the particle (3.9×10^{-6} m for clay; Barber, 2013), g is gravity (9.8 m s^{-2}), Dp is the density of clay particles (2860 kg m^{-3}; Schjønning et al., 2017), Df is the density of the fluid (1000 kg m^{-3}), and μ is the dynamic viscosity of the fluid (0.001 kg $(m\ s)^{-1}$).

Based on these hydrological flow estimations, the volumes (V) at depth (s) = 1 m required for the sediment ponds were calculated as 28.6 m^3, 7.1 m^3 and 16.6 m^3 for Locations A, B and C, respectively. While these estimated sediment pond sizes may allow optimum effectiveness, site constraints including high water table of the adjacent open drainage channels and limited land area, especially at Location A, necessitated resizing of the sediment pond sizes to ~ 4 m^3, ~ 7 m^3 and ~17 m^3 at sediment settling depths (s) of 0.5 m, 0.5 m and 1 m for Locations A, B and C respectively (Figure 2). These constraints led to an undersized sediment pond volume at Location A, while Locations B and C remained with their optimum sediment pond volumes. Based on these new volumes adjusted to suit the available land conditions, residence time (T), using T = V/R, was estimated as 3 hours, 23 hours and 24 hours for locations A, B and C, respectively. Following Barber (2013), the sediment ponds were configured into pond cells to enhance removal efficiency while adapting to the site conditions. For the sediment pond configurations, two sediment pond cells, each measuring 2.5 × 1.5 × 0.5 m (L x W x D) at

Location A, one sediment pond cell measuring 4.0 × 3.5 × 0.5 m at Location B and two sediment pond cells, each measuring 4.25 × 2.0 × 1.0 m at Location C, were excavated (Figure 5.2). The sediment ponds were manually levelled after digger excavation and crosschecked with a spirit level. This allowed accurate measurement of the accumulated sediment volume.

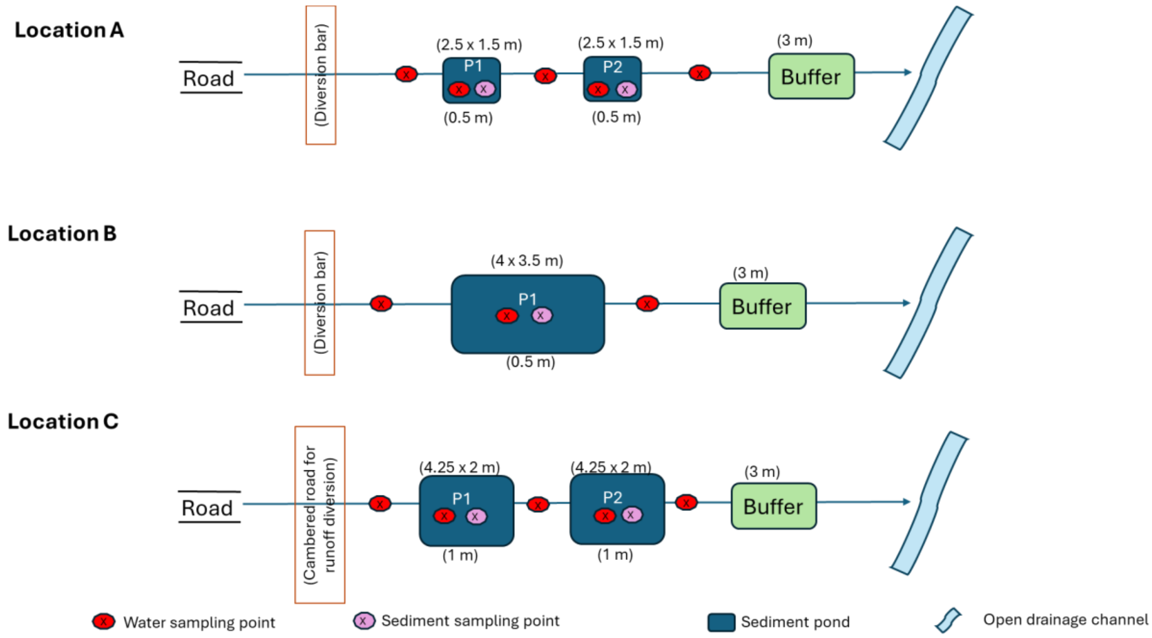


Figure 5.2 Schematic representation of mitigation measures and dimensions.

To prevent pond bank erosion, sediment ponds were excavated to create banked sides for stability (Barber, 2013) and lined with weed mats which enabled estimation of accumulated sediment volume. Edges (excluding exit and entry) along the sediment pond cell(s) were bunded and grassed to prevent overland flow from adjacent areas during rainfall events. At the exit of every sediment pond, a 1 m-long, 0.10 m-diameter plastic pipe was connected to the next sediment pond or discharged into the 3 m-wide riparian buffer.

The riparian buffer was installed at the end of sediment pond configuration at each location before the adjacent open drainage channel to meet the current recommendation of at least a 3 m-wide vegetated riparian buffer to prevent direct soiled runoff into waterbodies under the 2022 NAP 5 in the EPA Research Report No. 485 (Ó'Huallacháin et al., 2023). While such an additional measure is expected to further reduce the sediment and pollutant concentration in the runoff from the sediment ponds, the nature of vegetated riparian buffers does not allow direct measurement of downstream water quality at all locations, and therefore measurement at these locations was not undertaken. This study, therefore, only evaluates the efficiency of the sediment ponds on the farm.

5.3.4 Water and sediment sampling for testing mitigation measures

5.3.4.1 Water and sediment sampling

Water and sediment samples were taken during the hydrologically-active periods between the week of 22nd October 2024 to 19th March 2025, except for 3 weeks from late December 2024 to early January 2025 when the site was not accessible due to heavy snowfall. Using sampling points in Figure 5.2, two 50 ml paired (filtered and unfiltered) water samples were taken weekly from all water sampling points in all locations for N and P fractions analysis. In addition, 500 ml water samples were taken weekly at these water sampling points in all locations for total suspended solids (TSS) concentration measurement. Inlet water samples were collected from diverted roadway runoff flows at the entry points for each location. All the 50 ml (filtered and unfiltered) and the 500 ml water samples were stored and transported in cool boxes to the laboratory for water analysis and TSS within 24 h of sample collection.

To measure accumulated sediment volume in each pond, two 1 m graduated staffs were placed in each pond to measure the depth of accumulated sediment over the study duration (Cooper et al., 2019). The average depth readings of accumulated sediment from both graduated staffs within each pond were calculated every 4 weeks. For a particular pond cell, the calculated average depth and pond area were multiplied to estimate the accumulated volume for that 4-week period. After each 4-week measurement of accumulated sediment, ~0.5 kg of fresh (wet) sediment samples were collected from the base of each pond cell. The sediment samples were transported in cool ice boxes to the laboratory and then analysed for water-soluble P (WSP) to ascertain the sediment P composition.

5.3.5 Laboratory analysis

The unfiltered 50 ml grab water samples were analysed calorimetrically for NO_2-N, NH_4-N, TON, and TRP using a Thermo Fisher Scientific Gallery ™ Discrete Analyzer. The unfiltered samples were analysed for TP and TN was analysed using the Hach Ganimede P analyser and the Hach Ganimede N analyser, respectively. The filtered 50 ml grab water samples were analysed for DRP and TDP using a Thermo Fisher Scientific Gallery ™ Discrete Analyzer and a Hach Ganimede P analyser, respectively. All water samples, reagent blanks and check standards were analysed following the Standard Methods (APHA, 2005). All QC samples/check standards were prepared from certified stock standards from a different source than calibration standards. Quality control samples were analysed at the beginning and end of every sample batch, for every 10 samples within a batch, and if the QC fell outside limits, samples were repeated to the last correct QC. Blanks were included in every sample batch for analysis, and approximately 10 % of samples were repeated. Tolerances ranged up to a maximum of ±7.5 % of the nominal value. All instruments used were calibrated in line with the manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, PP was calculated by the difference between TP and TDP, and DUP was calculated by the difference between TDP and DRP. Total suspended sediment concentrations were measured using the standard gravimetric method (APHA, 2005).

For WSP analysis, portions of the sediment samples for each pond cell were prepared by air-drying and sieving through 2 mm, and 1 g of the prepared sediments were moistened with 2 ml of deionised water and allowed to stand for 22 hours. These were further moistened with 70 ml of deionised water, equilibrated for 1 h on a reciprocating shaker (van der Paauw, 1971) and filtered using Whatman No. 4 filter paper before the filtrate was quantified calorimetrically for P. Using the sediment mass (g) and total volume of deionised water (ml; converted to L) used for moistening the sediment, P concentration (mg L^{-1}) in the filtrate was converted to mg/g. The water soluble P values provide information on the concentration of the readily available P within the sediment and indicate how easily this readily available P can be released into the pond water.

5.3.6 Data analysis

Microsoft Excel software version 16.0 (2016) was used for data computing and preparation prior to statistical analysis, and R Studio version 4.3.2 (2023) was used for statistical procedures. To assess the efficiency of the sediment ponds deployed at the various locations, the water sampling results for the N and P fractions, TSS, and physical and chemical sediment characteristics were compared. The removal efficiency was defined as the percentage removal calculated as the difference in water quality parameter concentrations at the inlet sampling point of the sediment pond and the outlet sampling point of the sediment pond (Equation 5):

$$\text{Removal efficiency (\%)} = \frac{\text{Inlet water concentration} - \text{Outlet water concentration}}{\text{Inlet water concentration}} \text{ x } 100\ \% \qquad \text{Eq. 5}$$

All inlet and outlet water quality data were assessed for normality with the Shapiro-Wilk test and were not transformed. To test efficiency of sediment ponds statistically at individual locations, the inlet and outlet water quality data for each location were tested for statistically significant differences using the paired T-test for normally distributed water quality parameters (Barber, 2013; Robotham et al., 2021) and the Wilcoxon Signed-Rank (pairwise test) for non-normally distributed water quality parameters. All significant differences were observed at an alpha level of 0.05 (95 %) confidence level, and where alpha level was much lower, a 0.01 (99 %) confidence level was used. All water quality parameter values “<LOD” (below the Limit of Detection) or “not detectable” were treated as zero for analysis. Mean comparisons were undertaken for WSP, accumulated sediment volume and weather data (rainfall (precipitation) and temperature). Rainfall refers to the total precipitation, and as these heavy textured, poorly drained soils remained wet throughout the study period, precipitation/rainfall may be considered as very crucial for runoff. Concentrations of P and N fractions of the inlet and outlet of pond configuration systems for a location were examined as proportions of total P and N.

5.4 Results and Discussion

5.4.1 Sediment trapping in sediment pond configurations

During the study, accumulated sediment volumes (m^3) within pond cells increased by 0.169 m^3 in pond cell 1 and 0.128 m^3 in pond cell 2 at Location A, 0.088 m^3 in pond cell 1 at Location

B, and 0.077 m^3 in pond cell 1 and 0.038 m^3 in pond cell 2 at Location C. This indicates a 53 % to 567 % sediment accumulation (relative to the initial sampling volumes) across locations during the monitoring period (Table 5.1). These findings show that sediment accumulation in ponds is related to contributing area, with larger areas yielding more accumulated sediment in ponds.

Table 5.1 Accumulated sediment volumes (m^3) and percentage increase from start to the end of monitoring.

Location (contributing area (m^2))	Pond	Volume of sediment accumulated at initial measurement (m^3)	Volume of sediment accumulated at final measurement (m^3)	Mean volume of sediment accumulated at each measurement (m^3)	Total sediment volume increase relative to initial volume (%)
A (429.3 m^2)	Cell 1	0.038	0.206	0.13	450.0
	Cell 2	0.023	0.150	0.10	566.7
B (106.8 m^2)	Cell 1	0.049	0.137	0.09	178.6
C (249.5 m^2)	Cell 1	0.068	0.145	0.11	112.5
	Cell 2	0.072	0.111	0.09	52.9

The sedimentation process is influenced by factors including pond size and flow reduction capacity, runoff flow velocity, sediment size characteristics. Sediment size influences sedimentation, allowing coarse sediment to settle more quickly and fine particles to remain suspended until flow slows (Clarke, 2013; Levine, 2020; Ockenden et al., 2012). Higher mean sediment accumulation in the first pond cells at Locations A and B (Table 5.1) suggests coarse sediment trapping, which occupies more volume. Conversely, lower mean sediment accumulation in their respective second pond cells (Table 5.1) suggests fine sediment trapping which, due to their smaller size and lower weight, travel far and occupy less volume. Visual observations, especially during and immediately after rainfall, revealed cloudier water in second pond cells, suggesting resuspension of lightweight fine sediments. This aligns with findings of multi-pond studies, which also observed that first pond cells trapped heavier and less mobile sediments than subsequent cells (Barber, 2013; Robotham et al., 2021). Fine sediments are major P carriers that contribute to P losses (Ballantine et al., 2006; Shore et al., 2015). This may have contributed to the higher WSP concentrations in sediments of the second pond cells relative to the first pond cells at Locations A and C (Figure 5.3).

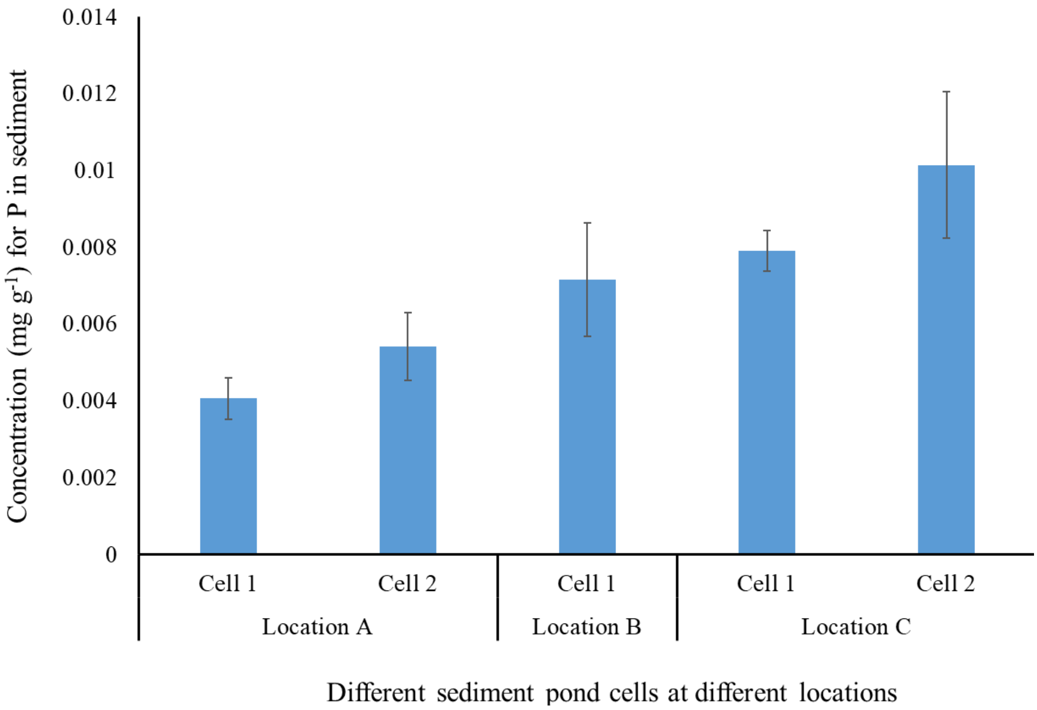


Figure 5.3 Water-soluble P (WSP) of sediment samples at different locations and pond cells.

Rainfall, the primary driver for sediment mobilisation from farm roadways (Fenton et al., 2021; Rice et al., 2022) had positive correlations with sediment accumulation ($R^2 = 0.65 – 1$) (Figure 5.4), suggesting that transport of accumulated sediments in ponds was dependent on rainfall. This correlation was even more pronounced in the first pond cells.

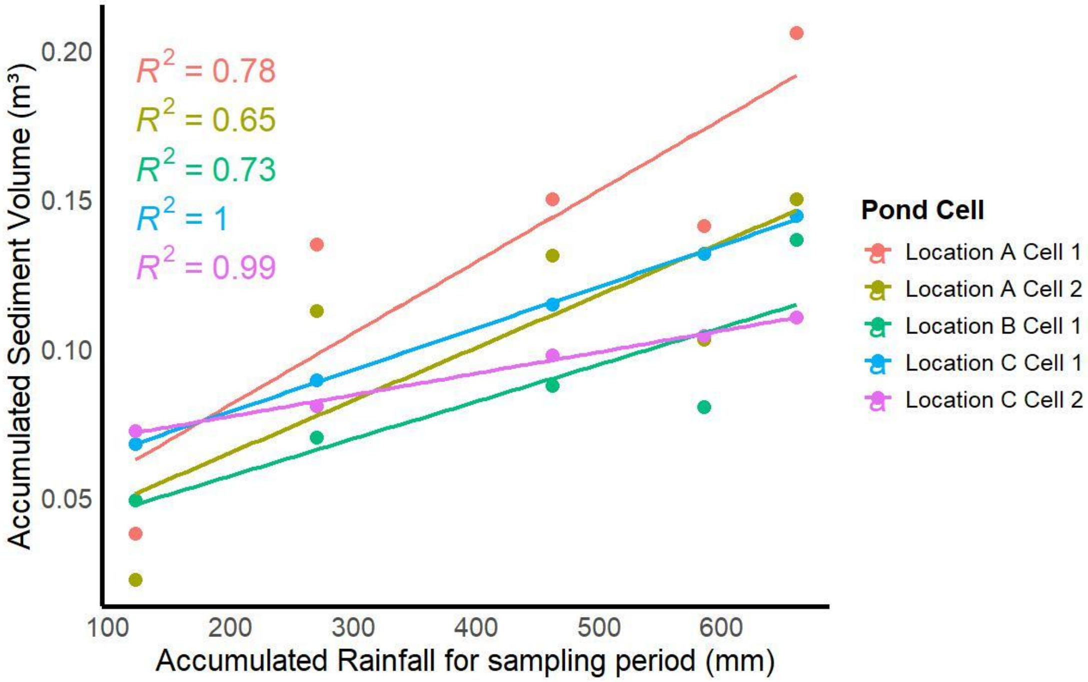


Figure 5.4 Correlation between accumulated sediment volume (m^3) and accumulated rainfall (mm) for every sampling period.

5.4.2 Nutrient and TSS removal in sediment ponds

5.4.2.1 Nitrogen removal efficiency in sediment ponds configurations

Over the sampling period, TN removal efficiencies in Location A and C (both two-cell configurations) were similar, at 30.9 ± 39.0 % and 27.4 ± 42.6 % removal respectively, while the one-cell pond configuration system at Location B recorded only 0.46 ± 13.8% removal (Figure 5.5). Nevertheless, the mean outlet TN concentrations for the sediment ponds at all locations (Appendix C, Table C1) were lower than the current N discharge limit of 10 mg L^{-1} under EU Urban Waste Water Treatment Directive (UWWTD 91/271) (European Commission, 2024). The TN removal efficiencies in Locations A and C are consistent with the average TN removal efficiency of 31 % for wet ponds in Koch et al. (2014), but relatively lower than Mallin et al. (2012) reported results of 66 – 96 % TN reduction in a 4.7 ha multi-segmented constructed wetland designed for a 24-hour duration, 1-in-100-year return period storm event. Within runoff treatment systems, sedimentation and microbial transformations (mineralisation, nitrification and denitrification) and plant uptake are the primary N removal mechanisms (Kill et al., 2018; Vymazal et al., 1998), and these factors considerably influence variation in N removal efficiencies. The relatively low TN removal in this study could be due to low temperatures measured during the study period (6.73 ± 0.26 ºC) which reduce the microbial transformations (Kill et al., 2018; Robotham et al., 2021), regular wet season runoff which limited hydraulic retention (Braskerud, 2002), and lack of vegetation in the lined study pond cells.

Organic N concentrations decreased at all three locations, albeit only significantly at Locations A and C (Appendix C, Table C1). These positive organic N removal efficiencies agree with Mallin et al. (2012), who reported an average 70 % organic N removal efficiency through a treatment system. In segmented pond systems, the first pond cell slows flow velocity and retains particulate nutrient forms. In contrast, flow in the one-cell pond configuration at Location B lacks segmentation, potentially leading to short-circuiting (with potential direct flow out of ponds) and limiting organic N removal via sedimentation.

Organic N exists in dissolved (DON) and particulate (PON) forms, and removal mechanisms may vary depending on its forms. Removal mechanisms include sedimentation for PON and microbial mineralisation for DON, depending on the labile or refractory composition of organic N for microbial breakdown (Bronk et al., 2007; Mallin et al., 2012). Ponds are generally static systems, where nearly all nutrient transformations occur through exchange processes (Boyd, 1995). Higher NO_3-N concentrations recorded at exit pond cell outlets (Figure 5.5) suggest that mineralisation of organic N to NH_4-N, followed by rapid nitrification to NO_3-N, may have occurred within the ponds. The statistically significant positive organic N removal efficiencies (Appendix C, Table C1) in the two-cell pond configuration may stem from enhanced PON sedimentation due to pond segmentation at Locations A and C.

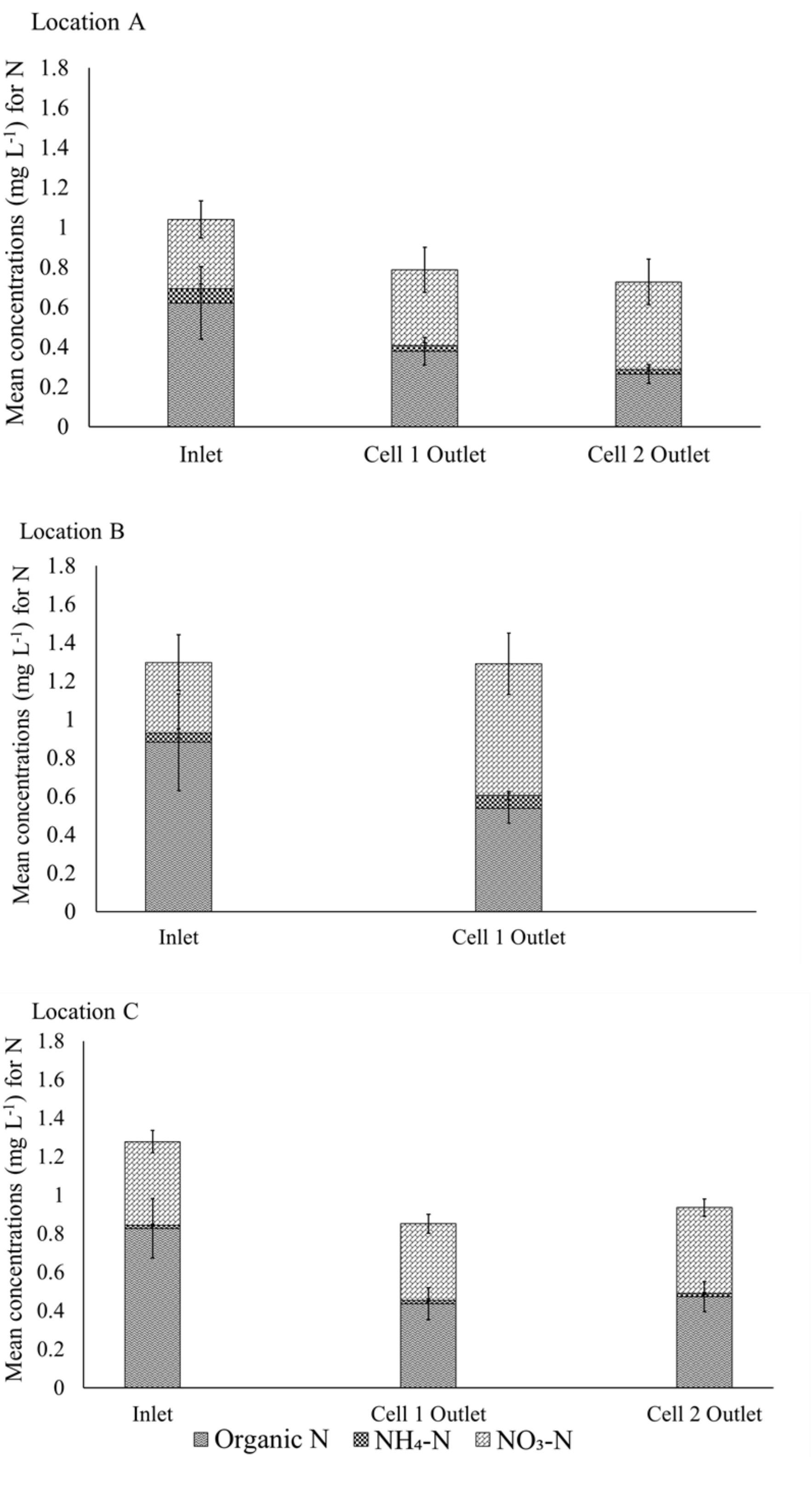


Figure 5.5 Nitrogen (N) mean ± standard error concentrations from sediment pond cells at Locations A, B and C on study farm.

Inorganic nitrogen removal efficiencies varied considerably across locations. At Locations B and C, NH_4-N concentrations increased, whereas it reduced in Location A (Appendix C, Table C1). The average influent NH_4-N concentrations were, however, very low, ranging from 0.02 to 0.072 mg L^{-1}. Similar NH_4-N removal inefficiencies of −61 ± 118 % were reported by Robotham et al. (2021) in a three small online-pond study. The positive mean NH_4-N removal efficiency in Location A (68.11 ± 66.00 %) may be due to shorter hydraulic residence time in these undersized pond cells which may have limited the ammonification of retained organic N in the first pond cell, leaving lower NH_4-N concentrations to travel to the second pond cell and then the outlet. On this assumption, where NH4-N removal inefficiencies recorded in the optimal size ponds maybe due to ammonification of retained organic N.

Through nitrification, NH_4-N concentrations convert into NO_3-N concentrations (Vymazal et al., 1998), adding to the initial NO_3-N concentrations and increasing NO_3-N leaving the ponds. Although not statistically significant, all three locations had negative mean NO_3-N removal efficiencies (Appendix C, Table C1). Studies by Kim et al. (2011), Mallin et al. (2012) and Robotham et al. (2021) report contrasting results of positive mean reductions. Their results may have varied from this study primarily due to the low mean air temperature over the monitoring period of 6.7 ± 0.3 ºC in which this study was conducted. The temperature may have inhibited microbial transformations (e.g. denitrification) for NO_3-N removal, compared to the reported mean removal efficiencies for all seasons. Incorporating vegetation within these pond cells to function as constructed wetlands (Tang et al., 2021) would improve NO_3-N removal via plant uptake and provide carbon for denitrification under anaerobic conditions. Furthermore, the riparian buffers at the end of each sediment pond may remove the NO_3-N, and therefore future studies could monitor the efficacy of both the sediment pond and the riparian buffer in removing nutrients.

5.4.2.2 Phosphorus and TSS removal in sediment ponds

The average influent TP concentration to the sediment ponds ranged from 0.08 mg L^{-1} (Location A) to 0.75 mg L^{-1} (Location C). Locations A and B had TP removal efficiencies of 17.0 ± 38.1 % and 11.7 ± 7.1 %, respectively (Figure 5.6), whereas Location C had a TP removal efficiency of -10.4 ± 9.2 % (Appendix C, Table C1). Excluding three sampling periods, all sampling periods showed negative TP removal at Location C for TP. Notwithstanding the negative TP removal at Location C, mean outlet TP for all locations remained lower than the current P discharge limit of 0.7 mg L^{-1} under the EU Urban Waste Water Treatment Directive (UWWTD 91/271) (European Commission, 2024). This discharge limit was used as a reference guide, as no specific discharge limit guidelines exist for farm water treatment measures.

All locations had positive mean PP removal efficiencies: 47.0 ± 60.3 % at Location A, 7.1 ± 17.4 % at Location B and 1.1 ± 4.4 % at Location C (Appendix C, Table C1). These reductions indicate effective PP removal by sedimentation, consistent with the observations of Shan et al. (2002). There was a similar trend for TSS (Figure 5.7), with mean removal efficiencies of 63.0 ± 79.2 %, 81.5 ± 90.9 % and 57.9 ± 84.7 % at Locations A, B and C, respectively. This demonstrates suspended sediments' contributions to P concentrations (Cooper et al., 2015; Evans et al., 2004), and highlights sediment pond systems' role in trapping particulate pollutants (Gu et al., 2017; Mekonnen et al., 2017). Total dissolved P, comprising DUP and DRP, dominated P in the inlet, ranging from 77.5 % (Location B) to 94.1 % (Location C) of

TP (Figure 5.6). Locations A, B and C had positive mean DRP removal efficiencies of 3.9 ± 19.2 %, 27.9 ± 44.6 %, and 3.0 ± 21.9 %, respectively. The DRP reductions are consistent with the 14.9 % and 29 ± 37 % mean removal efficiencies in the pond treatment studies of Barber (2013) and Robotham et al. (2021), respectively. Adsorption is the principal removal mechanism for dissolved P (Lai & Che, 2008), and this is influenced by the availability of the adsorbing sites. The WSP, which indicates readily available P, of pond sediments at Location C was relatively higher than at Locations A and B (Figure 5.3), indicating P-concentrated sediment. Concentrated P sediments have limited adsorption sites, and this may have potentially lowered adsorption, leading to low P removal at location C. Further research on equilibrium P concentration and sorption analysis on the sediment, however, may be required improve understanding on the adsorption.

The reduction trend of DRP was also observed for DUP at Locations A and B, but increased significantly at Location C ($p < 0.05$). The continuous hydraulic loading and base flow, driven by the connecting groundwater spring emerging through the cambered section of the reconstructed farm roadway into Location C's Pond cells, may have impacted the P removal. Kill et al. (2018) attributed low nutrient removal in a runoff treatment system to constant groundwater flow seeping from adjacent fields. Such conditions create consistent flow currents that reduce residence time (Brown et al., 1981), cause sediment resuspension (Saeed et al., 2019) to release P into the water column (Sinke et al., 1990; Søndergaard et al., 2003) as organic P (DUP), and increase aeration for microbial desorption (Stahlberg et al., 2006; Yu et al., 2022). This finding reinforces the importance of matching pond design to actual local hydrological context and provides a novel idea, by including other characteristics associated with flow such as permanence and seasonal dynamics, where present, into the pond volume estimations.

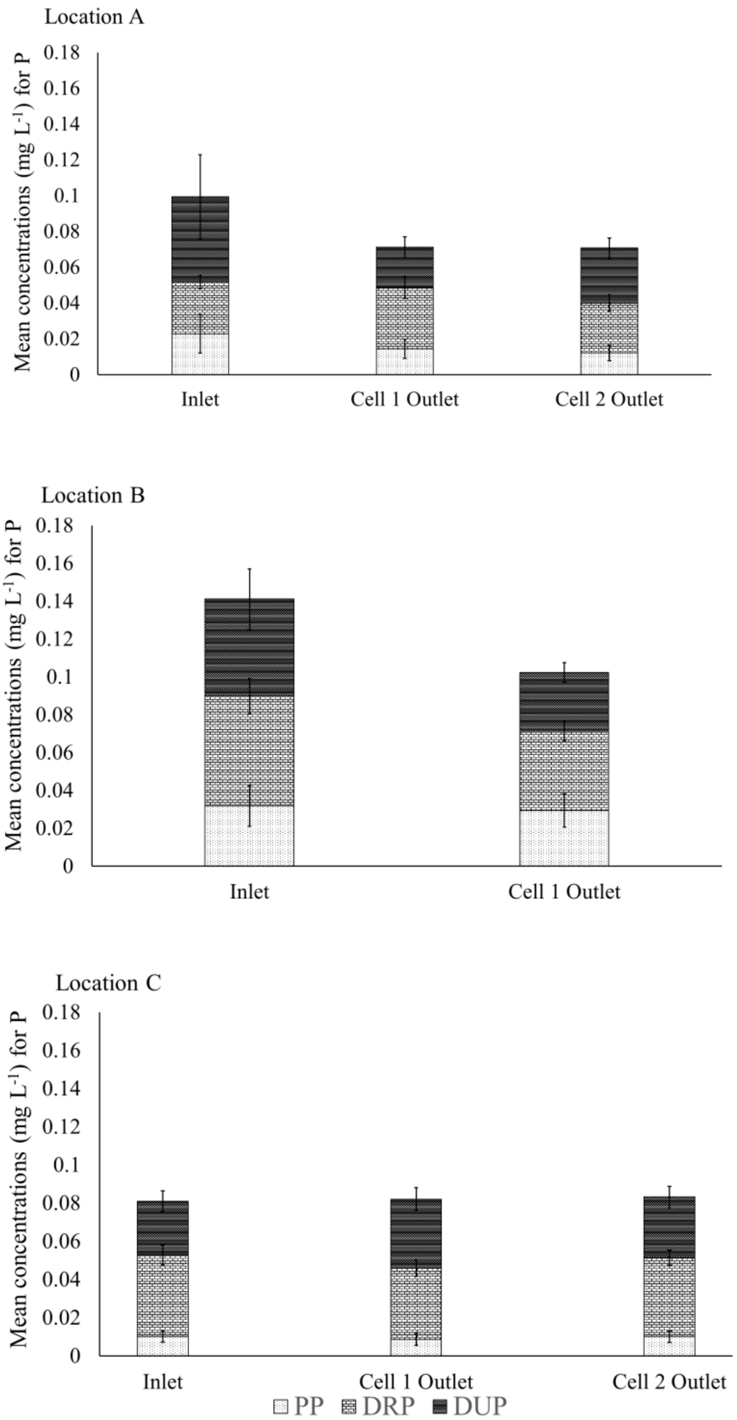


Figure 5.6 Phosphorus (P) mean ± standard error concentrations from sediment pond cells at Locations A, B and C.

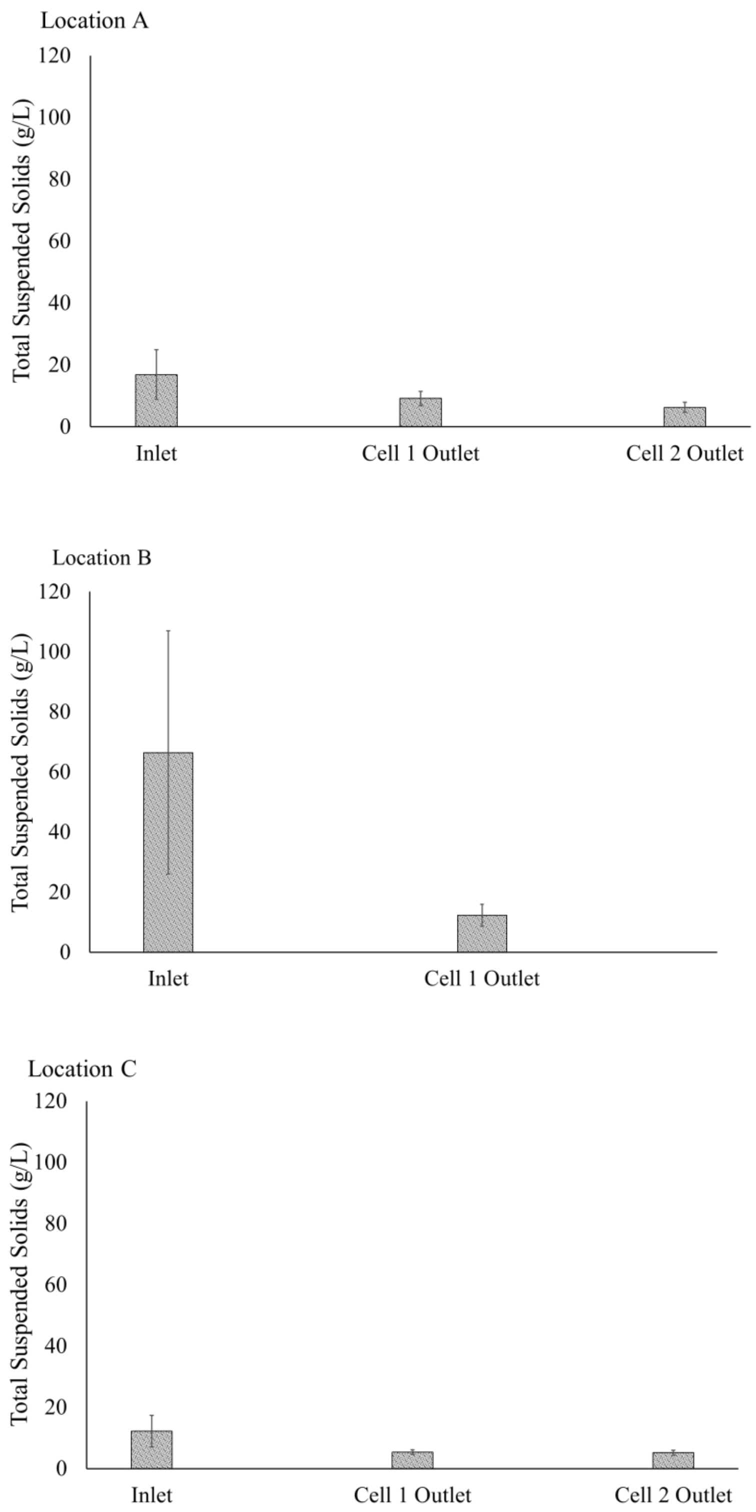


Figure 5.7 Mean total suspended solids (TSS) ± standard error concentrations from sediment pond cells at Locations A, B and C.

5.5 Conclusion

This study showed that sediment ponds, implemented at appropriate locations for managing farm roadway runoff loss to open drainage channels, are effective in removing sediment, TSS and particulate nutrients, but vary in the removal of dissolved nutrients. The trapping of sediment in ponds is dependent on the contributing area as a sediment source and rainfall as a mobiliser, while nutrient removal is dependent on the pond design and site conditions. Policy recommendations delivered through farm advisory services to farmers on future iterations of sediment ponds should promote pond segmentation them into smaller cells, incorporate provisions for accounting for site-specific hydrological conditions such as constant hydrological loadings from groundwater springs (if present), and inclusion of vegetation to improve dissolved nutrient forms to improve their hydrological and biogeochemical functioning. Segmenting ponds into individual cells provides the additional benefit of enabling segregation of pollutant forms to improve overall removal efficiency, without affecting the total retention volume. With the provision of this high and practical knowledge on sediment pond effectiveness (including those with constrained pond design sizes), farmers will be more likely to use sediment ponds for managing farm roadway runoff entering open drainage channels. Long-term monitoring of at least one year to capture all seasonal runoff variations and further research on equilibrium P concentration of sediment are required to make estimations for maintenance measures such as pond dredging. Data from such monitoring will allow estimations of whether maintenance is needed every 2, 5 or 10 years, depending on site-specific conditions.

6. Conclusions and Recommendations

6.1 Overview

Surface drainage systems are used to drain excess field water on poorly drained grasslands. However, the drained water is potentially nutrient-rich and poses a risk to receiving waterbodies. Nutrients in the drained water are dependent on spatially and temporally varying soil, hydrological pathways, climate and farm management factors connected to the surface drainage systems. This creates varying nutrient loss risks across a surface drainage system network. Therefore, there is a need to rank the connectivity risks to identify locations on a drainage network for targeted mitigation.

Chapter 2 reviewed how nutrients are lost to surface drainage systems and evaluated nutrient loss risk classification methods in identifying risky locations in surface drains. It was found that nutrient loss in surface drains is dependent on a complex interaction of soil, climate, hydrology and farm management factors along the nutrient transfer continuum. At the time of writing, no study (to the author's knowledge) had considered all the spatial and temporal variations in classifying the risk of nutrient loss in surface drains on poorly drained grasslands. The first experimental study (**Chapter 3**) validated the presence of hydrological pathways and associated nutrient loss risks. It improved the existing P-only connectivity risk classification system for surface drains by developing an integrated N and P connectivity risk system. This new classification system assigns a risk category for every individual surface drain and provides information on the dominant N and P species loss for each connecting hydrological pathway and suggests a targeted mitigation strategy. **Chapter 4** built on these findings by developing a semi-quantitative risk model to identify high-risk surface drains within a network. Chapter 4 found that the majority of the high-risk surface drainage channels on poorly drained dairy grasslands were associated with the farmyard but found that surface roadways were also significant. Using data from this risk assessment, **Chapter 5** applied a “right measure, right place” approach to design, construct and operate sediment ponds to remove nutrients (especially particulate nutrients) and sediment from farm roadway runoff. Although the findings of this study apply to grasslands on heavy textured soils in high rainfall areas, the application may differ for other grassland systems in other geographic areas with different soils, climates and agricultural practices.

6.2 Conclusions

The main conclusions from this study are:

1. The integrated connectivity risk ranking system developed in this study, considering both N and P, showed not all hydrological pathways are active for every surface drain. The source, connection and presence (and transformation process) of hydrological pathways to surface drains influence the speciation and concentration of N loss in surface drains. This provides valuable information for implementing a more targeted nutrient-specific mitigation strategy in surface drains in heavy textured grassland farms in high-rainfall areas and improves the previous P-only risk classification system.
2. Surface drains connected to the farmyard are the highest risk on a farm. However, where no connection to a farmyard exists, N and P loss into surface drains varies depending on the connecting hydrological pathway. Nitrate and DRP dominate losses from subsurface in-field drains, groundwater springs, upwelling and seepage, whilst NH_4-N and particulate P dominate losses from surface roadways. Instances of multiple hydrological pathways connecting to a single surface drain exist, and the selection of

appropriate mitigation measures in such instances, should take cognisance of their N and P loss risks.

3. A semi-quantitative risk assessment model was developed, which considered potential water quality risk impacts along the source-mobilisation-pathway-receptor (S-M-P-R), to produce a colour-coded risk classification system by which surface drains can be risk assessed. This risk classification system enables the production of risk maps that identify potentially high or very-high risk open drainage channels and highlights associated contributing parameters that would require attention during mitigation on dairy farms with heavy textured soils.
4. Surface drains with a moderate risk of N and P losses dominate the surface drainage network (68 %) across the farms studied in this thesis, whilst surface drains with a high-risk class comprised only 9 % of the surface drainage network, representing the lowest proportion within the network. This relatively low amount of high-risk surface drains means that mitigation measures may be targeted in specific areas, particularly when resources are limited. The high proportion of moderate risk class within the surface drainage network means that preventive measures may be taken to eliminate their escalation into a high-risk class in the future. Within the S-M-P-R, the source contributes most of the nutrient loss in surface drains. However, hydrological pathways, such as farm roadways, can also be potentially high contributors to nutrient loss.
5. Sediment ponds are effective in mitigating particulate nutrients and sediments from roadway runoff and are recommended for adoption by grassland farmers. When configured into discrete cells, sediment ponds are efficient in pollutant removal and particularly in sediment retention.

6.3 Recommendations

The main recommendations from this study are:

1. Further work is recommended in mapping the integrated nutrient loss connectivity risk classification on surface drains nationally and integrating the risk classification into the national pollution impact potential (PIP) maps using machine learning and AI for online access. This will improve the N and P loss predicting capabilities of existing nutrient loss risk systems and provide extra information for target mitigation.
2. A hydrological pathway risk assessment should consider flow volumes so that nutrient loads may be included in the integrated nutrient loss connectivity risk. The inclusion of flow volumes will provide information on actual nutrient loads to surface drains, and on low flow and high flow seasonal nutrient loss risk to further enhance the "right timing" of targeted mitigation. This will complement the "right place, right measure" approach in mitigation strategies.
3. Future work should consider temporal variations in the semi-quantitative risk assessment of nutrient losses in surface drainage systems. This should include varying seasonal rainfall characteristics (intensity and amounts) and within-drain vegetation characteristics (presence, form and abundance), to enhance an understanding of the temporal risk of losses.
4. Further research work is recommended to advance the risk assessment into catchment-scale hydrological models beyond field-scale to ascertain the cumulative hydrological losses and to inform policy development for catchment management.
5. Long-term monitoring of sediment ponds and an assessment of physicochemical properties of sediment is needed to improve knowledge on removal mechanisms and

management. Runoff flows entering sediment ponds vary seasonally, and long-term monitoring of at least one year will capture all seasonal flow variations to make insightful estimations of sediment build-up and nutrient removal efficiency. This information will inform the design of sediment ponds to reduce accidental overspills.

6. Further work is recommended on testing all mitigation measure options, including costings, to make inform decision on mitigating nutrient losses on surface drains. Such information will provide comparative assessment of the mitigation efficiencies and resources needed for different measures at catchment-scale. This may be used to develop data sheets to aid advisors in providing guidance to farmers during implementation.
7. Nutrients, especially N, quickly change and transform from one form to another as they move through the NTC. Future work may require emissions assessment on ammonia, nitrous oxide or nitric oxide to enable the complete loss risk assessment and highlight absence or presence of pollution swapping.

6.4 Wider implications

This study informs the effective management of nutrient losses associated with surface drains to enable the sustainable use of poorly drained dairy grasslands and increase economic returns and food production.

- The new knowledge will inform farmers on poorly drained grasslands about the presence and contribution of varying nutrient losses from various sources. With such knowledge, farmers become aware of nutrient loss risk that could be managed and mitigated, thereby improving their adherence to action programs to reduce diffuse sources losses (e.g., Good Agricultural Practice Regulations, in line with the Nitrates Directive (91/676/EEC)).

- The implementation of semi-quantitative risk assessment would support existing nutrient loss pathway maps such as the EPA pollution impact maps by improving the "right place, right measure" philosophy by identifying the right places in CSAs. As a tool to characterise nutrient loss risk from surface drains, it will reduce the waste of resources and will encourage the implementation of cost-effective targeted mitigation.
- Empirical evidence of the effectiveness of sediment ponds, even in constrained areas, hopefully will encourage their future use and will assist in meeting EU WFD regulations while enabling optimum farm production to support the Irish economy and increasing global animal-based food demands.

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Appendix A

A1: Frontiers in Environmental Science

An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T., & Tuohy, P. (2024).

Article associated with Chapter 3.

frontiers | Frontiers in Environmental Science

TYPE Original Research
PUBLISHED 19 February 2024
DOI 10.3389/fenvs.2024.1337857

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OPEN ACCESS

EDITED BY
Alan Steinman,
Annis Water Resources Institute, United States

REVIEWED BY
Adam Canning,
James Cook University, Australia
Katelyn Lawson,
Auburn University, United States

*CORRESPONDENCE
P. Tuohy,
patrick.Tuohy@teagasc.ie

RECEIVED 13 November 2023
ACCEPTED 23 January 2024
PUBLISHED 19 February 2024

CITATION
Opoku DG, Healy MG, Fenton O, Daly K, Condon T and Tuohy P (2024), An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management.
Front. Environ. Sci. 12:1337857.
doi: 10.3389/fenvs.2024.1337857

An integrated connectivity risk ranking for phosphorus and nitrogen along agricultural open ditches to inform targeted and specific mitigation management

D. G. Opoku[1,2], M. G. Healy[2], O. Fenton[3], K. Daly[3], T. Condon[1] and P. Tuohy[1]*

[1]Teagasc, Animal & Grassland Research and Innovation Centre, Moorepark Fermoy, Cork, Ireland, [2]College of Science and Engineering, Civil Engineering and Ryan Institute, University of Galway, Galway, Ireland, [3]Teagasc, Crops, Environment and Land Use Centre, Wexford, Ireland

Introduction: On dairy farms with poorly drained soils and high rainfall, open ditches receive nutrients from different sources along different pathways which are delivered to surface water. Recently, open ditches were ranked in terms of their hydrologic connectivity risk for phosphorus (P) along the open ditch network. However, the connectivity risk for nitrogen (N) was not considered in that analysis, and there remains a knowledge gap. In addition, the P connectivity classification system assumes all source–pathway interactions within open ditches are active, but this may not be the case for N. The objective of the current study, conducted across seven dairy farms, was to create an integrated connectivity risk ranking for P and N simultaneously to better inform where and which potential mitigation management strategies could be considered.

Methods: First, a conceptual figure of known N open ditch source–pathway connections, developed using both the literature and observations in the field, was used to identify water grab sampling locations on the farms. During fieldwork, all open ditch networks were digitally mapped, divided into ditch sections, and classified in terms of the existing P connectivity classification system.

Results and Discussion: The results showed that not all source–pathway connections were present across ditch categories for all species of N. This information was used to develop an improved open ditch connectivity classification system. Farmyard-connected ditches were the riskiest for potential point source losses, and outlet ditches had the highest connectivity risk among the other ditch categories associated with diffuse sources. Tailored mitigation options for P and N speciation were identified for these locations to intercept nutrients before reaching receiving waters. In ditches associated with diffuse sources, nitrate was introduced by subsurface sources (i.e., in-field drains and groundwater interactions from springs, seepage, and upwelling) and ammonium was introduced through surface connectivity pathways (i.e., runoff from internal roadways). On similar dairy farms where open ditches are prevalent, the integrated classification system and mapping procedure presented herein will enable a targeted and nutrient-specific mitigation plan to be developed. The same methodology may be applied to develop a bespoke integrated connectivity risk ranking for P and N along agricultural open ditches in other areas.

KEYWORDS

water quality, nutrient loss, grassland, drainage management, connectivity pathways, North Atlantic Europe, agricultural ditches

1 Introduction

Open ditch networks, also referred to as "surface ditch network," are installed in poorly drained soils to remove excess water, control the water table, and aid with grass production and utilization (Tuohy et al., 2016; Hertzberger et al., 2019). These networks comprise a series of connected and unconnected sections that receive nutrients from a variety of surface and subsurface pathways, all of which can then be transported to other sections or associated waterbodies (Kröger et al., 2007; Herzon and Helenius, 2008; Moloney et al., 2020). Connectivity is defined as the transfer of energy and matter across two landscape zones, whereas disconnectivity is the isolation of these zones (Chorley and Kennedy, 1971). Identifying the connectivity of these systems enables mitigation strategies to be implemented at optimal locations where nutrients can be reduced or restrained (e.g., intercepting the pathway, slowing the flow, or removing some of the nutrients in the water) to minimize the impact on the receiving waterbody (Fenton et al., 2021). Research continues to help farmers optimize farm management practices (baseline) and engineering solutions (above baseline) (Moore et al., 2010; Schoumans et al., 2014; Carstensen et al., 2020). Many studies on open ditches have focused on nutrient dynamics (Sukias et al., 2003), sediment attenuation capacity (Ezzati et al., 2020; Mattila and Ezzati, 2022), nutrient loss attenuation potential by vegetation (Soana et al., 2017; Zhang et al., 2020), dissolved organic carbon dynamics (Tiemeyer and Kahle, 2014), organic matter composition (Hunting et al., 2016), ditch management (Dollinger et al., 2015; Hertzberger et al., 2019), and indirect greenhouse gas emissions (Hyvönen et al., 2013; Clagnan et al., 2019). However, few studies have investigated the role played by open ditch connectivity in the transfer of nutrients from the source to the receptor. Such studies may provide vital information to ascertain the positioning of ditch mitigation option and the dominant nutrient species it is required to target. Moreover, there is a poor understanding of processes leading to the immobilization and transformation of nutrients within soil and drainage systems along the hydrological pathways into ditches (Deelstra et al., 2014). For efficient mitigation of nutrient loss from open ditch networks, a conceptual understanding of how nutrient sources and their pathways connect to the open ditch system must be established.

The general trend and pathways of agricultural pollutants have been well-documented and are summarized in Figure 1. In summary, nutrient entry into ditches is predominantly from diffuse sources and often through the complex surface and subsurface pathways determined by soil type, climate, landscape position, farm management, and nutrient input sources (manure or fertilizer type) (Granger et al., 2010; Monaghan et al., 2016; Gramlich et al., 2018). These factors regulate the hydrology, the primary driver of nutrient transfer, and the terrestrial and aquatic

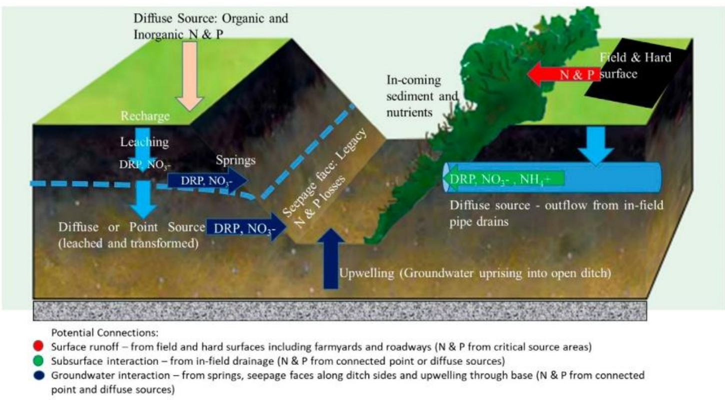

FIGURE 1
Conceptual figure of an open ditch showing all potential nitrogen and phosphorus sources (point and diffuse), pathways, and discharge connections [modified from Teagasc (2022) and Simpson et al. (2011)].

biogeochemistry that defines the type and form/species of nutrients entering open ditches and subsequently discharging to associated waterbodies (Sukias et al., 2003). Conceptually, phosphorus (P), either as particulate P (PP) or dissolved reactive phosphorus (DRP), and nitrogen (N), as ammonium (NH_4^+) or nitrate (NO_3^-), are transported from fields or hard surfaces like roadways through surface flow pathways into open ditches (Figure 1).

As shown in Figure 1, any groundwater-to-open ditch water connection represents a subsurface interaction distinct from in-field drain connections. In this scenario, typically, P is in the form of DRP, and NO_3^- represents mineralized N that has become mobilized due to infiltrating water. This N is primarily lost from diffuse sources in fields due to fertilization and grazing of animals. Clagnan et al. (2018) have shown the conversion of N to NH_4^+ in poorly drained soils, which can be discharged in waters from in-field drains within the groundwater-to-open ditch water connections (Needelman et al., 2007; Valbuena-Parralejo et al., 2019). The presence of NO_3^- in open ditch networks suggests more permeable connectivity pathways that eventually seep into open ditches along seepage faces or upwell as the water table rises, whereas the presence of NH_4^+ suggests less permeable routes before discharge occurs. Groundwater springs represent a distinct groundwater storage component that protrudes onto fields, which are often drained by the installation of an intersecting pipe into an open ditch below the spring. This creates a direct discharge point within the open ditch (Figure 1). The presence of this discharge may change during dry periods as the water level decreases below the base of the open ditch.

Moloney et al. (2020) used this concept to rank the connectivity risk (from highest to lowest) for P along agricultural open ditches. The riskiest open ditches were those directly connected to farmyards (farmyard connection ditches) and watercourses (outlet ditches), while the least risky open ditches included secondary and outflow ditches (disconnected ditches did not pose any risk of connectivity). The system devised by Moloney et al. (2020) conceptualized P sources and pathways with the aim of disconnecting P losses before discharge to associated waterbodies. The current study takes the same approach but creates an integrated connectivity risk ranking that considers both N, which discharges into the open ditch network via surface and subsurface pathways (Figure 1), and P. Such integration necessitates a thorough understanding of N and P biogeochemical cycles, how sources are connected along different surface and subsurface pathways to the open ditch network, and how this network is connected and delivered to the adjoining aquatic system (e.g., river). Accounting for attenuation along the pathway and within the open ditch network is a constraint within the current conceptual framework. Therefore, there is a need to integrate N into the connectivity risk ranking so that a more holistic mitigation management strategy may be designed (i.e., source protection on the farm and "right measure, right place" in the open ditch).

The objective of this study was to derive a farm-scale integrated open ditch risk ranking for both P and N loss risk based on connectivity to inform future mitigation management on heavy textured, grassland dairy farms. To fulfil this objective, seven farms were selected with open ditch networks on heavy textured soils. A conceptual figure illustrating the trends and pathways of agricultural pollutants for an open ditch is presented. The open ditch networks were mapped during a ground survey, and a qualitative water sampling campaign was conducted (based on the conceptual figure) to validate the presence or absence of pathways for N and P. This enabled an integrated classification of an open ditch network ranking to be developed. Mitigation options for each ditch class are presented.

2 Materials and methods

2.1 Site selection and characteristics

Seven grassland dairy farms on poorly drained soils geographically located across the SW and NE of Ireland were selected to represent a variety of agronomic dairy production systems and biophysical settings (Table 1). As per the Ireland EPA soil and subsoil maps (Fealy et al., 2009), the soil types on these farms varied from organic to mineral soils. The majority of these farm fields were imperfectly or poorly drained, necessitating an *ad hoc* network of artificial drainage installations on the farms. The grazing area of each farm ranged from 28 to 45 ha. Intensive dairy farm management practices were observed on all farms. Morgan's extractable soil P test (Wall and Plunkett, 2020) was used to determine the agronomic excesses and deficiencies in plant available P for fields of each farm. The farms in this study were located in high-rainfall areas with an average rainfall of 1092.5 mm. The average farm slope was measured on all seven farms, as it could influence open ditch connectivity.

2.2 Ground survey and mapping connectivity pathways for N into P connectivity risk ditch categories

A ground survey was carried out on all the farms during winter (November 2021 to March 2022) to characterize the field boundaries and surface and subsurface networks on each farm. This period was selected following multiple field visits carried out across all seasons in the previous year. This period was identified as the best hydrological period when connectivity pathways were active for grab sampling. Drainage network features such as open ditches connected to the farmyard and the proximity of the open ditch to waterbodies were noted on each farm during the ground survey. In addition, the connectivity pathways for N into open ditches from in-field drains, farm roadways, groundwater springs, seepage, and upwelling as per the conceptual figure (Figure 1) throughout the drainage network were noted during this time. During the ground survey, all drainage network data, such as of drain locations, flows and connections, and sampling locations, were recorded by using an electronic device with ESRI ArcGIS Field Maps mobile software (ESRI, 2024).

Open ditches were identified as manmade open drains usually sited along the field edges to carry excess water from the field and farm. Surface waterbodies (1st- and 2nd-order streams) in and around each farm, defined as those appearing on the national ordnance survey maps (6-inch maps) (osi.ie), were mapped onto each farm map before each ground survey.

TABLE 1 Summary of agronomic and soil data and associated in-field drainage percentages across case study farms.

Farm #	Farm size	NUE[a]	% of the number of fields with a high P index[b]	Soil OM[c] (%)	Annual rainfall (mm)	Farm topography slope angle range (°)	Dominant soil type	Drainage classes[d] (%)				Major soil type[d] (%)			% of fields with in-field drains[e]
	(ha)	(kg N/ ha)						Poor	Imperfect	Moderate	Well	Mineral	Humic	Organic	
1	43	27	16.3	16.2	1,086.3	2–3	Humic surface water gley	30.9	52.9	16.2	0	69.1	30.9	0	48.4
2	40	23	40.0	16.7	1,283.7	3–11	Humic surface water gley	8.8	39.7	35.1	16.4	68.4	31.6	0	34.1
3	45	24	19.6	30.6	1,002.4	0	Groundwater gley	50.1	38.5	11.4	0	46.2	31.0	22.8	72.5
4	37	32	10.3	18.0	1,320.2	4–8	Humic brown podzolic	45.1	0.9	54	0	58.4	41.6	0	13.6
5	41	35	59.4	8.4	900.0	0.6–0.9	Surface water gley	57.5	17.2	2.1	23.1	88.2	11.8	0	78.4
6	39	45	21.5	14.8	1,035.6	1–8	Typical surface water gley	42.1	3.5	25.1	29.3	84.3	10.9	4.9	25.2
7	28	42	41.7	12.1	1,019.6	5–7	Typical surface water gley	50.2	5.1	42.5	2.2	97.1	1.7	1.2	69.6

[a]Nitrogen use efficiency.
[b]High P index (index 4) fields have soils with excess P concentration (above 8 mg L-1, measured as Morgan's P, on grassland).
[c]OM, organic matter (Corbett et al., 2022a; Corbett et al., 2022b).
[d]Data from Tuohy et al. (2018); Tuohy et al. (2021).
[e]% field with in-field drain = (size of drained field/total farm size) × 100%.

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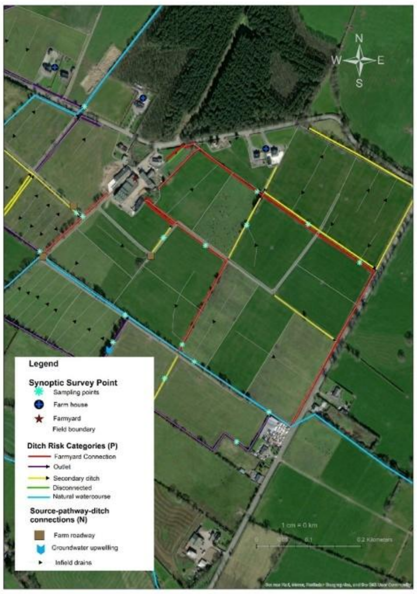

FIGURE 2
Example of a farm output map (for farm 5) showing the ranked classification risk along the open ditch network for P (color-coded into categories of connectivity risk) and all conceptualized N open ditch connectivity pathways to individual open ditch sections. For in-field drains, arrows indicate fall and flow direction toward open ditch sections, with a particular P risk indicated by the existing color-coding scheme of Moloney et al. (2020).

Information from the ground survey observations and qualitative interviews with farmers on drainage networks were used to digitize and map farm and field boundaries and the open ditch network (open ditches, sub-surface in-field drains, and drainage outlets) and associated connectivity pathways for N (Figure 2). For the open ditch network within each farm, each ditch was assigned a ditch category using their connection to a farmyard, watercourse, neighboring farm, other ditches on the same farm, and also their non-connection to any other part of the open ditch network after Moloney et al. (2020) (Table 2). These categories are as follows: (1) farmyard connection ditch, (2) outlet ditch, (3) outflow ditch, (4) secondary ditch, and (5) disconnected ditch (Figure 2) using ArcMap GIS software (version 10.5).

On each assigned ditch category, the connectivity pathways for N (Table 3), where present, were mapped within this open ditch network using the conceptual figure (Figure 1) as a guide during fieldwork to integrate the N connectivity pathway risk into the P connectivity risk open ditch categories. To identify the connectivity pathways, landscape position was taken into account, especially for assessing the interaction of groundwater with an open ditch section. Groundwater seeping through open ditch bank sides and groundwater upwelling through the base of the open ditch were identified as groundwater seepage and upwelling, respectively (Table 3), and were classified together as one connectivity pathway. Roadways were identified as a connectivity pathway when there were site observations of water flow and eroded/gully surface (due to continuous past water flows) from the farm roads into a nearby open ditch. Groundwater springs were identified as high-flow groundwater purging out into open ditches either over the surface or through pipes. Subsurface in-field drains were all piped drains directed into ditches but were differentiated from piped springs with their low and intermittent flows into the open ditches.

The length of the open ditches and farm and field boundaries were measured in ArcGIS and compared for each farm, as shown in Table 4. In addition, the occurrence of a particular N connectivity pathway was calculated as a percentage of the total number of N connectivity pathways observed for each farm and for each open ditch category.

2.3 Grab water sampling campaign to assess integrated nutrient connectivity pathways

Water quality parameters change over time, depending on the local climatic conditions and farming practices (Huebsch et al., 2013). In the present study, the objective was to establish a link or connection (see Figure 1) between the source and pathway to the open ditch network. Therefore, "snapshot" sampling in spring (March) presented a good opportunity to collect qualitative data.

In spring (March) 2022, a total of 210 water samples were collected directly from 105 sampling sites in open ditches throughout the drainage network across all farms during a one-time sampling event following the procedure of Moloney et al. (2020). These sampling sites reflected connectivity pathways presented in Figure 1. March was selected for sampling because

TABLE 2 Definition and description of open ditch categories for the P classification system of Moloney et al. (2020).

Ditch category	Description
1. Farmyard	A ditch/pipe that connects a farmyard to the drainage connection network or directly to a surface waterbody
2. Outlet	A ditch that connects the drainage network to a surface waterbody
3. Outflow/transfer	A ditch that carries drainage water across the farm boundary onto the neighboring land
4. Secondary	A ditch that typically flows perpendicular to the slope of the land connecting two larger open ditches or running through a field for excess water removal
5. Disconnected	A ditch that is not connected to the overall drainage network but may have groundwater connectivity potential

TABLE 3 Criteria for identifying N connectivity pathways on open ditch categories and associated source of connection.

N connectivity pathway	Source of connection	Criteria description[a]
In-field drains	Subsurface	Evidence of in-field pipe drains connecting into ditches, usually with less water flow
Farm roadway	Surface	Evidence of farm roadway and hard surface runoff connectivity with the open ditch network (directly during rainfall or indirect signs such as established rills and breakthrough points)
Groundwater springs	Subsurface	Evidence of natural springs or pipe springs (with high water flow) connecting into ditches
Groundwater upwelling or seepage	Subsurface	Evidence of groundwater seeping from either the base or side of a ditch into the ditch

[a]Criteria description (Teagasc, 2022).

TABLE 4 Summary of open ditch data including the proportion of the open ditch network accounted for by different P open ditch categories for each case-study farm.

Farm number	Field perimeter (m)	% Perimeter as ditch	Total ditch length (m)	Proportion of total ditch length (%)				
				1. Farmyard connection	2. Outlet	3. Outflow	4. Secondary	5. Disconnected
1	16,471.5	44.3	7,290.4	10.7	0	18.4	70.2	0.7
2	21,524.1	9.0	1,935.1	6.8	59.4	33.8	0	0
3	19,737.9	35.4	6,990.7	5.7	22.6	9.4	62.4	0
4	16,572.3	17.2	2,847.4	28.4	23.3	4.6	10.5	33.2
5	13,085.9	43.5	5,692.4	25.5	39.5	0	34.3	0.7
6	16,966.5	52.6	8,916.3	8.5	22.4	7.2	60.9	0.9
7	9,607.5	28.9	2,773.3	34.2	11.7	15.8	38.3	0
Average	16280.8	33.0	5206.5	17.1	25.6	12.7	39.5	5.1

this month is hydrologically active in Ireland and all pathways interact with the open ditch network (e.g., groundwater upwelling, seepage, and springs), as observed from the previous year's field visits. As this study aimed to validate established connectivity risk (water and the presence or absence of N and P) between open ditch types and adjoining surface waterbodies and did not aim to elucidate the load or impact of this connection, a temporal water sampling survey was not required. It is acknowledged that the connectivity level at the time of sampling water is influenced by the precipitation level (both antecedent and current). Therefore, sampling was undertaken when both surface and subsurface pathways were most active, and such data were used to validate the source and hydrologic connectivity with the open ditch network.

The number of samples collected was dictated mainly by the observations of connectivity pathways on open ditches during the initial fieldwork campaign. As such, open ditches that had surface or subsurface connectivity pathways (Table 3) noted in the earlier survey were prioritized for sampling. These observations were used to validate surface, subsurface, and groundwater flows that entered open ditches on the case study farms. However, some sampling points had no N connectivity pathways. Only four ditch categories from Table 2 (farmyard connection, outlet, outflow, and secondary ditches) were sampled for water across the seven case study farms. Shallow disconnected ditches (category 5 in Table 2) were dry, which indicated no N connectivity with perched or true water tables at the time of sampling. These acted as storage and recharge areas for groundwater during rainfall periods. At each water sampling location, two 50-ml samples (filtered on-site using 0.45-μm filter paper and unfiltered) were collected for dissolved and total P analyses, respectively. Grab water sampling was carried out in the mapped ditch categories on each farm, provided water was present in the open ditch. The grab water sampling taken directly from an open ditch was conducted within 1 m downstream of in-field drain outlets, farm roadways, groundwater springs, and groundwater seepage/upwelling, where present, in the open ditch categories. All water samples were kept in an ice box during sampling and transportation and then tested within 1 day of sample collection.

Filtered water samples were analyzed for DRP and total dissolved phosphorus (TDP) using a Gallery discrete analyzer (Gallery reference manual, 2016) and a Hach Ganimede P analyzer, respectively. The total dissolved phosphorus (TDP) was measured by acid persulfate oxidation under high temperature and pressure. The unfiltered water samples were analyzed for nitrite (NO_2-N), NH_4-N, total oxidized nitrogen (TON), and total reactive phosphorus (TRP) using the Gallery analyzer. Total phosphorus (TP) was analyzed using the Ganimede P analyzer. Phosphorus was measured colorimetrically by the ascorbic acid reduction method (Askew and Smith, 2005), where the 12-molybdophosphoric acid complex is formed by the reaction of orthophosphate ions with ammonium molybdate and antimony potassium tartrate (catalyst) and reduced ascorbic acid. All samples, reagent blanks, and check standards were analyzed at the Teagasc Johnstown laboratory following the Standard Methods (APHA, 2005).

All quality control (QC) samples/check standards are prepared from certified stock standards from a different source than calibration standards. Quality control samples were analyzed at the beginning and end of every batch, for every 10 samples within a batch, and if the QC fell outside limits, samples were repeated back to the last correct QC. Blanks were included in every batch, and approximately 10% of samples were repeated. Tolerances range up to a maximum of ±7.5% of the nominal value. All instruments used were calibrated in line with the manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, particulate phosphorus (PP) was calculated by the difference between TP and TDP, and dissolved unreactive phosphorus (DUP) was calculated by the difference between TDP and DRP.

2.4 Data analysis

To validate the link between the conceptualized connectivity source–pathways and their introduction of N and P into the open ditch system, data from the spring season synoptic survey were analyzed statistically to differentiate the nutrient concentrations for the various open ditch categories and also for the various connectivities to ascertain if they varied from each other. As the data for each water quality parameter were not normally distributed, Kruskal–Wallis analysis was undertaken to find out the significant differences between farmyard connection, outlet, outflow, and secondary ditch categories and also between the conceptualized N connectivity pathways (in-field drains, internal roadways, springs, and seepage/upwelling) within and across the outlet, outflow, and secondary ditch categories for all the water quality parameters (NH_4-N, NO_3-N, TN, DRP, DUP, TP, and PP). Data were analyzed using R studio software version 4.0.2 (2020). Where significant differences were observed using an alpha level of 0.05 (95% confidence level), the pairwise Wilcoxon rank-sum test was further used to find the differences between the means of the pairs. Microsoft Excel software version 16.0 (2016) was used to find a correlation between the number of occurrences of in-field drains and the percentage of drained fields on poorly draining soil farms.

3 Results

3.1 Analysis of the open ditch networks

All five ditch categories, classified by Moloney et al. (2020), were identified using the criteria outlined in that work. The average percentage of the total ditch network in all farms was 17.1%, 25.6%, 12.7%, 39.5%, and 5.1% for farmyard connection, outlet, outflow, secondary, and disconnected ditches, respectively (Table 4). Farm 2 contained the fewest drainage categories (3 out of 5).

3.2 Observations relating to conceptualized N connections within the open ditch networks

Based on the criteria for identifying N connectivity pathways (Table 3), 52% of all the open ditch network sampling points were observed to have N connectivity pathways interacting with them. The N connectivity pathways to open ditches considered in this study were mainly connected to secondary ditches, followed by farmyard connection, outflow, and outlet ditches, with no N connectivity pathway to disconnected ditches (Supplementary Table S1). For each ditch category (Table 2) sampled in this study, the percentages of the occurrence of different N connectivity pathways are shown in Figure 3. Among these N connectivity pathways across all ditch categories, in-field drains were the most common (representing 64%), followed by groundwater springs, internal roadways, and groundwater upwelling/seepage, respectively, representing 20%, 11%, and 5% of the sampling points (Supplementary Table S1). The occurrence of observed in-field drains was positively correlated to the percentage of drained fields on case study farms ($R^2 = 0.35$).

Farms 2 and 4, which had the lowest percentage of in-field drained fields (Table 1), had relatively high connectivity of groundwater springs to open ditches (Supplementary Table S1). Aside from farm roadway connectivity pathways to open ditches on Farm 2, roadway connectivity pathways to open ditches were found to be highest on farms with a flat topography, particularly farms 3 and 5. Groundwater upwelling/seepage connectivity to ditches was uncommon. There was an absence of groundwater upwelling and seepage connectivity pathways on outflow and farmyard connection ditches and roadway connectivity pathways on outlet ditches across all farms. In addition, there was evidence of multiple N connectivity pathways to individual ditches on some farms.

3.3 Validation of N connectivity pathway using the synoptic survey

The average TN and TP concentrations were significantly higher in farmyard connection ditches (Figure 4) than in outlet, outflow, and secondary ditches ($p < 0.01$). Across the outlet, outflow, and secondary ditch categories, NO_3-N was the dominant N species, contributing on average to 44.7% of TN at sampling points near N connectivity. Only 10.6% of TN comprised NH_4-N within these ditch categories. The highest average NO_3-N across these ditch categories was observed in groundwater springs (1.90 mg L^{-1}), followed by in-field drains (0.75 mg L^{-1}), groundwater upwelling (0.65 mg L^{-1}), and roadways (0.17 mg L^{-1}) (Supplementary Table S1). In addition, NO_3-N at groundwater springs was dissimilar ($p < 0.05$) to NO_3-N at roadways and in-field drains (Figure 5A). High concentrations of NO_3-N were also measured on roadways, where NH_4-N is conceptualized as being dominant (Figure 1), on secondary ditches. However, NH_4-N dominated TN across these ditches at sample points near roadways, with 25.3% composition as opposed to 6.9% of NO_3-N. Ammonium-N concentrations across these ditch categories were not statistically significant ($p > 0.05$).

No consistent trends in species of TP were observed across the outlet, outflow, and secondary ditch categories. Among these ditch categories, TP concentrations were relatively high in secondary ditches, in which PP was predominant (Figure 5B). Across the outlet, outflow, and secondary ditch categories, PP was statistically significant ($p < 0.05$), particularly between in-field drains and roadway connectivity pathways, and DRP was statistically significant ($p < 0.01$), particularly between roadways and groundwater springs. Comparing P species for each N

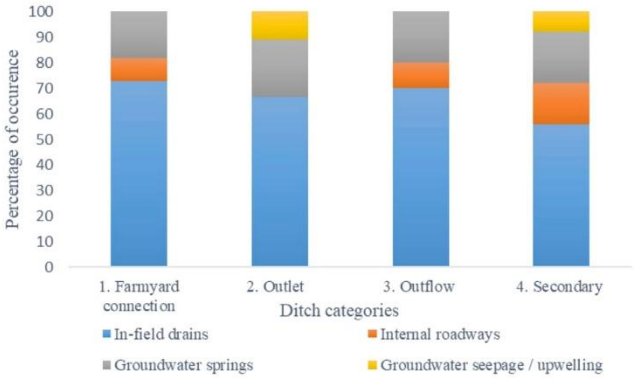

FIGURE 3
Percentages of the occurred N connectivity pathways for the ditch categories.

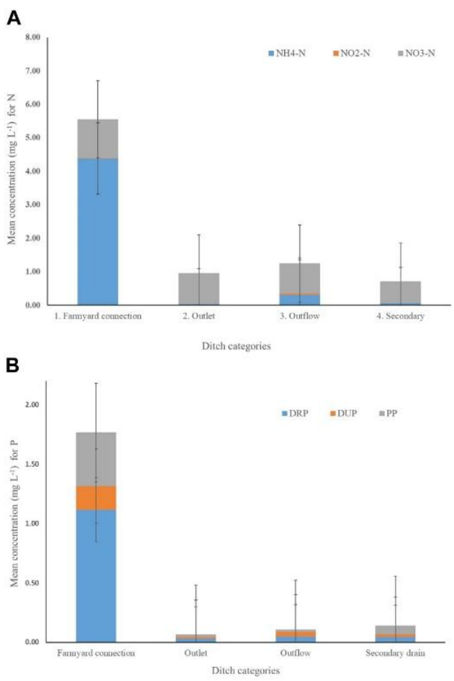

FIGURE 4
(A) Nitrogen (N) and **(B)** phosphorus (P) mean + standard error (SE) concentrations within the open ditch categories across case study farms.

connectivity pathway, average PP concentrations were found to be the highest in groundwater upwelling/seepage (0.24 mg L^{-1}), followed by roadways (0.12 mg L^{-1}), groundwater springs (0.04 mg L^{-1}), and in-field drain (0.02 mg L^{-1}) connectivity pathways, whereas average DRP concentrations were the highest in roadways (0.19 mg L^{-1}), followed by groundwater upwelling/seepage (0.04 mg L^{-1}), in-field drains (0.03 mg L^{-1}), and groundwater springs (0.01 mg L^{-1}).

4 Discussion

4.1 Observations on ditch categories and associated N connectivity pathways

Of the seven farms surveyed, disconnected and secondary ditches comprised the lowest and highest average percentages of the total ditch length, respectively. This result is consistent with that of Moloney et al. (2020), who recorded similarly low and high average percentages for total ditch length on varying soil grasslands in Ireland. Disconnected ditches are ineffective for excess field water removal within the drainage system and exist either as blocked normal ditches or as created disconnecting ditches that remove field runoff or precipitation water by infiltration or evaporation. Disconnected ditches, when wet, may hold water with vegetation and potentially provide denitrification or create pollution swapping by the release of greenhouse gases such as nitrous oxide (N_2O) or nitric oxide (NO).

Secondary ditches, as the most prevalent ditch category, had the most N connectivity pathways, of which in-field drains were the most prevalent (Figure 3). Secondary ditches connect to other ditch categories from the central farm fields, and due to the farm slopes in that areas, frequent shallow water table (Clagnan et al., 2018) for potential for connectivity pathways may occur. As the majority of the farms in this study contained poorly drained soils (Table 1), a positive, albeit weak, correlation ($R^2 = 0.35$) between the number of occurrences of in-field drains (Supplementary Table S1) and the percentage of drained fields (Table 1) on poorly draining soil farms was observed. Both the number of occurrences of in-field drains and the percentage of drained fields help in regulating water table levels

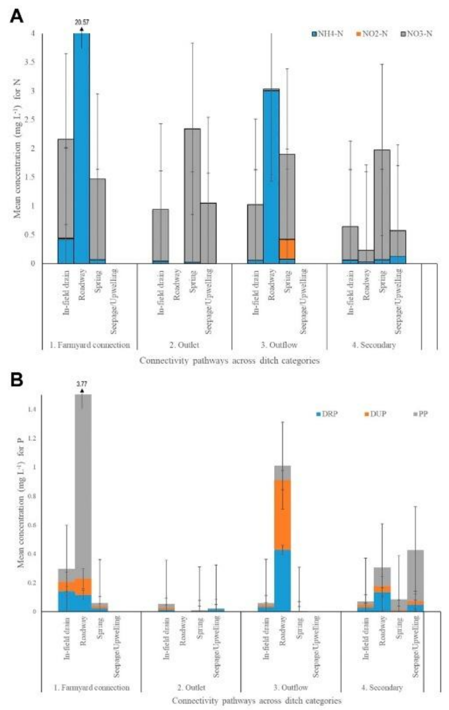

FIGURE 5
(A) Nitrogen (N) and **(B)** phosphorus (P) mean + standard error (SE) concentrations within associated connectivity pathways in sampled open ditch categories across case study farms.

and supporting grass growth functionality, so they were positively correlated.

4.2 Hydrochemistry across P ditch categories and consideration of N connectivity pathways

Higher TN and TP average concentrations were measured in farmyard connection ditches relative to the other ditch categories, which were similar to the findings of Moloney et al. (2020), Harrison et al. (2019), and Ezzati et al. (2020). In the farmyard connection ditches, the TN and TP concentrations were nearly 3 times higher than the TN standard limits of 2.5 mg L^{-1} set by the European Union for estuarine waters (Wuijts et al., 2022) and 15 times higher than TP standards of 0.1 mg L^{-1}, as proposed by Wetzel (2001). While both Edwards et al. (2008) and Mockler et al. (2017) identified farmyards as point sources for high nutrient loss, the former argued that runoff from farmyards has been overlooked and not duly considered a major nutrient loss hotspot. Such runoff may lead to high nutrient concentration fields near the farmyard relative to fields further away (Fu et al., 2010), and these potentially may enter open ditches near the farmyard to create major downstream water quality problems. Unlike ditches (associated with point sources), the lower TP and TN concentrations in outlet, outflow, and secondary ditch categories may be associated with diffuse nutrient sources. Studies have shown that diffuse sources, relative to point sources, have lower TN and TP concentrations (Pieterse et al., 2003; Edwards and Withers, 2008). Management of some of these diffuse sources is problematic as they are difficult to locate in a landscape (Harrison et al., 2019). However, their impact on the deterioration of receiving waterbodies is substantial, and therefore needs to be managed (Andersen et al., 2014; Bradley et al., 2015). Diffuse sources depend on landscape and other management factors, which influence diffuse N and P mobilization, transformation, and delivery into the ditches (Granger et al., 2010; Schoumans et al., 2014). However, notable among these factors are the hydrological conditions on which diffuse nutrient release strongly depends (Edwards and Withers, 2008; Chen et al., 2013). This, coupled with biogeochemical factors, which may vary within a landscape, influences the spatial and temporal distribution patterns of diffuse N and P, including the pathways by which they enter and leave farms (Clagnan et al., 2019; Grenon et al., 2021). Nutrient losses from the diffuse sources are delivered into open ditches along surface and subsurface pathways, creating hotspots of nutrient loss in certain open ditch categories, which need to be characterized and potentially mitigated. Climatic, landscape, and management factors all have a role to play in when and where impacts occur. These could have contributed to the higher TN concentrations in water samples that were measured near N connectivity pathways than at locations with no N connectivity pathways within the outlet, outflow, and secondary ditch categories, and also for TP in the outflow ditch category. This observation aligns with those of the reported works of Ibrahim et al. (2013) and Valbuena-Parralejo et al. (2019) on in-field drains, Fenton et al. (2021) and Rice et al. (2022) on roadways, Soana et al. (2017) on groundwater springs, and O'Callaghan et al. (2018) on groundwater upwelling/seepage.

Nitrate was the dominating N species in in-field drains, groundwater springs, and upwelling connectivity pathways in outlet, outflow, and secondary ditch categories (Figure 5A). This may be attributed to their connection to a subsurface N source, which comprises leached N from animal excreta and fertilizer that may have been nitrified to NO_3-N (Necpalova et al., 2012). In poorly drained grasslands, nitrification may have been increased by the high in-field drainage density (Table 1), which enhanced N preferential flow (Van Der Grift et al., 2016) and limited potential N attenuation (Clagnan et al., 2019; Valbuena-Parralejo et al., 2019). The average NO_3-N concentration was highest in groundwater springs and in-field drains. Factors such as the presence of these N connectivity pathways within the shallow subsurface region, nearness to the soil surface (where farm management mostly occurs), and exposure to N sources at the groundwater–ground surface intersection spots (particularly for groundwater springs; Infusino et al., 2022) could have contributed to the high NO_3-N concentrations in these locations. In contrast, NH_4-N was the most dominating N species measured for roadway connectivity pathways across the outlet, outflow, and secondary ditch categories, especially where animal excreta were observed. This observation aligns with that of Fenton et al. (2021), who observed that roadways draw surface nutrient sources, high in

NH_4-N, as runoff from soil and animal excreta into nearby ditches and streams. Although important, redox reactions were not considered in the present study.

For TP concentrations across outlet, outflow, and secondary ditch categories, P concentrations were relatively low compared to those in the farmyard connection ditch category. However, such TP concentrations in the outlet, outflow, and secondary ditch categories were still high enough to cause eutrophication downstream, if undiluted. High TP concentrations measured in secondary ditches may be related to the impacts of farm management activities including grazing and farm machinery movement, which is intense within the central fields of most farms where secondary ditches lie as connecting ditch links. These contribute to the erosion of ditch sides and associated deposition of soils in the secondary ditches, as reflected in the higher PP concentrations observed. High TP concentrations measured near roadways on outflow ditches may be due to animal excreta, run-on deposits from farmyards, fields, and poached surfaces as a result of animal and machinery movement (Fenton et al., 2021). Both PP and DRP can trigger eutrophication in waterbodies and may pose a risk to downstream waterbodies. However, this depends on their closeness, connection, and mitigation along the pathway to water sources within agricultural landscapes.

Such information from the study provides an additional insight into the source, connection, and presence (and transformation process) of N in ditch categories from a previous study by Moloney et al. (2020), who observed high NH_4^+ and NO_3^- concentrations in all ditch categories, except for the outlet ditch, where high NO_3^- and low NH_4^+ were measured, and disconnected ditches, where NO_3^- concentration was found to be dominant. The risk ranking of connectivity along the open ditch for N and P does not determine the impact of the nutrients being lost to the associated waterbody; it simply establishes the N connectivity pathway if it is present.

4.3 Deriving a connectivity risk for N into P agricultural open ditch categories

The evidence of N concentrations in the ditch water chemistry from Moloney et al. (2020) and the current study informs an improved ditch connectivity risk category system (Table 5). This is a valuable information tool for environmental sustainability officers to enhance water quality management and mitigation options for N and P losses on dairy grassland farms with heavy textured soils in high-rainfall areas. It considers both the connectivity pathways, through which N can be introduced to a ditch network, and their associated N species.

In the current study, all of the conceptualized N connectivity pathways (Figure 1) established from the literature were present, but not in all of the sampled P risk ditch categories developed by Moloney et al. (2020) (Supplementary Table S1). For instance, the established general trends and connectivity pathways of groundwater seepage and upwelling were not present on farmyard connection and outflow ditches. Moreover, the grab water sampling data results validated all the conceptualized N connectivity pathways present in ditches (Figure 5A), except groundwater seepage and upwelling. The dominance of high NO_3-N concentrations at in-field drains and springs and high NH_4-N concentrations at roadways within farmyard connection ditches indicated a point source of pollution arising from their connection to the farmyard aside from the hydrology-induced N concentrations. Farmyards pose the greatest nutrient loss risk on farms due to high nutrient concentration in discharges (Vedder, 2020), and like other point sources, they are independent of hydrology (Edwards and Withers, 2008). As such, primarily managing the farmyard wastewater before discharge into connecting ditches for mitigating nutrient connectivity to water sources is essential (NFGWS, 2020) before deployment along/within ditch interventions.

For the other sampled outlet, outflow, and secondary ditch categories, all N conceptualized pathways were observed, except for internal farm roadway on outlet ditches and groundwater seepage and upwelling on outflow ditches (Supplementary Table S1). In outlet, outflow, and secondary ditch categories, the ditch water synoptic data validated the conceptualized NO_3-N and NH_4-N for all the observed N connectivity pathways, except farm roadway connection on secondary ditches (which was invalid with NO_3-N dominance over conceptualized NH_4-N from hard field surface flow pathways). Nitrate dominated in-field drains, groundwater springs, upwelling, and seepage connectivity pathways, and NH_4-N dominated farm roadways across the outlet, outflow, and secondary ditch categories, as conceptualized in Figure 1.

Assessment of the N connectivity pathway within ditch category 5 could not be included in the study due to the unavailability of water samples in this ditch for validating conceptualized N connectivity pathways. Moloney et al. (2020) showed that disconnected ditches pose relatively less risk for nutrient loss among the ditch categories, and therefore, merit less focus during nutrient loss mitigation for surface water. However, such low nutrient concentrations could be leached into groundwater, and therefore may require mitigation interventions to prevent leaching.

To apply this research in practice, once open ditches are investigated and mapped, a category should be assigned for an individual open ditch, after which the available N connections for that ditch are noted. All of these connections, in combination, will aid in the future mitigation management strategy. It is unlikely, for example, that more than one mitigation option will be installed in a single open ditch. Therefore, the information gathered from Table 5 can be used to ensure that correct nutrients and their speciation are targeted for mitigation in the open ditch. Mitigation options may be a combination of those that limit diffuse and point sources. For example, with respect to diffuse sources, strict adherence to action programs to reduce losses is important (e.g., Good Agricultural Practice Regulations, in line with the Nitrates Directive (91/676/EEC)). With respect to roadway runoff, NH_4^+ mitigation options are available and have been outlined in Fenton et al. (2021) and Rice et al. (2022) (e.g., diversion bars to move runoff to a buffer area of at least 1.5 m, cambering farm roadways, and directing flow onto adjacent fields). Adopting a two-stage ditch design may reduce high PP concentrations (King et al., 2015; Hodaj et al., 2017; Faust et al., 2018). With respect to the subsurface N connectivity pathways (in-field drains, groundwater springs, upwelling, and seepage), in-ditch management practices may control the flow and

TABLE 5 An updated integrated ditch connectivity ranking that considers both phosphorus and nitrogen coupled with suggested strategies to reduce nutrients from ditches on dairy farms.

P Ditch category	Description	Validated N connection with category	Associated source	Future mitigation management
1. Farmyard Connection	A ditch/pipe that connects a farmyard to the drainage network or directly to a surface waterbody. These connections pose the highest risk and should be prioritized in terms of future management	Subsurface interaction	In-field drains (pipes, moles, or gravel moles; older variation) bring P and N from fields to the open ditch All forms of P and N are potentially lost through this pathway to the ditch, with NO_3^- and DRP dominating	Management practices that disconnect sub-surface drainage system discharges into the open ditch • These may include adherence to correct land drainage design, installation guidelines, and maintenance • Use of end-of-pipe land drainage mitigation options including low-grade weirs Baker et al., (2016), filter cells, cartridges, and structures (King et al., (2015); Goeller et al., (2020); Liu et al., (2020) (see discussion for details) Strict adherence to good farming practices to minimize diffuse losses and leaching of nutrients to sub-surface drainage system that are connected to the open ditch • These may include in-ditch measures such as sediment traps, bioreactors, and filters to slow the flow and control nutrient loads (Fenton et al., (2021)
		Surface runoff	Farmyards and hard surfaces including farm internal roadways bring P and N forms, dominated by NH_4^+ and PP from raw organic waste, loss to the ditch	Management practices that disconnect the farmyard from the open drainage ditch and internal farm roadway network are needed specifically within 100 m of the farmyard in this category • These may include measures that prevent roadway runoff from entering the open ditch using low-cost diversion bars or surface modifications Fenton et al., (2021). There must be a buffer of at least 3 m EPA Ireland, (2020) to reduce the runoff impacts of surface waters
		Groundwater interaction	Natural springs bring shallow groundwater P and N, dominated by NO_3^-, into open ditches through piped drains	Strict adherence to good farming practices to minimize diffuse losses • These may include end-of-pipe mitigation measure where the spring has been piped e.g., vegetated buffer spots Faust et al., (2018) and filter cells, cartridges, and structures using various materials Ibrahim et al., (2015); King et al., 2015; Penn et al., (2020) (see discussion for details). The full list of materials is reviewed in Ezzati et al. (2020)
2. Outlet	A ditch that connects the drainage network to a surface waterbody	Subsurface interaction	In-field drains (pipes, moles, or gravel moles; older variations) bring P and N forms, dominated by NO_3^-, from fields to the open ditch	Management practices that disconnect sub-surface drainage system discharges into the open ditch • These may include adherence to correct land drainage design, installation guidelines, and maintenance • Use of end-of-pipe land drainage mitigation options such as constructed wetlands (Tanner et al., (2005); King et al., (2015) (see discussion for details) Strict adherence to good farming practices to minimize diffuse losses and leaching of nutrients to sub-surface drainage system that are connected to the open ditch

(Continued on following page)

TABLE 5 (*Continued*) An updated integrated ditch connectivity ranking that considers both phosphorus and nitrogen coupled with suggested strategies to reduce nutrients from ditches on dairy farms.

P Ditch category	Description	Validated N connection with category	Associated source	Future mitigation management
				• These may include in-ditch measures such as sediment traps, bioreactors, and filters to slow the flow and control of nutrient loads (Fenton et al. 2021)
		Groundwater interaction	Natural springs bring shallow groundwater, dominated by NO_3^- concentration, into ditches through piped drains	Strict adherence to good farming practices to minimize diffuse losses • These may include end-of-pipe mitigation measures where the spring has been piped, e.g., vegetated buffers (Faust et al. 2018), filter cells, cartridges, and structures, using various materials (Ibrahim et al., 2015; King et al., 2015; Penn et al., 2020) beneath piped springs located on the ditch. The full list of materials is reviewed in Ezzati et al. (2020)
		Groundwater interaction	Seeping and upwelling deep groundwater, dominated by NO_3^-, enters into ditches	Strict adherence to good farming practices to minimize diffuse losses • In terms of groundwater upwelling or spring connectivity, in-ditch intervention that slows the flow and mitigates nutrients using bioreactors, two-stage ditches, filters, and vegetated ditches (King et al., (2015); Faust et al., (2018) may be introduced after spring connectivity and before the outlet to reduce dissolved and particulate nutrients entering waters
3. Outflow/ transfer	A ditch that carries drainage water across the farm boundary through neighboring land	Subsurface interaction	In-field drains (pipes, moles, or gravel moles; older variations) bring P and N, dominated by NO_3^-, from fields to the open ditch	This drainage water will pass to an adjoining farm and will be mitigated as another landowner's arm management plan. Some mitigation can occur in outflow ditches using mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across the farm landscape
		Surface runoff	Farm internal roadways introduce NH_4^+- and DRP-dominated hard surface water to the ditch	This drainage water will pass to an adjoining farm and will be mitigated as another landowner's farm management plan. Some mitigation can occur in outflow ditches using mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across the farm landscape
		Groundwater interaction	Natural springs connect shallow groundwater, dominated by NO_3^- concentration, to ditches	This drainage water will pass to an adjoining farm and will be mitigated as another landowner's farm management plan. Some mitigation can occur in outflow ditches using mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across the farm landscape

(Continued on following page)

TABLE 5 (*Continued*) An updated integrated ditch connectivity ranking that considers both phosphorus and nitrogen coupled with suggested strategies to reduce nutrients from ditches on dairy farms.

P Ditch category	Description	Validated N connection with category	Associated source	Future mitigation management
4. Secondary	A ditch that typically flows perpendicular to the slope of the land connecting two larger ditches. It can also occur as an open ditch running through a field to collect and remove large excesses of surface water	Subsurface interaction	In-field drains (pipes, moles, or gravel moles; older variations) bring P and N, dominated by NO_3^- from fields to the open ditch	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in secondary ditches using in-ditch mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm
		Surface runoff	Farm internal roadways introduce PP, DRP- and NO_3^--dominated, within the water from a hard surface to the ditch	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in secondary ditches using in-ditch mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm
		Groundwater interaction	Natural springs bring shallow groundwater, dominated by NO_3^-concentration, through piped drains over ditch sides to introduce both PP and NO_3^- into the ditch	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in secondary ditches using in-ditch mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm
		Groundwater interaction	Deep groundwater, dominated by NO_3^-, seeps through ditch side surfaces and/or upwells through the ditch base to introduce PP and NO_3^- into ditches	Mitigation is unlikely to occur in these open ditches as they do not discharge directly to waters but act as conduits. Some mitigation can occur in secondary ditches using in-ditch mitigation management practices provided for farmyard connection and outlet ditches as appropriate, which may increase the efficacy of mitigation across an individual farm
5. Disconnected	A ditch that is not connected to the overall ditch network and may be connected with groundwater	Surface and groundwater interaction	Diffuse source of NO_3^- interacts with the open ditch. Runoff may interact with the open ditch	Connectivity is not present to surface water within the open network, but there may be a groundwater connection which subsequently discharges to surface water. Precautionary practices should be taken at these locations to minimize recharge to groundwater by provision of a soil buffer

the nutrient content leaving the open ditch. These may include sediment traps (Wilkinson et al., 2014), vegetated ditches (Kröger et al., 2008; Soana et al., 2017; Faust et al., 2018), or in-ditch filters or bioreactors (King et al., 2015; Goeller et al., 2020; Liu et al., 2020). Nutrient filtering through vegetation (Moeder et al., 2017) or use of a medium (Ezzati et al., 2020) can only aim to mitigate a small amount of overall nutrients leaving the ditch due to hydraulic retention times needed and bypass flow during high storm events. Furthermore, mitigation practices including the construction of wetlands (Tanner et al., 2005), vegetated buffer zones (Faust et al., 2018), and low-grade weirs (Kröger et al., 2012; Littlejohn et al., 2014; Baker et al., 2016) that may be placed at the end of ditches after the connectivity pathways, especially for farmyard connection and outlet ditch categories, would help limit nutrient loss from these farms. Therefore, all measures need to be considered a package and not in isolation when trying to minimize nutrient and sediment loads leaving an open ditch system. It is worth noting that cooperation at the local level is needed to prevent other mitigation-related problems (such as the polluter pays principle regarding outflow ditches between neighboring farmers) to ensure mitigation occurs before waters are impacted.

5 Conclusion

Distinctly different from a P-only classification system, the integrated connectivity risk classification system for N and P showed that not all source–pathway interactions within open ditches are active. This is a valuable information tool that enables a much more specific and targeted nutrient-specific mitigation approach to be implemented on open ditches in heavy textured grassland dairy farms in high-rainfall areas. The new system avoids the pitfalls of a P-only classification system (i.e., mitigating for P but allowing N to affect water quality unabated). The findings of this study are limited to these field sites and may (or may not) differ in other geographic areas with different soils, climates, agricultural practices, etc. However, the same methodology may be applied to other areas to develop a bespoke integrated connectivity risk ranking for P and N along agricultural open ditches to inform targeted and specific mitigation strategies on those farms. Further assessment of the temporal and spatial variability of soil, weather, drainage system, and general hydrogeochemistry, which influences nutrient connectivity, may be needed to rank the N and P risk in each ditch category.

Data availability statement

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation.

Author contribution

DO: conceptualization, data curation, formal analysis, investigation, methodology, validation, visualization, writing–original draft, and writing–review and editing. MH: conceptualization, funding acquisition, investigation, methodology, software, supervision, validation, visualization, and writing–review and editing. OF: conceptualization, funding acquisition, investigation, methodology, software, supervision, validation, visualization, and writing–review and editing. KD: conceptualization, investigation, methodology, validation, visualization, and writing–review and editing. TC: funding acquisition, methodology, and writing–review and editing. PT: conceptualization, funding acquisition, investigation, methodology, project administration, resources, software, supervision, validation, visualization, and writing–review and editing.

Funding

The author(s) declare that financial support was received for the research, authorship, and/or publication of this article. The authors are grateful to Teagasc for the award of a Walsh Scholarship to the first author (grant number: RMIS-1381) to conduct this research.

Acknowledgments

The authors are grateful to Simon Leach, Asaf Shnel, and Denis Brennan for GIS, fieldwork, and laboratory assistance provided.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2024.1337857/full#supplementary-material

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A2: Frontiers in Environmental Science

A semi-quantitative risk model for heavy textured grassland dairy farms that identifies where nutrients in a surface drainage channel network require mitigation.

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T., & Tuohy, P.

Article associated with Chapter 4.

frontiers | Frontiers in Environmental Science

TYPE Original Research
PUBLISHED 23 September 2024
DOI 10.3389/fenvs.2024.1435418

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REVIEWED BY
Laura Christianson,
University of Illinois at Urbana-Champaign, United States
Pankaj Tiwari,
University of Kalyani, India

*CORRESPONDENCE
P. Tuohy,
patrick.tuohy@teagasc.ie

RECEIVED 20 May 2024
ACCEPTED 09 September 2024
PUBLISHED 23 September 2024

CITATION
Opoku DG, Healy MG, Fenton O, Daly K, Condon T and Tuohy P (2024) A semi-quantitative risk model for dairy farms to pinpoint and break source-pathway connections between nutrient sources and open drainage channel sections.
Front. Environ. Sci. 12:1435418.
doi: 10.3389/fenvs.2024.1435418

A semi-quantitative risk model for dairy farms to pinpoint and break source-pathway connections between nutrient sources and open drainage channel sections

D. G. Opoku[1,2], M. G. Healy[2], O. Fenton[3], K. Daly[3], T. Condon[1] and P. Tuohy[1]*

[1]Animal and Grassland Research and Innovation Centre, Teagasc Moorepark, Fermoy, Ireland, [2]Civil Engineering and Ryan Institute, College of Science and Engineering, University of Galway, Galway, Ireland, [3]Environmental Research Centre, Teagasc Johnstown Castle, Wexford, Ireland

Introduction: On intensive grassland dairy farms in high rainfall areas with poorly drained soils, networks of open drainage channels linked to in-field drainage systems are needed to enable farm operations. Nitrogen and phosphorus point and diffuse sources may be connected to this open drainage channel network along surface and subsurface pathways, with negative impacts upon delivery to the downstream aquatic system.

Methods: This study developed a semi-quantitative risk assessment model by: (1) selecting parameters (categorical or continuous) representing the nutrient transfer continuum and (2) scoring (relative magnitude and impact) the risk of nutrient source connectivity and delivery for every open drainage channel section across seven dairy farms.

Results and Discussion: A Risk Index Classification System consisting of low, medium, high, or very high-risk class was developed, with high or above requiring a mitigation plan. Results showed that 23%, 68%, 9% and 0% of all open drainage channels on study farms were identified as low, moderate, high and very high-risk, respectively. A range from 2% to 25% per farm of the open drainage channels was classified as high-risk that potentially needed mitigation, although none was identified as very high-risk. Two-thirds of the high-risk open drainage channels were connected to the farmyards, with potential for high nutrient loss from point sources. A combined approach of source management and targeted breaking of the pathway (e.g., in-channel filters, water diversion bars) may help minimise nutrient losses from high risk open drainage channels on poorly draining soils.

KEYWORDS
water quality, agriculture, nitrogen, phosphorus, environment, mitigation

1 Introduction

Agricultural landscapes in areas of high annual precipitation and heavy textured soils are characterised by high densities of open drainage channels, which provide outfalls for in-field drainage systems (Shore et al., 2015; Tuohy et al., 2018). Open drainage channels, comprising drainage ditches and smaller streams, are networked to collect and drain away excess water from different parts of a farm to larger water courses (Kröger et al., 2007). Within the open drainage channel network, streams exist as intermittent or perennial natural channels, whereas drainage ditches exist as man-made channels that may be intermittent or perennial, depending on their landscape position and their interplay with subsurface water and groundwater. These open drainage channels perform many functions (Daly et al., 2017; Ezzati et al., 2020) including storage and release of nutrients by sediments, transportation and interception of farm surface and subsurface runoff which may carry nutrients to the larger water courses.

It is important to minimise the source of nutrients and intercept instantaneous and legacy nutrients from farms in high rainfall areas (Fenton et al., 2021; Peyton et al., 2016; Valbuena-Parralejo et al., 2019). In these areas, open drainage channels form an integral part of the source, mobilisation, pathway, and receptor (S-M-P-R) component of the nutrient transfer continuum (Haygarth et al., 2005) (defined as the framework that captures the nutrient-loss influencing factors from source to receptor). Water drained in both natural and man-made open drainage channels may be nutrient-rich from different nutrient sources that are mobilised through point (e.g., farmyard (Martínez-Suller et al., 2010; Vero et al., 2020), farm roadway (Fenton et al., 2021; Rice et al., 2022) and diffuse (Daly et al., 2017; Roberts et al., 2017)) sources. Where hydrological connectivity exists with the surrounding environment, nutrients from these sources travel through different pathways (Wall et al., 2011) to enter open drainage channels. The nutrients are either transformed or remain unchanged along the pathway to the open drainage channel, before being transported to the adjoining waterways (Clagnan et al., 2018). Aside from nutrient transformation, these nutrients can be buffered and/or retained to prevent connectivity losses as they go through the processes and pathways (Deelstra et al., 2014). Understanding the nutrient dynamics and loss risks occurring within an open drainage channel system is critical to assessing, managing and mitigating nutrient losses from farms (Collins et al., 2016; Herzon and Helenius, 2008).

Moloney et al. (2020) ranked connectivity risk for phosphorus (P) loss along man-made open drainage channels and showed that varying levels of connectivity to nutrient source, depending on their geographical position, exist between man-made open drainage channels and surface waters. The highest to lowest connectivity for P loss was as follows: farmyard connection ditch, outlet (a ditch that connects the drainage network to a surface water body), outflow (a ditch that carries drainage water across the farm boundary through neighbouring land), secondary, or disconnected ditch. Opoku et al. (2024) further developed this concept by creating an integrated (i.e., P and nitrogen (N)) ranked connectivity risk incorporating nutrient loss from sources within open drainage channels. That study showed that other factors, i.e., farm management practices, landscape characteristics, and surface and subsurface hydrological connectivity of directly connecting areas, described the risk of P and N loss in categories of man-made open drainage channels. These factors vary spatially and temporally (Harrison et al., 2019; Mellander et al., 2017; Withers and Lord, 2002), even in a very small distance (Adams et al., 2022), and therefore may vary in the nutrient loss risk they pose for individual open drainage channels at different geographic locations on farm. Characterising these factors for individual open drainage channels is essential to assess the risk of connectivity for nutrient losses from an open drainage channel network, but is not well studied. In previous nutrient loss risk studies, open drainage channels were risk assessed largely as a (transport) pathway factor for nutrient loss based on either their presence, density, connectivity to high-risk fields or sloping conditions (Buczko and Kuchenbuch, 2007; Magette et al., 2007; Roberts et al., 2017; Schoumans and Chardon, 2003), thereby limiting a holistic assessment (Granger et al., 2010). Furthermore, in studies where these factors have been used in assessing farm nutrient loss connectivity (Deelstra et al., 2014; Gramlich et al., 2018), their influences on connectivity to open drainage channels under their respective nutrient transfer continuum sections to enable complete understanding of their nutrient loss risks (Haygarth et al., 2005; Murphy et al., 2015) and improve regulations (Wall et al., 2011) have not been evaluated. Such an evaluation could be achieved by exploring a risk assessment of the factors under the nutrient transfer continuum of open drainage channels and may allow mitigation efforts to be optimised to prevent nutrient losses to open drainage channels and transfer to adjoining water bodies.

Risk assessment provides an overall appraisal of the connectivity components for each element (S-M-P-R) of the nutrient transfer continuum to inform their combined implications and relationships for nutrient loss to open drainage channels on farms (Jordan et al., 2005). Risk can be assessed quantitatively (where data are sufficient; Adkin et al., 2014), qualitatively (where data are insufficient; Nag et al., 2020), and semi-quantitatively (a blend of the two, e.g., Rice et al., 2022)). Subjective expert judgment may be used to approximate risk values to inform decision-making (Redmill, 2002; Rice et al., 2022). Different assessment approaches to identify and characterise landscape hotspots for nutrient losses have been documented. These include direct nutrient concentration measurements in open drainage channels (Ezzati et al., 2020; Mattila and Ezzati, 2022), a combination of some nutrient transfer continuum parameters (Alder et al., 2015; Hayes et al., 2023; Fenton et al., 2022), or predictive models (Radcliffe et al., 2015; Vadas et al., 2007; Vadas et al., 2015). A risk assessment to identify open drainage channel sections associated with high-risk nutrient runoff connectivity using all possible field management data, and landscape and hydrological connectivity data across the nutrient transfer continuum for heavy textured farms has not been developed to date. Undertaking an appraisal incorporating these elements will help identify and rank high-risk areas (also known as critical source areas; McDowell et al., 2024) on the open drainage channel network for heavy textured grassland dairy farms for targeted mitigation.

The objective of this study was to develop a semi-quantitative risk model for heavy textured grassland dairy farms that identifies open drainage channel network sections that pose a risk of contributing nutrients to the adjoining aquatic water courses and which require mitigation. Instead of considering only nutrient source connectivity to classify open drainage channel risks for nutrient losses (as in Opoku et al. (2024)), the current study builds on this theory and captures all relevant S-M-P-R factors under the open drainage network nutrient transfer continuum to rank the nutrient loss risk in the open drainage channel network on a farm. To conduct this research, data were collected during field and desk-based studies across seven heavy textured grassland farms in Ireland. These farms are considered representative of heavy

TABLE 1 Nutrient transfer continuum element, parameter description, units, type, relative magnitude score, relative impact score, and denotation.

Nutrient transfer continuum element	Parameter Description	Parameter unit	Parameter type	Relative magnitude (M) score[a]	Denotation	Relative impact (I) score[b]
Source (Point)	Connection to farmyard		Categorical	0 3	No Yes (e.g., pipe discharge, seepage from leaking tanks)	10
Source (Diffuse)	Soil P	mg/L	Categorical	1 3	Adequate (<8.0 m/L) Excessive (≥8.0 mg/L)	5
Source (Diffuse)	N Fertiliser (kg) applied	kg N ha^{-1}	Continuous	Weighted to 0–3		8
Source (Diffuse)	P Fertiliser (kg) applied	kg P ha^{-1}	Continuous	Weighted to 0–3		8
Source (Diffuse)	Nutrient deposition associated with grazing (e.g., urine, dung pats)	Grazed or non-grazed field × grazing frequency	Continuous	Weighted to 0–3 *(Based on grazing field (1 = not grazed, 3 = grazed) × grazing frequency)*		6
Source (Diffuse)	Fertiliser application count	*# per* field	Continuous	Weighted to 0–3		3
Mobilisation	Rainfall	mm	Continuous	1 2 3	Low (<1,000 mm) Moderate (1,000–1,300 mm) High (>1,300 mm)	10
Pathway	Farm roadway runoff		Categorical	0 1 2 3	No[c] Yes – flat slope Yes - moderate slope Yes – steep slope	4
Pathway	Farmyard surface runoff		Categorical	0 1 2 3	No[c] Yes – flat slope Yes - moderate slope Yes – steep slope	3
Pathway	Field surface runoff		Categorical	0 1 2 3	No[c] Yes – flat slope Yes - moderate slope Yes – steep slope	6
Pathway	Subsurface connection from infield drains		Categorical	0 3	No Yes (e.g., low flow discharge from pipes)	4
Pathway	Groundwater connection to ditch		Categorical	0 3	No Yes (e.g., springs, upwelling and seepage)	3
Receptor	Connection to watercourse		Categorical	0 3	No Yes	7

[a]Relative Magnitude score (M) = the relative magnitude of contributing nutrients to an open drainage channel network.
[b]Relative Impact score (I) = subjective evaluation of relative relevance (on a 1–10 scale) for nutrient contribution to an open drainage channel network.
[c]A barrier, e.g., buffer prevents connectivity of this runoff according to EPA (2020) and USDA (2001) with the surface water (man-made or natural) body.

textured, poorly draining soils in Ireland, all receive high rainfall and were subjected to high-resolution data collection on a vast range of static and dynamic variables related to farm management.

2 Materials and methods

2.1 Nutrient transfer continuum framework

A semi-quantitative risk assessment model was developed based on seven intensive grassland heavy textured dairy farms. Using expert opinion and the literature, various parameters that best describe the nutrient transfer continuum between a source and an open drainage channel network (Dollinger et al., 2015; Kleinman et al., 2011; Needelman et al., 2007) were collated and categorised into S-M-P-R components as in Table 1.

2.1.1 Justification to S-M-P-R parameters

2.1.1.1 Source

In a nutrient loss risk assessment, identifying potential sources and their characteristics is critical (Carton et al., 2008; McDowell et al., 2024). Farmyards are largely associated with potential nutrient sources, and connection to them imposes high-risk of direct or indirect discharges of point source nutrients into the open drainage channel network (Moloney et al., 2020; Opoku et al., 2024; Vero et al., 2020). Soil P status of fields directly connected to open drainage channels offers a potential source contribution of soil nutrients that can be readily lost, and dictates the amount of P that can be applied in a mineral or organic soil (Moloney et al., 2020), and is therefore essential as a source parameter. The organic matter proportion in mineral and organic soils determines the adsorption or repulsion of dissolved nutrients unto soil particles (Roberts et al., 2017; Tejada and Gonzalez, 2008) and therefore influences the soil P status. Soil P Indices of 1, 2 and 3 are defined as low risk, while index 4 is defined as high-risk, with all organic soils categorised as index 4 by default (Daly, 2005; Wall and Plunkett, 2016). The amount of P and N fertiliser (kg) applied is one of the major nutrient sources that influences surface and subsurface nutrient losses in open drainage channels (Hart et al., 2004; Ibrahim et al., 2013; Richards et al., 2015; Watson and Foy, 2001). The rate of fertiliser application increases soluble reactive P (SRP) and total P (TP) concentrations in overland flow and drainage water (Watson et al., 2007). On these connecting fields, fertiliser application count is another source parameter that contributes nutrient loss to open drainage channels and may increase nutrient losses especially under wet soil conditions. The grazing status of a field connecting to open drainage channel specifies the risk of another major nutrient source that determines probability of livestock wastes (faeces and urine) being deposited near an open drainage channel (Bilotta et al., 2007; Gary et al., 1983; Hubbard et al., 2004) and damage to soils (that may be high nutrient rich) by trafficking and poaching to runoff into open drainage channels (Cassidy et al., 2017; Doody et al., 2014; Pietola et al., 2005). Its impact varies with grazing frequency (the number of times a grazing field is accessed by animals for grazing), with frequently grazed fields more susceptible to increase nutrient losses (Cassidy et al., 2017; Doody et al., 2014; Hubbard et al., 2004).

2.1.1.2 Mobilisation

Rainfall is the prime mobilising parameter that controls the transfer of nutrients within and around the open drainage channel (Pérez-Gutiérrez et al., 2020; Vadas et al., 2011; Yao et al., 2021).

2.1.1.3 Pathway

Farm roadways that are connected to open drainage channels under the nutrient transfer continuum acts as pathway by which runoff, carrying nutrients, is transferred into the open drain (Maher et al., 2023; Rice et al., 2022). Along the farm roadway network, nutrients may be contributed from the road surface (Davison et al., 2008; Edwards and Withers, 2008; Fenton et al., 2022). The farmyard is another pathway, which comprises hard standing areas that collect rainfall that becomes runoff to the adjacent open drainage channels (Edwards et al., 2008; Vero et al., 2020). The field surface influences runoff to connecting open drainage channels. Field surface is dependent of the soil drainage class (well, moderate, imperfect, and poorly-draining soils) and this dictates the runoff pathway between surface and subsurface pathways (Houlbrooke and Monaghan, 2009). There is high P loss risk from overland flow in poorly drained soils, moderate P loss risk from imperfectly drained soils, low P loss risk from both moderate and well-drained soils (Magette et al., 2007). The subsurface in-field drain pathway influences soil drainage capacity and subsequently the surface and subsurface pathways (Houlbrooke and Monaghan, 2009). Subsurface in-field drains enhance infiltration and other processes in soils (Opoku et al., 2024). Groundwater upwelling or seeping pathways introduces nitrate (NO_3-N) and P into open drainage channels, but depends on many factors such as landscape position and soil type (Opoku et al., 2024). Groundwater composition may be high in nitrate concentrations, especially if the soil processes are modified by drainage (Edwards and Withers, 2008).

2.1.1.4 Receptor

The receptor is associated with the final direct impact on a watercourse (Wall et al., 2011). Watercourse in this regard is defined as any natural river, stream, or lake (but not a man-made drainage channel) (Department of Agriculture Food and the Marine, 2018) identifiable on an Ordnance Survey Ireland 6-inch map (www.osi.ie). In this study, all natural open drainage channels were assumed to have a final connection to a watercourse, with or without any proximity observed during the ground survey.

2.2 Scoring continuous and categorical parameters

The parameters were assigned individual risk scores that were scored arithmetically in a magnitude-impact matrix (Teunis and Schijven, 2019). For each open drainage channel, the risk score for every parameter was calculated by multiplying the score for magnitude (M) for contributing nutrients to an open drainage channel by the score for its relative impact (I) (Table 1) (after Shariff and Zaini, 2013).

Within the risk assessment, data for some parameters were measured quantitatively as *continuous data* (e.g., N fertiliser (kg) rate applied; Table 1), while others were assessed qualitatively as *categorical data* during field observation (e.g., connection to a farmyard; Table 1). As such, the M value for each parameter differed depending on the parameter type.

For continuous parameters, the M value was weighted between 0 and 3 using the formula (Equation 1):

$$(X_i - X_{min}) \times 3 / (X_{max} - X_{min}) \quad (1)$$

where X_i is the on-farm observed data value; X_{min} and X_{max} are the minimum and maximum values observed across all farms.

For categorical parameters, the value was based on literature and/or expert judgement. Either "0" or "1" was scored as the "lowest" and "3" as the "highest" values (Table 1). For each open drainage channel, a total risk score was calculated by summing up all the risk scores for each continuous and/or categorical nutrient transfer continuum parameter for that open drainage channel. A total risk score represents the degree of risk (i.e., the scale of likelihood or propensity at which an open drainage channel contributes nutrients to a watercourse) associated with the blend of complex parameters (Table 1) for nutrient loss across all the open drainage channels on a given farm. Although the risk assessment takes into account the influence of the contributing area to an open drainage channel, the approach of weighting the contributions over the area rather than adding their impacts ensured an unbiased assessment where a larger area of fields surrounding the stretch of an open drainage channel could have led to high-risk. The risk assessment is simple to use, relying on easily accessible farm data, and can be used to assess the relative risk agricultural open drainage channels pose to water quality, without quantifying the nutrient loss.

2.3 Fieldwork to collect nutrient transfer continuum parameter data

Seven farms, dominated by heavy textured soils of a wide variety of bio-physical settings, were selected. These farms represented varying open drainage channel network density and connectivity risk compositions. During winter (November 2021 to March 2022), a field survey was conducted in which all open drainage channel networks were mapped as per Opoku et al. (2024) and Moloney et al. (2020). Open drainage channel network features such as connection to the farmyard, field slope, the proximity to water bodies, and connectivity pathways for nutrients into the open drainage channel network from in-field drains, farm roadways, groundwater springs, seepage and upwelling throughout the open drainage channel network, were noted on each farm. All the information characterising the open drainage channel network was recorded using an electronic device with ESRI ArcGIS Field Maps mobile software (version 21.4.0) (ESRI, 2024) during the field survey. This information was transferred to 'geographic information system' (GIS) mapping software, ArcMap GIS software (version 10.5). Data on other parameters for the nutrient transfer continuum elements was obtained from previous studies (Corbett et al., 2022a; Corbett et al., 2022b; Tuohy et al., 2021) and ongoing data collection by participating farmers and field agents. The data were downloaded and collated with data from the field survey, and the parameters in Table 1 were assigned an M score for every open drainage channel network across the farms.

In applying nutrient loss risk magnitude to areas that have never been calibrated, errors may prevail due to the unknowns in parameter settings and adjustments, and reliance on experts' opinions to set model parameters without calibration (Sharpley et al., 2017). However, the adoption of systems that are assessed and approved (as suggested by Bhandari et al. (2017); Nelson et al. (2017)) enhanced the robust calibration of the parameters for the risk assessment.

2.4 Formation of risk classification system

Total risk score values for every open drainage channel for all seven farms were split into four categories of equal intervals to produce four potential risk classes (i.e., low risk, moderate risk, high-risk, and very high-risk). The range was determined by the possible highest and lowest score that could occur as per the risk assessment scoring system developed. The risk classes were developed by Equation 2:

$$\left(TRS_{high} - TRS_{low}\right) / 4 = I_e \tag{2}$$

where TRS_{high} and TRS_{low} are the highest and lowest total risk score values recorded across the seven farms, and $I_{.e.,}$ is the interval between the four risk classes. These were colour-coded as green, yellow, orange, and red, respectively, on farm maps. Such maps provide information on the open drainage channels that are potential critical hotspots for nutrient losses on heavy-textured dairy farms. Risk classes in high and very high-risk ranges are identified as hotspots that may require mitigation measures.

2.5 Synoptic water sampling across dairy farms

Water quality parameters change over time, depending on the local climatic conditions and farming practices (Huebsch et al., 2013). At 105 sampling points throughout the drainage network across all farms, a total of 210 water samples (a pair of filtered and unfiltered at each sampling point) were collected during each season (sampling event) for 4 seasons (Spring (March) 2022 to Winter (January) 2023). The sampling was carried out across all 4 seasons to capture hydrological fluctuations and conditions, including surface and subsurface connectivity as per Opoku et al. (2024). As this study aimed to assess the risk of the open drainage channels, the water N and P chemistry only validated the potential nutrient losses from the open drainage channel network surroundings and did not aim to elucidate the load or impact of this connection. Except for disconnected ditches (which were mostly dry), all man-made open drainage channels (farmyard connection, outlet, outflow, and secondary ditches; Moloney et al., 2020) and natural open drainage channels were sampled. At each water sample location, two 50 mL samples (filtered on-site using 0.45 μm filter paper and unfiltered) were collected for dissolved and total P analyses, respectively. All water samples were kept in an ice box during sampling and transportation, and then tested within 1 day of sample collection.

Filtered water samples were analysed for dissolved reactive phosphorus (DRP) and total dissolved phosphorus (TDP) using a Gallery discrete analyser (Gallery reference manual, 2016) and a Hach Ganimede P analyser, respectively. Total dissolved phosphorus (TDP) was measured by acid persulphate oxidation, under high temperature and pressure. The unfiltered water samples were analysed for nitrite (NO_2-N), NH_4-N, total oxidised nitrogen (TON), and total reactive phosphorus (TRP) using a Gallery analyser. Total phosphorus was analysed using the Ganimede P analyser. Phosphorus was measured colourimetrically by the ascorbic acid reduction method (Askew and Smith, 2005), where the 12-molybdophosphoric acid complex is formed by the reaction of orthophosphate ion with ammonium molybdate and antimony potassium tartrate (catalyst) and reduced ascorbic acid. All samples, reagent blanks, and check standards were analysed following the Standard Methods (APHA, 2005). All quality control (QC) samples/check standards are made from certified stock standards from a different source than calibration standards. Quality control samples were analysed at the beginning and end of every batch, and every 10 samples within a batch, and if the QC fell outside limits, samples were repeated back to the last correct QC. Blanks were included in every batch and approximately 10% of samples were repeated. Tolerances range up to a maximum of ±7.5% of nominal value. All instruments used were calibrated in line with manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, particulate phosphorus (PP) was the difference between TP and TDP, and dissolved unreactive phosphorus (DUP) was the difference between TDP and DRP.

TABLE 2 The characteristics (length (m) and number) of open drainage channels *per* farm.

Farm #	Number of open drainage channels per farm	Average length	Length of all open drainage channels per farm (m)		
			Total length	Natural open drainage channel average length	Man-made open drainage channel average length
1	25	291.50	7,290	n/a	203
2	9	271.38	3,799	382	188
3	40	509.23	25,971	1898	170
4	16	397.44	14,308	716	142
5	19	372.71	14,163	1,030	197
6	49	134.95	10,526	322	122
7	13	204.27	4,494	860	139

TABLE 3 Risk classification system (risk class and score ranges) for risk assessment model for open drainage channels on heavy textured dairy farms.

Risk class	Risk score classification ranges	
Low	14.0	60.7
Moderate	60.8	107.5
High	107.6	154.3
Very high	154.4	201.0

3 Results and discussion

3.1 Open drainage channel characteristics

The total length and the number of open drainage channels in the farms are shown in Table 2. The length of an open drainage channel characterised the field area of contribution influencing the connectivity and potential risk of nutrient loss to an open drainage channel. Opoku et al. (2024) reported that multiple connectivity pathways may exist on a single open drainage channel. Although the relationship between the presence of connectivity pathways in open drainage channels and the length of the open drainage channels was not assessed in that study, longer open drainage channel lengths may have high connectivity, resulting in a potentially higher risk of nutrient loss. However, other parameters such as soil chemistry (Daly et al., 2017; Ezzati et al., 2020), slope, design (Hodaj et al., 2017), and vegetation (Soana et al., 2017) may also influence nutrient loss.

3.2 Risk classification system

Table 3 presents the risk classification system ranges based on the minimum and maximum possible total risk score from the risk assessment scoring system. These risk classification ranges were the basis on which risk class output maps for open drainage channel networks on each farm were developed (Figure 1).

Although the possible lowest and highest total risk score are 14.0 and 201.0 according to the risk assessment scoring system (Table 3), the actual lowest and highest total risk scores recorded for the open drainage channels for the farms studied were 35.9 (Farm 4) and 144.4 (Farm 4), respectively. This indicates the highest total risk score across the farms reached only about 72% of the potential maximum total risk score. Of the 171 open drainage channels on all seven farms, 23%, 68%, 9%, and 0% were ranked as low, moderate, high, and very high-risk classes, respectively (Figure 2). Data from individual farms were similar to the overall trend (Figure 2), except for Farm 6, where the majority (57%) of the open drainage channels ranked as low-risk.

Across the high-risk open drainage channels, the total risk score varied, with 144.4 being the highest recorded (a farmyard connection ditch) on Farm 4 and 109.9 being the lowest (a farmyard connection ditch) on Farm 7. The 9% high-risk open drainage channels across the study farms were mostly on farmyard connection and outlet ditches (Table 4). This result is similar to Opoku et al. (2024) and Moloney et al. (2020), who found that farmyard connection ditches were potentially the riskiest.

Agricultural pressures on waterbodies in Ireland are associated with excess nutrients, mainly present as NO_3-N or DRP (EPA, 2023a). Phosphorus dominates in poorly drained soils, such as those included in this study, while N loss is more likely to vary depending on other specific site conditions (EPA, 2023a). In Ireland, the EPA considers good water in rivers to have NO_3-N concentrations of less than 1.8 mg L^{-1} and DRP concentrations of less than 0.035 mg P L^{-1} (EPA, 2023b). While open drainage channels assessed in these study farms are different water bodies from rivers as defined on national ordnance survey maps (6-inch maps) (www.osi.ie), comparisons of NO_3-N and DRP concentrations on the open drainage channels with the water quality standards for rivers act as a guide to show if a water sample is high or low.

The annual mean DRP concentrations in the open drainage channels, which ranged from 0.09 mg L^{-1} in moderate-risk class to 0.40 mg L^{-1} in high-risk class (Figure 3), were higher than the surface water standard of 0.035 mg L^{-1}. The annual mean NO_3-N concentrations on the open drainage channels were lower across the risk classes, with ranges of 0.59 mg L^{-1} in low-risk class to 1.18 mg L^{-1} in moderate-risk class (Figure 3) relative to the standard of 1.8 mg NO_3-N L^{-1}. This is consistent with the poorly draining conceptual model of the EPA in Ireland, as P losses dominate nutrients relative to N losses. While this may be beyond the

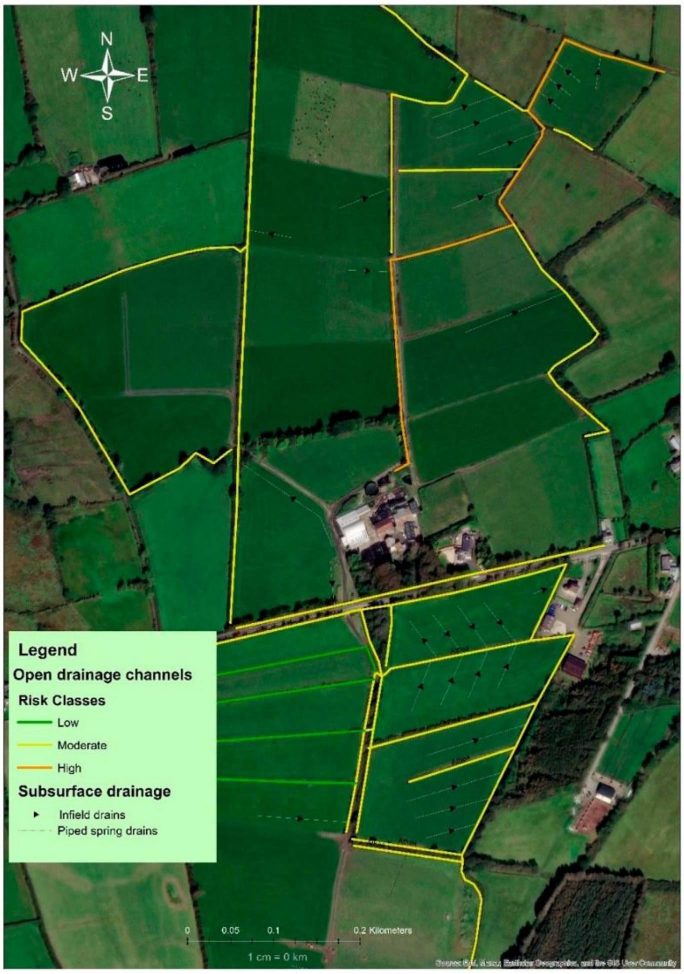

FIGURE 1
A map of a heavy textured grassland dairy farm (Farm #1 from Table 2) showing the risk classes of the open drainage channel network.

scope of the present study, 32% of sampling locations had high NO_3-N concentrations, indicating the N connectivity pathways that may be introducing NO_3-N into these open drainage channels (Opoku et al., 2024). Average P and N concentrations per risk class increased as the risk of the open drainage channels increased, except for average P concentrations for the moderate-risk class (Figure 3). This could be due to the anthropogenic and natural characteristics that create hydrochemical variation in the farm landscapes that contribute nutrients to the open drainage channels. With this caveat, this showed that the water quality seasonal grab samples validated the total risk score.

3.3 Assessment of the nutrient transfer continuum elements on the open drainage channels

The contribution of the source to the average total risk score of open drainage channels per farm ranged from 44.2% (Farm 2) to 63.5% (Farm 5) (Figure 4). Similarly, the contribution of the source to the total risk score of each of the high-risk open drainage channels ranged from 40.3%–70.2% (Figure 5).

The high proportion for source total risk score indicates that the multiple sources of nutrients, either from connection to farmyard,

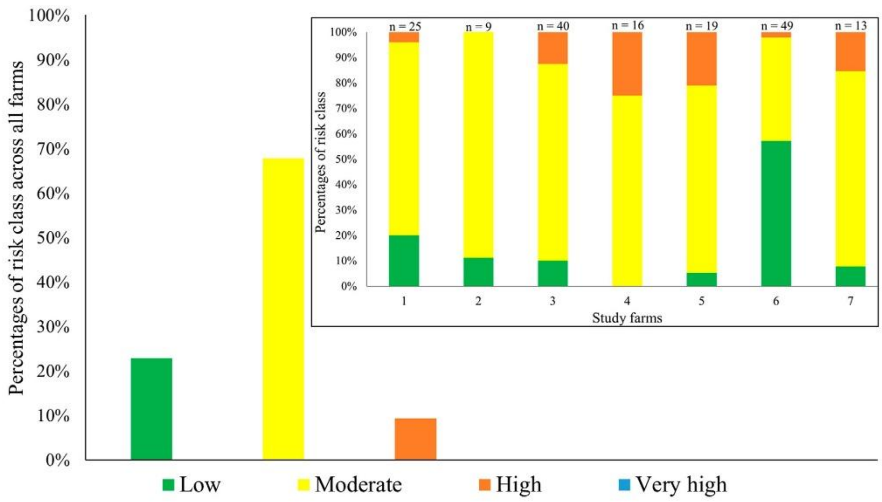

FIGURE 2
Percentages of risk classes for open drainage channels across all farms and within farms (inset).

TABLE 4 Number of high-risk channels (indicated by a 'X') by open drainage channel category.

Farm #	Natural open drainage channel	Farmyard connection ditch	Outlet ditch	Outflow ditch	Secondary ditch	Disconnected ditch
1		X				
2						
3		X	XXX		X	
4	X	XX	X			
5		XXX				
6		X				
7		XX				

legacy soil P, fertiliser application, and grazing input parameters, primarily influenced the risk of nutrient losses (Cassidy et al., 2017; Moloney et al., 2020) to these open drainage channels. While most (62.5%) of the high-risk open drainage channels connect farmyards, the high-risk open drainage channel with the highest source contribution to a total risk score recorded (70.2%) (i.e., secondary ditch on Figure 5) had no farmyard connection. This could be attributed to the open drainage channel's connectivity with high soil P-status fields, which received high fertiliser application for the duration of this study. This, together with surface and subsurface sources, may have led to the high total risk score on the other 37.5% of the whole high-risk open drainage channels with no connection to farmyards.

Along a connected pathway to the open drainage channel, the mobilisation of nutrients from the source was integral in most of the open drainage channels. The percentage of mobilisation contribution to the average total risk score of the open drainage channels per farm ranged from 10.2% to 31.5% (Figure 4). Rainfall is the primary factor by which mobilisation occurs for nutrient losses (Wang et al., 2020). Rainfall characteristics, including the intensity, duration and frequency, may influence the hydrological conditions that are critical to the surface and subsurface nutrient movement (Pérez-Gutiérrez et al., 2020). This necessitates the need to break the pathway to prevent the mobilised nutrient from the source to the receptor.

Nutrients enter the open drainage channels through multiple (surface, shallow subsurface and groundwater) pathways. The pathway contribution to the average total risk score per farm ranged from 10.5% to 18.4% (Figure 4). Heavy textured farms have multiple subsurface and surface connectivity pathways through which nutrients are lost (Clagnan et al., 2019; Granger et al., 2010; Opoku et al., 2024), and these may have contributed to

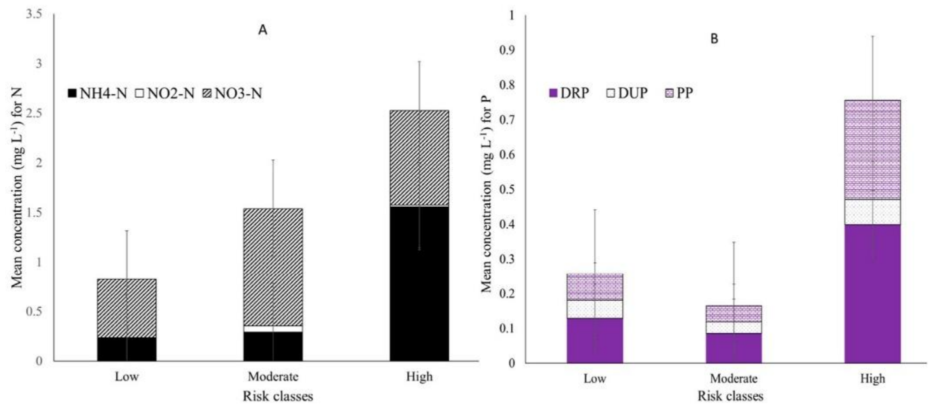

FIGURE 3
(A) Nitrogen and **(B)** phosphorus mean plus standard error concentrations from seasonal water sampling from within open drainage channels as per the risk classes across the case study farms.

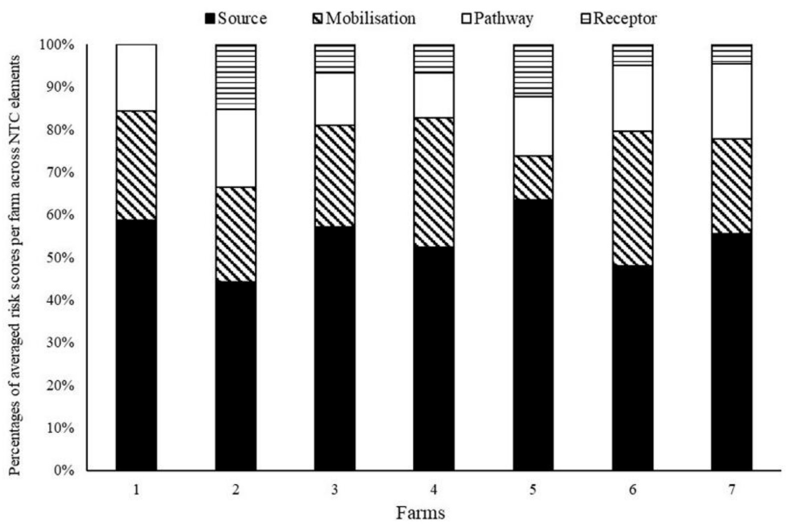

FIGURE 4
Percentages of averaged risk scores per farm across nutrient transfer continuum elements.

the high-risk open drainage channels. Eighteen-point-six percent and 18.6% of the high-risk open drainage channels received risk scores from roadway and farmyard runoff surface connectivity pathways to the total risk score, respectively, while 87.5% and 31.3% of the high-risk open drainage channels received risk scores from in-field drains and groundwater subsurface connectivity pathways, respectively. Although the pathway percentage contribution to the total risk score of the high-risk open drainage channels ranged from 10.1%–22.6%, the highest pathway contribution to total risk score for an open drainage channel was 44.9% which was a moderate-risk open drainage channel on Farm 6.

The connection to the receptor was not present on all high-risk open drainage channels. However, contributions from 14.5% to 18.6% of the total risk score of high-risk open drainage channels with connection to receptor for the study farms (Figure 5). This

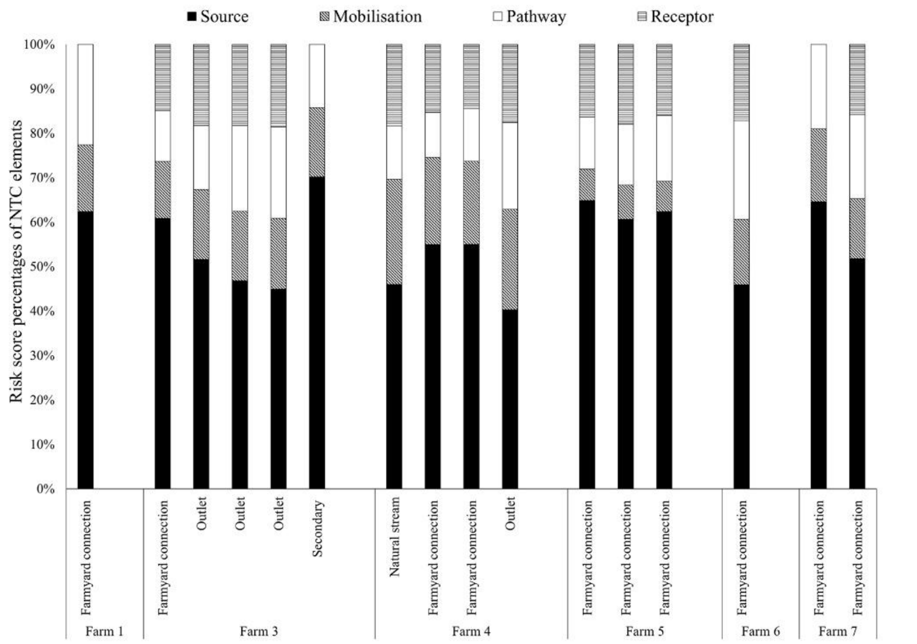

FIGURE 5
Risk score percentages of nutrient transfer continuum elements for farms with high-risk open drainage channels, excluding Farm 2 which had no high-risk open drainage channels.

informs the importance of considering the delivery of the final nutrient loss through the open drainage channels and may inform the mitigation type.

3.4 Mitigation of the high-risk open drainage channels

In Ireland, the EU Nitrates Directive is implemented through the Nitrates Action Plan (NAP), which applies to all farms in the country. This programme of measures outlines best farming practices to achieve good water quality outcomes for different farm enterprises. The EPA in Ireland identifies "breaking the pathway" on poorly draining soils, such as those in the present study, as an effective way to break the connectivity of surface or near-surface runoff between sources and waters. Opoku et al. (2024) classified the open drainage channel network into different ditch categories. Building on this work, the present study identifies open drainage channel sections within these large networks to be of higher risk and which may need mitigation. A combination of targeted measures is therefore necessary to improve water quality. This may include (1) source management (2) breaking the pathway (stopping runoff or near-runoff being delivered to waters), and (3) installation of in-channel filters (to slow the flow and attenuate a proportion of nutrients in dissolved and particulate forms from discharging through that open drainage channel section). On poorly draining soils this combined treatment train (Bourke et al., 2022) may prevent high nutrient-content water discharging from high-risk open drainage channel sections to the broader aquatic environment. Scrutiny of individual high-risk total risk score for different open drainage channel sections enables an advisor and farmer to identify specific sources, pathways, and in-channel actions as required. These may differ due to site-specific factors and cannot therefore be generic. Farmers are more inclined to accept less costly measures (van den Berg et al., 2023), and therefore these should be considered during the selection of mitigation measures (McDowell et al., 2024; King et al., 2015).

Opoku et al. (2024) and Fenton et al. (2021) detailed potential mitigation measures and costs available in terms of "break the pathway" mitigation options and costs. A few examples include: re-directing runoff away from internal roadways and the farmyard to collection or buffer areas with low-cost diversion bars or water bars (Fenton et al., 2021); installation of riparian (spatially targeted and linear) buffers along natural streams (Stutter et al., 2021) to control nutrient losses from the upslope field and connected internal farm roadways (Palmer, 2012; Yuan et al., 2009); targeted engineered mitigation measures including low-grade weirs (Faust et al., 2018), bunded drains, filter cells (Teagasc, 2022); and management of in-channel sediments through maintenance or characterisation of soil/sub-soil layer chemistry (Shore et al., 2015), which is both a sink and source of nutrients (Daly et al., 2017).

4 Conclusion

Assessments of nutrient loss from open drainage channels on poorly draining (heavy textured) soils are largely associated with predictions of surface runoff from critical hotspots. The risk assessment developed in this study combines potential water quality impacts from surface, subsurface, and groundwater characteristics of connecting fields to produce a colour-coded model of different potential water quality risk levels by which open drainage channels can be risk assessed. This risk assessment enables the production of risk maps that identify potential high- or very-high risk open drainage channels on dairy farms with heavy textured soils and assesses the nutrient transfer continuum elements to inform mitigation. Unlike previous open drainage channel risk assessment studies of Moloney et al. (2020) and Opoku et al. (2024), this study critically assesses all the source-mobilisation-pathway-receptor multi-parameters of the open drainage channel nutrient transfer continuum framework, provides in-depth information regarding high-risk open drainage channels to elucidate which parameters require attention during mitigation. The findings of this study apply to dairy farms on heavy textured soils in high rainfall areas, and may (or may not) differ in other geographic areas with different soils, climates and agricultural practices. However, it should be noted that the same methodology can be applied anywhere to develop a semi-quantitative risk assessment that will inform mitigation management. Future work incorporating varying risks encountered over time across wider farm characteristics will improve the risk scoring system to produce a more robust model that can be applied more generally on farms.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

DO: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Validation, Visualization, Writing–original draft, Writing–review and editing. MH: Conceptualization, Funding acquisition, Investigation, Methodology, Supervision, Validation, Visualization, Writing–review and editing. OF: Conceptualization, Funding acquisition, Investigation, Methodology, Supervision, Visualization, Writing–review and editing. KD: Conceptualization, Investigation, Methodology, Validation, Visualization, Writing–review and editing. TC: Funding acquisition, Methodology, Project administration, Writing–review and editing. PT: Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Visualization, Writing–review and editing.

Funding

The author(s) declare that financial support was received for the research, authorship, and/or publication of this article. The authors are grateful to Teagasc for the award of a Walsh Scholarship to the first author [grant number: RMIS-1381] to conduct this research.

Acknowledgments

The authors are grateful to Teagasc for the award of a Walsh Scholarship to the first author [grant number: RMIS-1381] to conduct this research, and to Simon Leach, Asaf Shnel and Denis Brennan for GIS, fieldwork and laboratory assistance provided.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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A3: Agricultural Water Management

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Article associated with Chapter 5.

Agricultural Water Management 322 (2025) 110007

Contents lists available at ScienceDirect

Agricultural Water Management

journal homepage: www.elsevier.com/locate/agwat

Examination of nutrient and sediment loss mitigation for farm roadway runoff on an Irish dairy farm

D.G. Opoku [a,b], M.G. Healy [b], O. Fenton [c], P. Tuohy [a,*]

[a] Teagasc, Animal & Grassland Research and Innovation Centre, Moorepark Fermoy, Co., Cork, Ireland
[b] Civil Engineering and Ryan Institute, College of Science and Engineering, University of Galway, Co, Galway, Ireland
[c] Teagasc, Crops, Environment and Land Use Centre, Johnstown Castle, Wexford, Co., Wexford, Ireland

ARTICLE INFO

Handling Editor - Dr R Thompson

Keywords:
Open drainage channels
sediment ponds
Water quality
Treatment train
nutrient removal efficiency
sediment retention efficiency

ABSTRACT

In Ireland, farm roadway runoff is a potential farm-scale pollution contributor of nutrients and sediments to connecting open drainage channels that pose a challenge to meeting European Union Water Framework Directive goals. To date, recommended mitigation measures such as swales, sediment ponds, bunded drains, willow beds, among others, have not been widely tested for efficiency and therefore are limiting farmers' willingness to implement them. This study quantifies the efficiency of bespoke sediment ponds at three locations (in a treatment train with diversion bars and riparian buffers) as farm roadway runoff mitigation measures by identifying runoff connectivity to open drainage channels, co-developing and co-implementing with farmers and monitoring for efficiency in nutrient and sediment removal. The study results suggest sediment ponds are efficient for removing sediment, total suspended solids and particulate nutrients, but vary in their effectiveness in removing dissolved nutrients due to biogeochemical and hydrological processes. The study concludes that sediment ponds are efficient for reducing roadway runoff pollution to open drainage channels but need to be designed to incorporate segmentation, consider all site conditions and encourage vegetation growth for enhanced nutrient and sediment removal, which may facilitate uptake among farmers. Long-term monitoring would be required to inform maintenance procedures and scheduling.

1. Introduction

Agricultural pollution from nutrient and sediment losses remains a concern for water quality degradation globally (McDowell et al., 2020; Shortle and Horan, 2017). In the European Union (EU), pollution from agriculture contributes to 22 % of surface water and 28 % of groundwater pollution (European Environment Agency (EEA) (EEA), 2021). To alleviate this environmental concern, multiple international, regional and local policies and regulations for managing agricultural pollution have been developed and continue to be adapted for practical implementation. In 2000, the EU developed the Water Framework Directive (WFD) (2000/60/EC; Official Journal of the European Community OJEC, 2000) for member states to adopt an integrated approach for managing waterbodies to reduce pollution and improve water quality to a "good status" by set deadlines. As part of the WFD integrated approach on water quality management, the Nitrates Directive (91/676/EEC) targets reducing agricultural pollution to waterbodies through good agricultural practices (Official Journal of the European Community OJEC, 1991) and requires EU member states to develop a Nitrates Action Programme (NAP) in reaching this goal.

In Ireland, programmes of measures to fulfil the goals of the WFD are set out and revised within the NAP (Department of Housing, Local Government and Heritage DHLGH, 2021a) to minimise both diffuse and point agricultural pollution potential. The NAP measures include, but are not limited to, limits on farm stocking rates and nutrient application rates, prohibitions on organic and chemical fertiliser application at environmentally-sensitive periods, minimum storage capacity for livestock manures and minimum set-back distances from waters (Department of Housing, Local Government and Heritage DHLGH, 2021b).

List of abbreviations: CSAs, Critical Source Areas; DON, Dissolved Organic Nitrogen; DRP, Dissolved Reactive Phosphorus; DUP, Dissolved Unreactive Phosphorus; N, Nitrogen; NAP, Nitrates Action Programme; NH_4-N, Ammonium as Nitrogen; NO_3-N, Nitrate as Nitrogen; P, Phosphorus; PON, Particulate Organic Nitrogen; PP, Particulate Phosphorus; TN, Total Nitrogen; TON, Total Oxidized Nitrogen; TP, Total phosphorus; WSP, Water-soluble P.

* Corresponding author.
E-mail address: Patrick.Tuohy@teagasc.ie (P. Tuohy).

https://doi.org/10.1016/j.agwat.2025.110007
Received 9 September 2025; Received in revised form 7 November 2025; Accepted 18 November 2025
Available online 27 November 2025

Recent iterations of the Nitrates Directive (S.I. No. 605 of 2017) acknowledge the risk of pollution from farm roadway runoff into connected open drainage channels and stipulate that "there shall be no direct runoff of soiled water from farm roadways to waters from 1 January 2021", alongside mitigation guidance options under the NAP to manage farm-scale agricultural pollution. Recent research on roadway runoff shows nutrient losses occurring both on open and closed periods on grassland farms (Fenton et al., 2024b; Sifundza et al., 2024). It has been found that 8.4 % (Rice et al., 2022) to 11.6 % (Maher et al., 2023) of roadway sections are connected to open drainage channels, while farm roadway and open drainage channel densities are highest on heavy textured soils. During rainfall events, nutrients within and on farm roadway sections connected to open drainage channels form critical source areas (CSAs) (Opoku et al., 2024a, 2024b) and are a sub-component of the nutrient transfer continuum (Fenton et al., 2022). After identification of CSAs, breaking the pathway before delivery of nutrient-rich roadway runoff to open channels is advised on farms (Fenton et al., 2021; Lucci et al., 2010; Opoku et al., 2024b).

Mitigation measures exist in broad terms, but bespoke solutions are needed for specific runoff problems. Primarily, approaches for preventing roadway runoff connectivity focus on breaking the pathway with on-roadway flow diversion structures and retention mitigation systems to reduce the transfer of nutrient and sediment losses to open drainage channels (Fenton et al., 2021; Tanner et al., 2023). The NAP recommends multiple mitigation measures and highlights a "right measure, right place" approach in their use to address diffuse pollutant sources, including farm roadways (Department of Housing, Local Government and Heritage DHLGH, 2024). However, the implementation of these recommended mitigation measures has generally only occurred on European Innovation Partnership (EIP) participant farms with no efficiency testing to guide future iterations and improvements of the mitigation measures. This limits knowledge of the efficiency of these mitigation measures, especially as they have tailored designs.

The efficiency of mitigation measures likely varies depending on the geo-positioning and design of the measure (Tanner et al., 2020; Thomas et al., 2016) and on the CSA characteristics (Tanner et al., 2020) such as farm management (e.g. grazing, stocking rate), rainfall, landscape characteristics (e.g. slope, soil) and contributing roadway area (e.g. size, composition, length and slope). These factors influence the impact on the hydrological and biogeochemical processes that determine the efficiency of mitigation measures (Persson and Wittgren, 2003). Furthermore, their efficiency may be influenced by available farmland sizes, which is often a constraint due to farmers' inability to release farm areas (Lastra-Bravo et al., 2015; Wilcock et al., 2012) for environmental measures. Ryan et al. (2025) observed that farmers are inclined to undertake evidence-based measures and those that require high-level knowledge or understanding for effective implementation. Assessing the efficiency of NAP-recommended mitigation measures in breaking the pathway and slowing farm roadway runoff to reduce agricultural nutrient and sediment transfer to connected open drainage channels will provide a thorough understanding of the context under which these mitigation measures may be effective. Such an understanding of their efficiency will improve knowledge of the mitigation measures in managing farm roadway runoff on Irish farms.

This study selects an Irish dairy farm with a high density of farm roadways and open drainage channels. The study aims to (1) use existing tools to examine and identify farm roadway CSAs where connectivity runoff enters open drainage channels, (2) co-develop and implement bespoke mitigation measures for these identified locations with the farmer, and (3) monitor the efficiency of the implemented mitigation measures at these locations under practical conditions.

2. Materials and methods

2.1. Study site

A dairy farm (45 ha) situated in the south-west of Ireland was selected (Fig. 1) following a previous semi-quantitative risk assessment on open drainage channels where farm roadway runoff connectivity was a prevalent issue and locations with a high risk of roadway runoff were identified (Opoku et al., 2024b). The location has a 10-year average annual rainfall of 1541 mm. The annual agronomic soil testing for phosphorus (P) using Morgan's reagent (Peech and English, 1944) carried out on the fields showed that 10.3 % had high soil P index 4 (> 8.0 mg L^{-1} P). The site has undulating topography with steep slopes (4 – 8 °) and soils classified as "heavy-textured". The soils vary from mineral to humic (Fealy et al., 2009), and are mainly moderately drained (55 %) or poorly drained (45 %), with 13.6 % of the fields having in-field drains installed. The nature of the soils and the topography enable overland flow and potential runoff from CSAs of sediment, nitrogen (N) and P into open drainage channels. The fields in the central parts of the farm have mostly moderately draining soils and therefore have a potential for infiltration (leaching) of nutrients, which complicates the task of isolating pollutant loss pathways on the farm.

2.2. Identifying runoff connectivity and critical source areas

Using the semi-quantitative risk assessment of Opoku et al. (2024b), three locations on the farm (A, B and C) with "high risk" open drainage channels with roadway runoff connectivity as a major contributor were identified (Fig. 1). These locations were cross-checked with the national EPA nutrient loss pathway map (https://gis.epa.ie/EPAMaps/Water) for risky pathways and delivery points to identify roadway runoff CSAs with a high likelihood of nutrient and sediment loss. The identified points of surface runoff delivery to open drainage channels from the national EPA nutrient loss pathway map were noted for further assessment. Following this, a ground survey assessment and visual assessment (Fenton et al., 2024a) was conducted during hydrologically-active periods to fine-tune these farm roadway CSAs and to identify the optimum locations for mitigation measures. At all locations, specific criteria of direct connectivity points where roadway runoff enters open drainage channels, the impact of the connectivity and the associated visual indicators, were used in the ground survey assessment. Direct roadway runoff entry into open drainage channels, with impacts of potential nutrient and sediment delivery, were observed at all locations. Open drainage channels routinely run through small culverts beneath farm roadways. Due to topography, these culverts lay in valley sections of the roadway, and they influenced roadway runoff behaviour. The delivery points were determined by local road configuration at these points. Other visual indicators including the disturbed roadway surface created by farm machinery tracks and livestock movements indicating freshly deposited and legacy P bound sediments, and the formation of runoff rills on the sections of farm roadways indicating overland flow, were also identified at all locations.

2.3. Co-developing and co-implementation of mitigation measures

Several farm visits were undertaken to determine possible mitigation solutions for the three identified farm roadway CSAs in consultation with the farmer. For all three locations, a treatment train mitigation measure of diversion-sediment pond-vegetated riparian buffer was proposed because it combines multiple measures with diverse functions to compensate for the limitations of each measure (Nicholson et al., 2012; Quinn et al., 2007). The diversion bar/cambered roadway diverts runoff to the sediment pond for primary treatment (sedimentation) and subsequently to the riparian buffer for secondary treatment (removal of dissolved pollutants). At Locations A and B, on-roadway concrete-based diversion bars extending 0.3 m beyond the edge of the roadway were

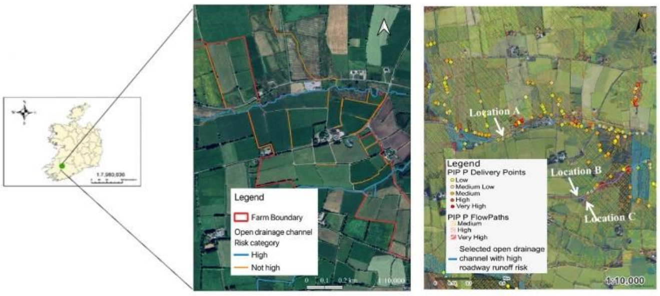

Fig. 1. Map of Ireland indicating farm location, "high risk" open drainage channels with roadway runoff connectivity (Opoku et al. 2024b), and the farm roadway CSA locations on the "high risk" open drainage channels (https://gis.epa.ie/EPAMaps/Water).

installed to direct runoff to the sediment ponds. For Location C, the farm roadway was resurfaced using gravel and cambered to divert roadway runoff towards the sediment ponds. A constant groundwater spring flow from an adjacent field through the cambered section into the sediment pond was observed at this location. The process of co-designing mitigation measures with landowners or advisors typically involves compromises associated with many factors that affect the final mitigation designs. These may for example, limit the size of the installed sediment pond from optimal e.g., this decision could be based on land availability at the delivery point or an unwillingness of the landowner to use that land.

The optimal sediment pond volume was calculated based on the hydraulic loading rate of the site to ensure optimal pollutant removal through sediment retention (Smith and Muirhead, 2023; Robotham et al., 2021). For each location, the sediment pond volume, V (m^3), was calculated using:

$$V = R \times T \tag{1}$$

where R is the peak discharge rate (m^3s^{-1}), and T is the residence time (s). The peak discharge rate, R, in Eq. 1 was calculated using (Barber, 2013):

$$R = C \times A \times I \tag{2}$$

where C is a dimensionless runoff coefficient dependent on hydrological factors (the soil type, land use, degree of imperviousness, slope, surface roughness, antecedent moisture condition, duration and intensity of rainfall, recurrence interval of rainfall, interception and surface storage variables); A is the contributing farm roadway area (m^2), and I is the average intensity of rainfall ($m\ s^{-1}$). A value of 0.5 was assigned to C, which was estimated for forest roadways (Jordán and Martínez-Zavala, 2008) of similar gravel and unpaved characteristics. Using local meteorological records, rainfall intensity, I, for a 6-hour duration, 1-in-5-year return period, storm event was used – 5.67 mm hr^{-1} (1.57×10^{-6} m s^{-1}). Contributing farm roadway areas of 429.3 m^2 over an 8.4 ° slope (Location A), 106.8 m^2 over a 6.7 ° slope (Location B) and 249.5 m^2 over a 7.3 ° slope (Location C) were used.

The residence time, T, in Eq. 1 was calculated using:

$$T = s/Vs \tag{3}$$

where s (m) is the travel distance set for sediments to fully settle in the sediment pond (using s at 1 m) and Vs (ms^{-1}) is the velocity of sediment settling for clay sediment, calculated using Stokes' Law:

$$Vs = \frac{d^2 g(Dp - Df)}{18\mu} \tag{4}$$

where, d is the diameter of the particle (3.9×10^{-6} m for clay; Barber, 2013), g is gravity (9.8 m s^{-2}), Dp is the density of clay particles (2860 kg m^{-3}; Schjønning et al., 2017), Df is the density of the fluid (1000 kg m^{-3}), and μ is the dynamic viscosity of the fluid (0.001 kg (m $s)^{-1}$).

Based on these hydrological flow estimations, the volumes (V) at depth (s) = 1 m required for the sediment ponds were calculated as 28.6 m^3, 7.1 m^3 and 16.6 m^3 for Locations A, B and C, respectively. While these estimated sediment pond sizes may allow optimum effectiveness, site constraints including high water table of the adjacent open drainage channels and limited land area, especially at Location A, necessitated resizing of the sediment pond sizes to ~ 4 m^3, ~ 7 m^3 and ~17 m^3 at sediment settling depths (s) of 0.5 m, 0.5 m and 1 m for Locations A, B and C respectively (Fig. 2). These constraints led to an undersized sediment pond volume at Location A, while Locations B and C remained with their optimum sediment pond volumes. Following Barber (2013), the sediment ponds were configured into pond cells to enhance removal efficiency while adapting to the site conditions. For the sediment pond configurations, two sediment pond cells, each measuring 2.5 × 1.5 × 0.5 m (L x W x D) at Location A, one sediment pond cell measuring 4.0 × 3.5 × 0.5 m at Location B and two sediment pond cells, each measuring 4.25 × 2.0 × 1.0 m at Location C, were excavated (Fig. 2). The sediment ponds were manually levelled after digger excavation and crosschecked with a spirit level. This allowed accurate measurement of the accumulated sediment volume.

To prevent pond bank erosion, sediment ponds were excavated to create banked sides for stability (Barber, 2013). The sediment ponds were lined with heavy-duty woven weed mats up to the banks, which enabled estimation of accumulated sediment volume. The heavy-duty weed mats used had 0.63 porosity, calculated as 1 - (bulk density of the woven weed mat material/bulk density of a polypropylene solid) (Kalazić et al., 2023), using manufacturer's product specifications of mass per area of 100 g/m^2 and thickness of 0.3 mm which yielded a bulk density of 333 kg/m^3. The bulk density of polypropylene solid was taken as 900 kg/m^3 (Jones et al., 2025). The weed mat was double-lined to reduce porosity. It was assumed that the base weed mat layer in contact with the wet, fine (clayey) soil particles of these heavy textured soils under the pond water pressure clog over time, further reducing porosity and infiltration potential. The weed mats were firmly attached to the sides and base of the ponds with wires and ground cover pegs. Edges (excluding exit and entry) along the sediment pond cell(s) were bunded and grassed to prevent overland flow from adjacent areas during rainfall events. At the exit of every sediment pond, a 1 m-long, 0.10 m-diameter plastic pipe was connected to the next sediment pond or discharged into the 3 m-wide riparian buffer.

The riparian buffer was installed at the end of sediment pond configuration at each location before the adjacent open drainage channel to meet the current recommendation of at least a 3 m-wide vegetated riparian buffer to prevent direct soiled runoff into waterbodies under the 2022 NAP 5 in the EPA Research Report No. 485 (Ó

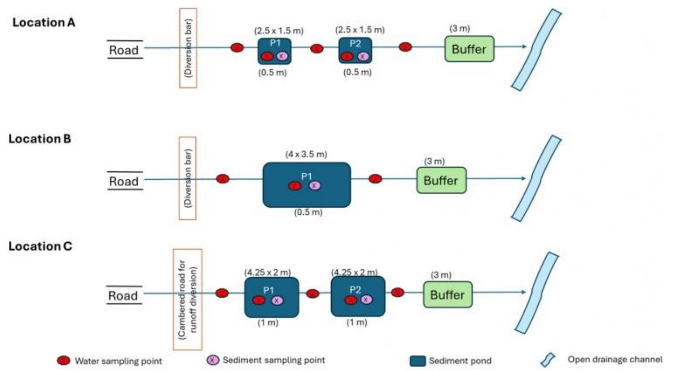

Fig. 2. Schematic representation of mitigation measures and dimensions.

Huallacháin et al., 2023). While such an additional measure is expected to further reduce the sediment and pollutant concentration in the runoff from the sediment ponds, the nature of vegetated riparian buffers does not allow direct measurement of downstream water quality at all locations, and therefore measurement at these locations was not undertaken. This study, therefore, only evaluates the efficiency of the sediment ponds on the farm.

2.4. Water and sediment sampling for testing mitigation measures

2.4.1. Water and sediment sampling

Water and sediment samples were taken during the hydrologically-active periods between the week of 22nd October 2024–19th March 2025, except for 3 weeks from late December 2024 to early January 2025 when the site was not accessible due to heavy snowfall. Using sampling points in Fig. 2, two 50 ml paired (filtered and unfiltered) water samples were taken weekly from all water sampling points in all locations for N and P fractions analysis. In addition, 500 ml water samples were taken weekly at these water sampling points in all locations for total suspended solids (TSS) concentration measurement. Inlet water samples were collected from diverted roadway runoff flows at the entry points for each location. All the 50 ml (filtered and unfiltered) and the 500 ml water samples were stored and transported in cool boxes to the laboratory for water analysis and TSS within 24 h of sample collection.

To measure accumulated sediment volume in each pond, 1 m graduated staffs were used to measure the depth of accumulated sediment over the manually levelled pond area over the study duration (Cooper et al., 2019). In each pond, two 1 m graduated staffs were firmly fixed to the pond base along a transect, one near the inlet and the other near the outlet, to measure accumulated sediment depth. The average of the two measured depths was multiplied with the pond's cross-sectional area to estimate accumulated sediment volume. Reference markings on the pond banks were made during installation to assess and correct any positional shifts. The average depth readings of accumulated sediment from both graduated staffs within each pond were calculated every 4 weeks. For a particular pond cell, the calculated average depth and pond area were multiplied to estimate the accumulated volume for that 4-week period. After each 4-week measurement of accumulated sediment, ~0.5 kg of fresh (wet) sediment samples were collected from the base of each pond cell. The sediment samples were transported in cool ice boxes to the laboratory and then analysed for water-soluble P (WSP) to ascertain the sediment P composition.

2.5. Laboratory analysis

The unfiltered 50 ml grab water samples were analysed calorimetrically for nitrite (NO_2-N), ammonium (NH_4-N), total oxidized nitrogen (TON), and total reactive phosphorus (TRP) using a Thermo Fisher Scientific Gallery ™ Discrete Analyzer. The unfiltered samples were analysed for total phosphorus (TP) and total nitrogen (TN) was analysed using the Hach Ganimede P analyser and the Hach Ganimede N analyser, respectively. The filtered 50 ml grab water samples were analysed for dissolved reactive phosphorus (DRP) and total dissolved phosphorus (TDP) using a Thermo Fisher Scientific Gallery ™ Discrete Analyzer and a Hach Ganimede P analyser, respectively. All water samples, reagent blanks and check standards were analysed following the Standard Methods (American Public Health Association (APHA) (APHA), 2005). All quality control (QC) samples/check standards were prepared from certified stock standards from a different source than calibration standards. Quality control samples were analysed at the beginning and end of every sample batch, for every 10 samples within a batch, and if the QC fell outside limits, samples were repeated to the last correct QC. Blanks were included in every sample batch for analysis, and approximately 10 % of samples were repeated. Tolerances ranged up to a maximum of ±7.5 % of the nominal value. All instruments used were calibrated in line with the manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, particulate phosphorus (PP) was calculated by the difference between TP and TDP, and dissolved unreactive phosphorus (DUP) was calculated by the difference between TDP and DRP. Total suspended sediment concentrations were measured using the standard gravimetric method (American Public Health Association (APHA) (APHA), 2005).

For WSP analysis, portions of the sediment samples for each pond cell were prepared by air-drying and sieving through 2 mm, and 1 g of the prepared sediments were moistened with 2 ml of deionised water and allowed to stand for 22 h. These were further moistened with 70 ml of deionised water, equilibrated for 1 h on a reciprocating shaker (van der Paauw, 1971) and filtered using Whatman No. 4 filter paper before the filtrate was quantified calorimetrically for P. Using the sediment mass (g) and total volume of deionised water (ml; converted to L) used for moistening the sediment, P concentration (mg L^{-1}) in the filtrate was converted to mg/g.

2.6. Data analysis

Microsoft Excel software version 16.0 (2016) was used for data computing and preparation prior to statistical analysis, and R Studio version 4.3.2 (2023) was used for statistical procedures. To assess the efficiency of the sediment ponds deployed at the various locations, the water sampling results for the N and P fractions, TSS, and physical and chemical sediment characteristics were compared. The removal efficiency was defined as the percentage removal calculated as the difference in water quality parameter concentrations at the inlet sampling point of the sediment pond and the outlet sampling point of the sediment pond (Eq. 5):

$$\text{Removal efficiency } (\%) = \frac{\text{Inlet water concentration} - \text{Outlet water concentration}}{\text{Inlet water concentration}} \text{ x } 100\ \% \tag{5}$$

All inlet and outlet water quality data were assessed for normality with the Shapiro-Wilk test and were not transformed. To test efficiency of sediment ponds statistically at individual locations, the inlet and outlet water quality data for each location were tested for statistically significant differences using the paired T-test for normally distributed water quality parameters (Barber, 2013; Robotham et al., 2021) and the Wilcoxon Signed-Rank (pairwise test) for non-normally distributed water quality parameters. All significant differences were observed at an alpha level of 0.05 (95 %) confidence level, and where alpha level was much lower, a 0.01 (99 %) confidence level was used. All water quality parameter values "<LOD" (below the Limit of Detection) or "not detectable" were treated as zero for analysis. Mean comparisons were undertaken for WSP, accumulated sediment volume and weather data (rainfall (precipitation) and temperature). Rainfall refers to the total precipitation, and as these heavy textured, poorly drained soils remained wet throughout the study period, precipitation/rainfall may be considered as very crucial for runoff. Concentrations of P and N fractions of the inlet and outlet of pond configuration systems for a location were examined as proportions of total P and N.

3. Results and discussion

3.1. Assessment for critical source area identification and co-designing of sediment ponds

Visual or ground assessment was guided by surface flow pathways and delivery points available via the EPA nutrient loss pathway maps, which provide details such as actual direction of runoff flow to open drain channels. Although the EPA nutrient loss pathway map use hydrological factors such as soil drainage and topological properties to show delivery points for where roadway runoff into open drainage channels may occur, the ground assessment that indicated that anthropogenic influence including design and characteristics of installed culvert also influenced roadway runoff behaviour. These influences determined flow direction towards to open drainage channels and contributed to temporary localised ponding of roadway runoff prior to discharge across all surveyed locations.

Aside from animal waste (urine and faeces) deposition being key pollutant sources in farm roadway runoff (Fenton et al., 2024b; Sifundza et al., 2024), the eroded rills on farm roadways and the localised ponding observed informed potential risk of nutrient-rich sediment from disturbed roadway surfaces noted as CSAs. This agrees with Maher et al. (2023), who reported that farm roadway characteristics influence farm animal movement, and this makes some sections of the farm roadway more vulnerable to surface disturbance and high waste depositions (Sifundza et al., 2024). The movement of farm machinery and animals is expected to disturb the roadway surfaces back into the localised ponding, potentially increasing pollutant discharge into open drainage channels at certain times. The identification of temporary localised ponding during the ground assessment influenced the decision to use diversion bars and in some cases road cambering (as in Location C) to divert flow and limit the temporal occurrence of roadway ponding.

Furthermore, the farmer co-development aided identification of unused land spaces on the farm for implementation of the sediment ponds. Utilising unused spaces has also been reported as a potential option for successful implementation of environmental management measures in Batáry et al. (2015) and Burland and von Cossel (2023). This increases farmers' commitment as they do not lose their productive farmlands, thus having minimal to no practical impacts. Lastly, the involvement of farmers in co-designing promoted cooperation which ensured on-farm safety as farm animals were prohibited from accessing sediment ponds and ensuring smooth operation of the installed sediment ponds.

3.2. Sediment trapping in sediment pond configurations

During the study, accumulated sediment volumes (m^3) within pond cells increased by 0.169 m^3 in pond cell 1 and 0.128 m^3 in pond cell 2 at Location A, 0.088 m^3 in pond cell 1 at Location B, and 0.077 m^3 in pond cell 1 and 0.038 m^3 in pond cell 2 at Location C. This indicates a 53–567 % sediment accumulation (relative to the initial sampling volumes) across locations during the monitoring period (Table 1). These findings show that sediment accumulation in ponds is related to contributing area, with larger areas yielding more accumulated sediment in ponds. Although precautions were taken to limit porosity of the weed mat and ensure accurate accumulated sediment volume estimations (Section 2.3) with the method adopted, very fine sediment (and associated P concentration) may be expected to travel through weed mat, leading biased underestimation in all sediment ponds. However, this methodology as opposed to previous sediment measurements in Barber and Robotham where circular plates (saucers) were placed at the base of pond to trap sediment was best suited for these sediment ponds (of full capacity for nearly all the sampling period) as drawing out these plates may lead to sediments washing off and causing random error as opposed to systematic error.

The sedimentation process is influenced by factors including pond size and flow reduction capacity, runoff flow velocity, sediment size characteristics. Sediment size influences sedimentation, allowing coarse sediment to settle more quickly and fine particles to remain suspended until flow slows (Clarke, 2013; Levine, 2020; Ockenden et al., 2012). Higher mean sediment accumulation in the first pond cells at Locations A and B (Table 1) suggests coarse sediment trapping, which occupies more volume. Conversely, lower mean sediment accumulation in their respective second pond cells (Table 1) suggests fine sediment trapping which, due to their smaller size and lower weight, travel far and occupy less volume. Visual observations, especially during and immediately after rainfall, revealed cloudier water in second pond cells, suggesting resuspension of lightweight fine sediments. This aligns with findings of multi-pond studies, which also observed that first pond cells trapped heavier and less mobile sediments than subsequent cells (Barber, 2013; Robotham et al., 2021). Fine sediments are major P carriers that contribute to P losses (Ballantine et al., 2006; Shore et al., 2015). This may have contributed to the higher WSP concentrations in sediments of the second pond cells relative to the first pond cells at Locations A and C (Fig. 3).

Rainfall, the primary driver for sediment mobilisation from farm roadways (Fenton et al., 2021; Rice et al., 2022), had positive

Table 1
Accumulated sediment volumes (m^3) and percentage increase from start to the end of monitoring.

Location (contributing area (m^2))	Pond	Volume of sediment accumulated at initial measurement (m^3)	Volume of sediment accumulated at final measurement (m^3)	Mean volume of sediment accumulated at each measurement (m^3)	Total sediment volume increase relative to initial volume (%)
A (429.3 m^2)	Cell 1	0.038	0.206	0.13	450.0
	Cell 2	0.023	0.150	0.10	566.7
B (106.8 m^2)	Cell 1	0.049	0.137	0.09	178.6
C (249.5 m^2)	Cell 1	0.068	0.145	0.11	112.5
	Cell 2	0.072	0.111	0.09	52.9

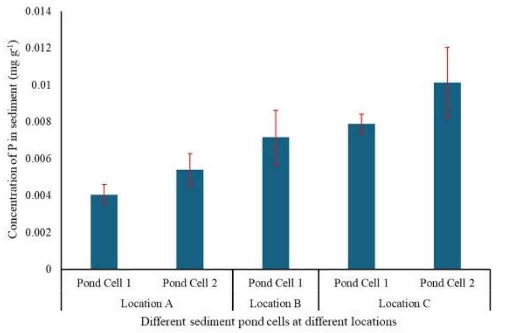

Fig. 3. Water-soluble P (WSP) of sediment samples at different locations and pond cells.

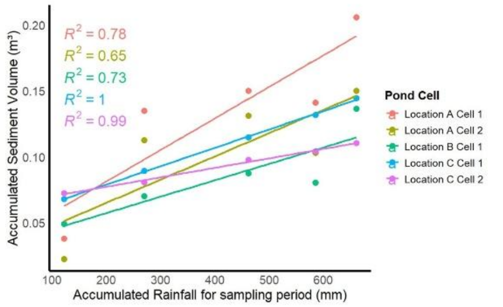

Fig. 4. Correlation between accumulated sediment volume (m^3) and accumulated rainfall (mm) for every sampling period.

correlations with sediment accumulation ($R^2 = 0.65 - 1$) (Fig. 4), suggesting that transport of accumulated sediments in ponds was dependent on rainfall. This correlation was even more pronounced in the first pond cells.

3.3. Nutrient and TSS removal in sediment ponds

3.3.1. Nitrogen removal efficiency in sediment ponds configurations

Over the sampling period, TN removal efficiencies in Location A and C (both two-cell configurations) were similar, at 30.9 ± 39.0 % and 27.4 ± 42.6 % removal respectively, while the one-cell pond configuration system at Location B recorded only 0.46 ± 13.8 % removal

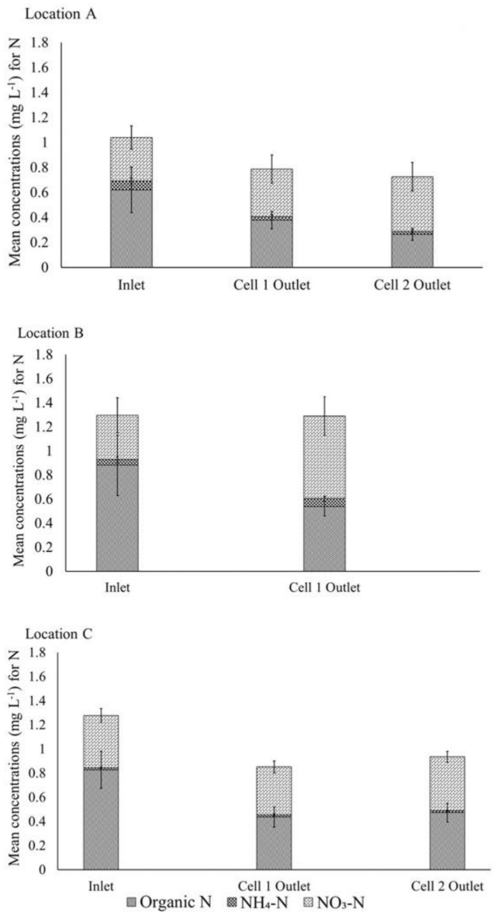

Fig. 5. Nitrogen (N) mean ± standard error concentrations from sediment pond cells at Locations A, B and C on study farm.

(Fig. 5). Nevertheless, the mean outlet TN concentrations for the sediment ponds at all locations (Table S1) were lower than the current N discharge limit of 10 mg L^{-1} under EU Urban Waste Water Treatment

Directive (UWWTD 91/271) (European Commission, 2024). The TN removal efficiencies in Locations A and C are consistent with the average TN removal efficiency of 31 % for wet ponds in Koch et al., 2014, but relatively lower than Mallin et al. (2012) reported results of 66 – 96 % TN reduction in a 4.7 ha multi-segmented constructed wetland designed for a 24-hour duration, 1-in-100-year return period storm event. Within runoff treatment systems, sedimentation and microbial transformations (mineralisation, nitrification and denitrification) and plant uptake are the primary N removal mechanisms (Kill et al., 2018; Vymazal et al., 1998), and these factors considerably influence variation in N removal efficiencies. The relatively low TN removal in this study could be due to low temperatures measured during the study period (6.73 ± 0.26 °C) which reduce the microbial transformations (Kill et al., 2018; Robotham et al., 2021), regular wet season runoff which limited hydraulic retention (Braskerud, 2002), and lack of vegetation in the lined study pond cells. The study site, characterised by a wet, heavy textured soils and a high (10-year average) annual rainfall of 1541 mm, is prone to frequent runoff due to prolonged soil saturation during the wet season and this limits TN removal efficiency. Kim et al. (2011) showed relatively lower N removal during the wet season due to influence from seasonal factors like temperature (Persson and Wittgren, 2003). Based on this assumption, the low temperatures in the wet season likely inhibited microbial activities, limiting N removal efficiency in this study. The wet season is a critical period for roadway runoff mobilisation under the Irish temperate conditions, and with the observed low N removal efficiency in this study, it is imperative to enhance sediment pond performance. Vegetation in sediment ponds enhanced TN removal efficiencies in previous studies (Beutel et al., 2009; Wang and Sample, 2014). While the sediment ponds assessed in this study had no vegetation growth as they were firmly lined with heavy-duty weed mats (which blocks vegetation growth) to allow for accurate measurement of accumulated sediment volumes, the introduction and assessment of sediment ponds with vegetation, under the cold and wet temperate Irish will inform options for optimal N removal.

Organic N concentrations decreased at all three locations, albeit only significantly at Locations A and C (Table S1). These positive organic N removal efficiencies agree with Mallin et al. (2012), who reported an average 70 % organic N removal efficiency through a treatment system. In segmented pond systems, the first pond cell slows flow velocity and retains particulate nutrient forms. In contrast, flow in the one-cell pond configuration at Location B lacks segmentation, potentially leading to short-circuiting (with potential direct flow out of ponds) and limiting organic N removal via sedimentation.

Organic N exists in dissolved (DON) and particulate (PON) forms, and removal mechanisms may vary depending on its forms. Removal mechanisms include sedimentation for PON and microbial mineralisation for DON, depending on the labile or refractory composition of organic N for microbial breakdown (Bronk et al., 2007; Mallin et al., 2012). Ponds are generally static systems, where nearly all nutrient transformations occur through exchange processes (Boyd, 1995). Higher NO_3-N concentrations recorded at exit pond cell outlets (Fig. 5) suggest that mineralisation of organic N to NH_4-N, followed by rapid nitrification to NO_3-N, may have occurred within the ponds. The statistically significant positive organic N removal efficiencies (Table S1) in the two-cell pond configuration may stem from enhanced PON sedimentation due to pond segmentation at Locations A and C.

Inorganic nitrogen removal efficiencies varied considerably across locations. At Locations B and C, NH_4-N concentrations increased, whereas it reduced in Location A (Table S1). The average influent NH_4-N concentrations were, however, very low, ranging from 0.02 to 0.072 mg L^{-1}. Similar NH_4-N removal inefficiencies of −61 ± 118 % were reported by Robotham et al. (2021) in a three small online-pond study. The positive mean NH_4-N removal efficiency in Location A (68.11 ± 66.00 %) may be due to shorter hydraulic residence time in these undersized pond cells which may have limited the ammonification of retained organic N in the first pond cell, leaving lower NH_4-N concentrations to travel to the second pond cell and then the outlet. On this assumption, where NH_4-N removal inefficiencies recorded in the optimal size ponds maybe due to ammonification of retained organic N.

Through nitrification, NH_4-N concentrations convert into NO_3-N concentrations (Vymazal et al., 1998), adding to the initial NO_3-N concentrations and increasing NO_3-N leaving the ponds. Although not statistically significant, all three locations had negative mean NO_3-N removal efficiencies (Table S1). Studies by Kim et al. (2011), Mallin et al. (2012) and Robotham et al. (2021) report contrasting results of positive mean reductions. Their results may have varied from this study primarily due to the low mean air temperature over the monitoring period of 6.7 ± 0.3 °C in which this study was conducted. The temperature may have inhibited microbial transformations (e.g. denitrification) for NO_3-N removal, compared to the reported mean removal efficiencies for all seasons. Incorporating vegetation within these pond cells to function as constructed wetlands (Tang et al., 2021) would improve NO_3-N removal via plant uptake and provide carbon for

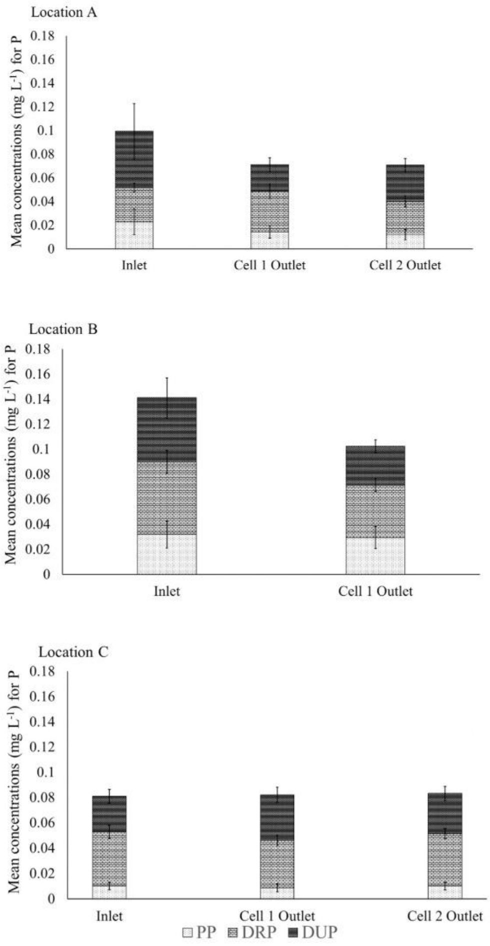

Fig. 6. Phosphorus (P) mean ± standard error concentrations from sediment pond cells at Locations A, B and C.

denitrification under anaerobic conditions.

3.3.2. Phosphorus and TSS removal in sediment ponds

The average influent TP concentration to the sediment ponds ranged from 0.08 mg L^{-1} (Location A) to 0.75 mg L^{-1} (Location C). Locations A and B had TP removal efficiencies of 17.0 ± 38.1 % and 11.7 ± 7.1 %, respectively (Fig. 6), whereas Location C had a TP removal efficiency of −10.4 ± 9.2 % (Table S1). Removal efficiencies of total phosphorus in previous studies show considerable variations, ranging from −34 % and 89 % in Robotham et al. (2021) and Mallin et al. (2012), respectively. This is primarily due to varying physical and biochemical processes. However, Yazdi et al. (2021) reported a 10 % TP removal efficiency during colder weather in a year-long sediment pond study which is comparable to the observed results at Locations A and B in this study which was also conducted under cold and wet seasons. The TP removal inefficiency at Location C aligns with Robotham et al. (2021) who reported TP removal inefficiency in a multi-pond study due to hydrological fluxes. In this study, the TP removal inefficiency at Location C is similarly attributed to hydrological flux from the continuous spring flow entering the sediment ponds at Location C. Notwithstanding the negative TP removal at Location C, mean outlet TP for all locations remained lower than the current P discharge limit of 0.7 mg L^{-1} under the EU Urban Waste Water Treatment Directive (UWWTD 91/271) (European Commission, 2024).

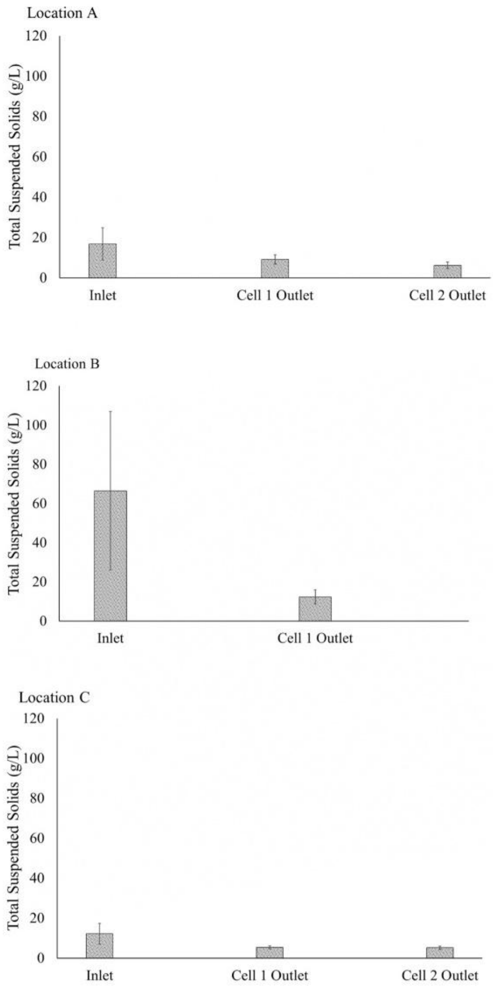

Fig. 7. Mean total suspended solids (TSS) ± standard error concentrations from sediment pond cells at Locations A, B and C.

All locations had positive mean PP removal efficiencies: 47.0 ± 60.3 % at Location A, 7.1 ± 17.4 % at Location B and 1.1 ± 4.4 % at Location C (Table S1). These reductions indicate effective PP removal by sedimentation, consistent with the observations of Shan et al. (2002). There was a similar trend for TSS (Fig. 7), with mean removal efficiencies of 63.0 ± 79.2 %, 81.5 ± 90.9 % and 57.9 ± 84.7 % at Locations A, B and C, respectively. This demonstrates suspended sediments' contributions to P concentrations (Cooper et al., 2015; Evans et al., 2004), and highlights sediment pond systems' role in trapping particulate pollutants (Gu et al., 2017; Mekonnen et al., 2017). Total dissolved P, comprising DUP and DRP, dominated P in the inlet, ranging from 77.5 % (Location B) to 94.1 % (Location C) of TP (Fig. 6). Locations A, B and C had positive mean DRP removal efficiencies of 3.9 ± 19.2 %, 27.9 ± 44.6 %, and 3.0 ± 21.9 %, respectively. The DRP reductions are consistent with the 14.9 % and 29 ± 37 % mean removal efficiencies in the pond treatment studies of Barber (2013) and Robotham et al. (2021), respectively. Adsorption is the principal removal mechanism for dissolved P (Lai and Che, 2008), and this is influenced by the availability of the adsorbing sites. The WSP, which indicates readily available P, of pond sediments at Location C was relatively higher than at Locations A and B (Fig. 3), indicating P-concentrated sediment. Concentrated P sediments have limited adsorption sites, and this may have potentially lowered adsorption, leading to low P removal at location C. Further research on equilibrium P concentration and sorption analysis on the sediment, however, may be required improve understanding on the adsorption.

The reduction trend of DRP was also observed for DUP at Locations A and B, but increased significantly at Location C ($p < 0.05$). The continuous hydraulic loading and base flow, driven by the connecting groundwater spring emerging through the cambered section of the reconstructed farm roadway into Location C's pond cells, may have impacted the P removal. Kill et al. (2018) attributed low nutrient removal in a runoff treatment system to constant groundwater flow seeping from adjacent fields. Such conditions create consistent flow currents that reduce residence time (Brown et al., 1981), cause sediment resuspension (Saeed et al., 2019) to release P into the water column (Sinke et al., 1990; Søndergaard et al., 2003) as organic P (DUP), and increase aeration for microbial desorption (Stahlberg et al., 2006; Yu et al., 2022). This finding reinforces the importance of matching pond design to actual local hydrological context and provides a novel idea, by including other characteristics associated with flow such as permanence and seasonal dynamics, where present, into the pond volume estimations. Through this research, farmers are aware of external factors that can influence sediment pond performance and are encouraged to provide relevant local hydrological knowledge to engineers or scientists to ensure optimal sediment pond designs for effective nutrient and sediment removal. This approach, where farmers provide vital local knowledge in co-designing, is expected to ensure successful implementation of sediment ponds on Irish farms. Previous studies of Amblard et al. (2023), Campling et al. (2021) and Richard et al. (2020) support this as an effective tool for integrating local farm-specific insights to improve design constraints and ensure successful implementation of agri-environmental measures for improving water quality.

4. Conclusion

This study showed that sediment ponds, implemented at appropriate locations for managing farm roadway runoff loss to open drainage channels, are effective in removing sediment, TSS and particulate nutrients, but vary in the removal of dissolved nutrients. The trapping of sediment in ponds is dependent on the contributing area as a sediment source and rainfall as a mobiliser, while nutrient removal is dependent on the pond design and site conditions. Policy recommendations delivered through farm advisory services to farmers on future iterations of sediment ponds design should promote pond segmentation, accounting for site-specific hydrological conditions such as constant hydrological loadings from groundwater springs (if present), and inclusion of vegetation to improve their hydrological and biogeochemical functioning for enhanced nutrient removal. With the provision of this high and practical knowledge on sediment pond effectiveness (including those with constrained pond design sizes), farmers will be more likely to use sediment ponds for managing farm roadway runoff entering open drainage channels. Long-term monitoring of at least one year to capture all seasonal runoff variations and further research on equilibrium P concentration of sediment are required to make estimations for maintenance measures such as pond dredging.

Funding

The authors are grateful to Teagasc for the award of a Walsh Scholarship to the first author [grant number: RMIS-1381] to conduct this research.

CRediT authorship contribution statement

D.G. Opoku: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Validation; Visualization; Roles/Writing – original draft; Writing – review & editing. **M.G. Healy:** Conceptualization; Funding acquisition; Investigation; Methodology; Supervision; Validation; Visualization; Writing – review & editing. **O. Fenton:** Conceptualization; Funding acquisition; Investigation; Methodology; Supervision; Validation; Visualization; Writing – review & editing. **P. Tuohy:** Conceptualization; Funding acquisition; Investigation; Methodology; Project administration; Resources; Supervision; Software; Validation; Visualization; Writing – review & editing.

Declaration of Competing Interest

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

Acknowledgments

The authors are grateful to Tomas Condon, Asaf Shnel and Wendy Izedonmwen for the fieldwork assistance and to Jessyca DeMedeiros for laboratory assistance provided.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agwat.2025.110007.

Data availability

Data will be made available on request.

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A4: International Symposium on Climate-Resilient Agri-Environmental Systems (ISCRAES) Conferences Proceedings.

Artificial drainage nutrient loss risk classification system for grassland farms to inform future mitigation management.

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T. & Tuohy, P.

Article associated with Chapter 3.

THE 2ND INTERNATIONAL SYMPOSIUM ON CLIMATE-RESILIENT AGRI-ENVIRONMENTAL SYSTEMS

ISCRAES 2022

28-31 August 2022
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Book of Abstracts

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ISSN No. 2737-727X

This book contains all abstracts presented at the conference and published in 2022 by the ISCRAES 2022 Organising Committee in collaboration with Prudence College Dublin

solids to determine the filter functionality of the envelope and destructive sampling of the envelope post experiment in order to determine the ingress of sediment into the envelope. Results showed that an aggregate of 0.7 – 3 mm performed best from a flow rate perspective. From a sediment loss perspective, the best performing aggregate was in the 2 – 10 mm range. Overall the results showed that an aggregate range from 2 – 10 mm is optimal for a clay textured soil. However, aggregate sizes up to 20 mm would be acceptable. The adoption of more appropriate material specifications will optimise performance and extend the lifetime of installed drainage systems in mineral soils.

Keywords: Drainage materials; Drain envelopes; Hydrology; Land use; Soil management.

Artificial drainage nutrient loss risk classification system for grassland farms to inform future mitigation management

Opoku, D.G.[1,2], M.G. Healy[2], O. Fenton[1], K. Daly[1], T. Condon[1] and P. Tuohy[1]

[1]Teagasc, Ireland, [2]National University of Ireland, Galway, Ireland. Corresponding author: daniel.opoku@teagasc.ie

Open drains are an integral part of any poorly drained grassland farm infrastructure. Drainage systems provide advantages in terms of managing the water table height to enable grass production, however they also create a conduit for nutrient [phosphorus (P) and nitrogen (N)] losses to receiving waters. Recently, a farm-scale connectivity risk ranking for P loss along agricultural open drains was developed. The objective of the current field study was to incorporate N into this classification system, while classifying the impacts from in-field drains, groundwater upwelling and spring interactions. This will help guide future agricultural open drain management. An extensive fieldwork campaign was conducted across 10 heavy textured soil farms. In each farm, open drain networks were mapped and ranked in terms of their connectivity risk of P and N loss. In-field drains and connectivity to open drains and outlets, and groundwater interactions were noted. Using this information, an overall conceptual picture of the dominant loss pathways for both P and N was developed for each site. Spatial and temporal water samples were collected at key locations along the network and analysed for dissolved reactive P, particulate P, nitrate and ammonium. All the data were transferred to GIS and an integrated map of each farm was developed. A new classification system for the entire drainage network will be presented.

Keywords: open drains, nutrient loss, open drain classification, heavy textured grassland, drainage management

A5: Farming for a better future: Resilient and sustainable farming systems – Open Day Proceedings Teagasc, Johnstown Castle Environment Research Centre

Targeted mitigation: Breaking surface connectivity on farms – drainage ditches and roadways

Heerey, L., Opoku, D. G., Sifundza, L., Maher, P. J., Condon, T., Fenton, O., Daly, K., Tuohy, P., & Murnane, J

Article associated with Chapter 3.


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FARMING FOR A
BETTER FUTURE
Resilient and Sustainable
Farming Systems
OPEN DAY
Teagasc, Johnstown Castle
Environment Research Centre,
Co. Wexford
Tuesday, 16 July 2024

Targeted mitigation

Drainage ditches & Farm roadways

Main Points

Drainage ditches and roadways can act as sources and pathways for nutrients and sediment to enter waterbodies.

- Target risky ditch: farmyard connection
- Slow the flow: drops sediment and phosphorus
- Maintenance – P will build up over time

- Roadway sediment holds high concentrations of N and P
- Pathway – connectivity to nearby waterbodies
- Divert away from waterbodies

Funding

Department of Agriculture, Food and the Marine, and Teagasc

Projects: Road-Ready, Teagasc Heavy Soils Programme, SENSUS

Take home messages

- **Drainage ditches directly connecting farmyard to river pose greatest risk**
- **Roadway sediment contains very high concentrations of nutrients**
- **Need to select "right measure for right place"**

Targeted Mitigation: Breaking surface connectivity on farms – drainage ditches and roadways

Linda Heerey[1], Daniel Gyamfi Opoku[2], Lungile Sifundza[1], Paul J. Maher[2], Tomas Condon[2], Owen Fenton[1]; Karen Daly[1]; Patrick Tuohy[2]; John Murnane3.

[1]Teagasc, CELUP, Johnstown Castle, Co. Wexford; [2] Teagasc AGRIC, Moorepark; [3]University of Limerick

Summary:

Drainage ditches

- Drainage ditches are designed to move excess water away quickly from agricultural land to nearby rivers and lakes. However, they can potentially transport sediment and nutrients.
- In particular, drainage ditches which directly connect a farmyard to a river/lake pose the greatest risk for transporting nutrients.
- A range of in-ditch and pathway-control measures aim to mitigate against nutrient loss by breaking the pathway between the farm and the river/lake.
- In general, these measures aim to slow the flow of water so that the phosphorus and sediment being carried by the water is dropped, and to allow nitrogen to be attenuated.
- Very important that all measures are maintained and cleaned out, otherwise they risk becoming a source.

Farm roadways:

- Under the Nitrates Action Programme, water on farm roadways must not directly enter open drains or rivers/lakes.
- Sediment on roadways has been found to contain significantly high concentrations of nutrients all year round, and runoff from farm roadways can negatively impact water quality.
- Nutrient concentrations are high for all farm enterprises (i.e., beef, dairy and sheep).
- Particular areas of concern on farm roadways include the immediate area around the farmyard, and areas where livestock may be stalled (i.e., at junctions, bends).
- Connectivity can occur directly (e.g., runoff into drains, rivers, lakes etc.), or indirectly (e.g., farmyards).
- Mitigation measures aim to break connectivity between the source and watercourse, and a custom approach is best here.
- Examples include cambering road towards field (cross fall 1:25), concrete berms to direct runoff away from open waters, moving entry points to paddocks away from water course to reduce sediment/nutrient entering water course.

Other resources & online information

Email: owen.fenton@teagasc.ie; karen.daly@teagasc.ie

Acknowledgements: Thank you to Patricia Berry, Jessyca DeMedeiros, Brendan Healy, Simon Leach, Luis Lopez-Sangil, Linda Moloney Finn, Carmel O'Connor, and Wendy Pierce in Teagasc, Johnstown Castle for their assistance with laboratory sample analysis and creating figures for papers.

A6: Irish Dairying – Innovating for the Future. Open Day Proceedings. Teagasc Moorepark, Animal and Grassland Research Innovation Centre

Assessing connectivity risks on surface open drains to minimise nutrient losses

Opoku, D. G., Healy, M. G., Fenton, O., Daly, K., Condon, T., & Tuohy, P.

Article associated with Chapter 4.

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MOOREPARK '25
IRISH DAIRYING – INNOVATING FOR THE FUTURE

FBD INSURANCE

Assessing connectivity risks on surface open drains to minimise nutrient losses

Daniel Gyamfi Opoku[1,2], **Mark Healy**[2], **Owen Fenton**[3], **Karen Daly**[3], **Tomas Condon**[1] and **Patrick Tuohy**[1]

[1]*Teagasc, Animal & Grassland Research and Innovation Centre, Moorepark, Fermoy, Co. Cork;* [2]*Civil Engineering and Ryan Institute, College of Science and Engineering, University of Galway, Co. Galway;* [3]*Teagasc, Crops, Environment and Land Use Centre, Johnstown Castle, Wexford, Co. Wexford*

Summary

- Open drains are conduits for excess water from farm drainage systems
- Open drains contribute to nutrient losses to water sources through varied connectivity risks that require risk assessment
- On a per-farm basis, 2% to 25% of the open drains were classified as high-risk, and two-thirds of these were connected to the farmyards

Introduction

Approximately 30% of Ireland's managed grasslands are imperfectly or poorly drained and require the installation of drainage systems, especially under high rainfall conditions, to reach optimum grass production. Open drains, as part of such drainage systems, comprise of drainage ditches and smaller streams. These exist in high densities, and are networked to collect and drain away excess water from different parts of a farm to larger water courses.

The transport of water by open drains are key conduits for potential nutrient (phosphorus (P) and nitrogen (N)) loss, contributing to downstream water quality status. Nutrient transport by open drains varies spatially across the open drain network and depends on the open drains' connectivity to surface and subsurface flows. This is determined by complex factors including soil type, climate, landscape position, farm management, and nutrient input sources. Furthermore, nutrients are transformed or remain unchanged during transport through open drains, thus posing different risks prior to delivery to larger water courses. Research on how nutrient loss risk varies spatially within open drain networks is limited, with little understanding of the connectivity risks posed by each open drain. Understanding the connectivity and ranking the risk posed by individual open drains is essential for implementing effective targeted mitigation strategies to reduce nutrient losses from poorly-drained grassland soils in high rainfall areas.

This study highlights a semi-quantitative risk model to rank open drains by connectivity risk for nutrients loss and was undertaken across seven poorly-drained grassland farms. This model was designed based on field risk scoring of the landscape, hydrology, biogeochemical and management factors under the nutrient transfer continuum (NTC) framework. The NTC defines the nutrients loss processes from source, mobilisation, and pathway to receptor (S-M-P-R). The model enabled spatial identification of very high, high, medium, and low risk open drains to allow for targeted mitigation measures on risky (very high/high) open drains to break source-pathway connections for nutrients loss.

Results

A Risk Index Classification System was developed, and the results showed 2% to 25% of the open drains were classified as high-risk, with none identified as very high-risk (Figure 1). Two-thirds of the high-risk open drains were connected to the farmyards, with potential for high nutrient loss from farmyard point sources.

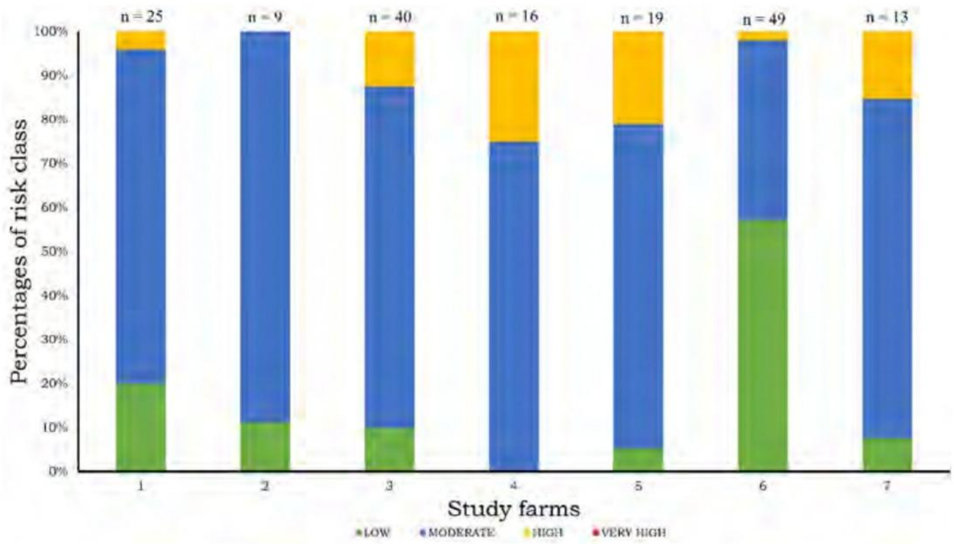

Figure 1. *Percentages of risk classes for open drainage channels for the seven study farms (n = number of open drains)*

Conclusion

A comprehensive but high data demanding risk assessment that characterises the complexity of connectivity risks in open drains on poorly-drained grassland farms is used to rank open drains. Open drains with farmyard connection pose the highest connectivity risk, and should be prioritised for mitigation. This risk assessment optimised resource use and supported evidence-based decision-making for developing and improving targeted mitigation measures to help improve water quality.

Acknowledgements

The authors are grateful to Teagasc for the award of a Walsh Scholarship to the first author [grant number: RMIS-1381] to conduct this research.

Appendix B

Table B 1. Observed nitrogen N connectivity percentage (%) occurrence across open ditches

	Number of occurred N connectivity pathways	N Connectivity pathways (%)			
		In-field drains	Internal roadways	Groundwater springs	Groundwater seepage / upwelling
Average across connectivity pathways	55	64	11	20	5
Farmyard connection	11	73	9	18	-
Outlet	9	67	-	22	11
Outflow	10	70	10	20	-
Secondary	25	56	16	20	8
Across the seven studied farms					
1	13	54	8	31	8
2	5	20	20	60	0
3	8	88	13	0	0
4	3	33	0	67	0
5	13	62	23	0	15
6	10	80	0	20	0
7	3	100	0	0	0

Table B 2. Average concentrations of N and P species at N connectivity pathways across outlet, outflow and secondary ditches.

N Connectivity	NH_4-N	NO_2-N	NO_3-N	TN	DRP	DUP	PP	TP
In-field drains	0.06	0.00	0.75	1.56	0.03	0.02	0.02	0.06
Roadway	0.62	0.00	0.17	2.46	0.19	0.13	0.12	0.42
Groundwater spring	0.06	0.10	1.90	2.39	0.01	0.00	0.04	0.05
Groundwater seepage/upwelling	0.09	0.00	0.65	1.36	0.04	0.02	0.24	0.29

APPENDIX C

Table C 1. Mean (± SE) inlet and outlet concentrations (mg L^{-1}), mean (± SE) removal efficiency (%) and statistical P-values of the different sediment pond configuration systems at location A, B and C for water quality parameters sampled.

Parameter	Mean Inlet Concentration (mg L^{-1})	Mean Outlet Concentration (mg L^{-1})	Mean Removal Efficiency%	P-value
Location A				
Nitrogen				
TN	1.04 ± 0.18	0.72 ± 0.11	30.95 ± 39.02	NS
NO_3-N	0.35 ± 0.09	0.44 ± 0.11	-26.41 ± 22.44	NS
NH_4-N	0.072 ± 0.02	0.02 ± 0.01	68.11 ± 66.00	*
Org N	0.62 ± 0.18	0.27 ± 0.05	57.39 ± 73.51	*
Phosphorus				
TP	0.08 ± 0.02	0.07 ± 0.01	17.00 ± 38.11	NS
DRP	0.03 ± 0.00	0.03 ± 0.01	3.99 ± 19.18	NS
PP	0.02 ± 0.01	0.01 ± 0.00	47.01 ± 60.25	NS
DUP	0.05 ± 0.02	0.03 ± 0.01	35.11 ± 75.80	NS
TSS	16.85 ± 8.07	6.24 ± 1.68	62.96 ± 79.16	**
Location B				
Nitrogen				
TN	1.30 ± 0.22	1.29 ± 0.19	0.46 ± 13.79	NS
NO_3-N	0.37 ± 0.14	0.68 ± 0.16	-86.36 ± 10.58	NS
NH_4-N	0.05 ± 0.02	0.07 ± 0.21	-504.82 ± 4.94	NS
Org N	0.88 ± 0.25	0.54 ± 0.08	39.02 ± 68.94	NS
Phosphorus				
TP	0.14 ± 0.03	0.10 ± 0.02	11.73 ± 7.11	NS
DRP	0.06 ± 0.01	0.042 ± 0.01	27.87 ± 44.56	NS
PP	0.03 ± 0.01	0.03 ± 0.01	7.09 ± 17.41	NS
DUP	0.05 ± 0.02	0.03 ± 0.01	39.84 ± 67.28	NS
TSS	66.45 ± 40.47	12.32 ± 3.69	81.46 ± 90.90	NS
Location C				
Nitrogen				
TN	1.28 ± 0.18	0.93 ± 0.10	27.39 ± 42.57	**
NO_3-N	0.43 ± 0.06	0.44 ± 0.04	-2.53 ± 23.09	NS
NH_4-N	0.02 ± 0.01	0.02 ± 0.01	-12.10 ± 8.40	NS
Org N	0.83 ± 0.15	0.47 ± 0.08	42.90 ± 49.41	**

Phosphorus				
TP	0.75 ± 0.01	0.08 ± 0.01	-10.44 ± 9.18	*
DRP	0.04 ± 0.01	0.04 ± 0.00	3.04 ± 21.92	NS
PP	0.01 ± 0.00	0.01 ± 0.00	1.14 ± 4.35	NS
DUP	0.03 ± 0.01	0.03 ± 0.01	-13.53 ± 1.00	*
TSS	12.25 ± 5.13	5.15 ± 0.79	57.93 ± 84.67	NS

34 * and ** indicates significance at $p < 0.05$ and $p < 0.01$, respectively. NS = non-significant.