Ecological Indicators 155 (2023) 111067

Contents lists available at ScienceDirect

Ecological Indicators

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

Quantifying nutrient and sediment erosion at riverbank cattle access points using fine-scale geo-spatial data

Alison Scott [a,b,*], Rachel Cassidy [b], Joerg Arnscheidt [a], David Rogers [a], Phil Jordan [a]

[a] School of Geography and Environmental Sciences, Ulster University, Coleraine, UK
[b] Agri-Environment Branch, Agri-Food and Biosciences Institute, Newforge Lane, Belfast, UK

A R T I C L E I N F O

Keywords:
Cattle access point
Photogrammetry
Terrestrial laser scanning
Total phosphorus
Sediment
Erosion

A B S T R A C T

Unrestricted cattle access to the riparian zone can exacerbate riverbank erosion in grazed grassland catchments. Knowledge gaps include the magnitude of erosion and other environmental pressures at cattle access points. This study aimed to address this by using two high resolution geo-spatial methods; 1) aerial photogrammetry and 2) terrestrial laser scanning to measure cumulative, seasonal, and annual erosion rates at nine unmitigated cattle access points in Northern Ireland. Total, fine sediment and total phosphorus exports were determined through bulk density and deep soil core sampling campaigns of exposed bank faces. Accumulated erosion was estimated using method 1) at 1.0 – 49.5 t and 0.51 – 16.64 kg for total sediment and total phosphorus, respectively. Using method 2) median annual export coefficients of 0.19 – 0.21 t m^{-1} and 0.065 – 0.087 kg m^{-1} (normalised to streambank length) were determined for total sediment and total phosphorus transfers respectively and these mostly occurred during the grazing season (median 84% for both sediment and total phosphorus). In terms of livestock pressures, these annual exports equate to 0.34 – 0.40 t LU^{-1} yr^{-1} and 0.103 – 0.111 kg LU^{-1} yr^{-1} for total sediment and total phosphorus, respectively (1.19–1.89 LU ha^{-1}). The conventional measure of protective fencing is likely to prevent such transfers to rivers. Scaling a nationwide agri-environment scheme over six years which installed 2,493 km of riparian fencing (and assuming from this study that 1.9 % of all riparian field boundaries had cattle access impact), this measure potentially saved 9,047–9,999 t yr^{-1} and 3,095 – 4,143 kg yr^{-1} of total sediment and total phosphorus, respectively, from entering water courses.

1. Introduction

Sediment inputs to freshwater systems are a significant environmental pressure in agricultural catchments (O'Sullivan et al., 2019) and can impact the whole aquatic ecosystem (Noe et al., 2020). Increased sedimentation from agricultural sources can occur due to a variety of processes including soil surface erosion from land when crop cover is removed and direct streambank erosion (Sherriff et al., 2015). Intensification of agricultural land use over several decades has exacerbated the issue (Van Zanten et al., 2014). While natural processes such as heavy rainfall and fluvial dynamics often drive this erosion, agricultural activities such as unsuitable, ill-timed tillage practices and unrestricted livestock access to river and lake banks can amplify their impacts (Holland, 2004; O'Callaghan et al., 2019).

Bank erosion has been identified as the main contributor of fine sediment (soil particles < 2 mm in diameter, including sand, silt, and clay) to surface waters globally (Collins et al., 2010; Zaimes et al., 2021). For example, a review by Fox et al. (2016) attributes up to 92% of in-stream suspended sediment loads in the United States to bank sources. Kronvang et al. (2013) also found bank erosion to be the primary source of in-stream sediment (90–94%) in the Odense catchment, Denmark during a three-year study. Excessive fine sediment inputs can have wide ranging ecological consequences including reduced primary production due to insufficient light penetration in turbid waters (Izagirre et al., 2009) and siltation of gravel beds with negative impacts on the provision of habitats for macroinvertebrates and spawning grounds for fish (Evans et al., 2006).

Bank erosion can also contribute nutrient inputs to surface waters. Particulate nutrients, primarily phosphorus (P) can be bound to sediment and follow its transfer dynamics (Dorioz et al., 2006). River and

Abbreviations: AES, Agri-environmental scheme; BD, Bulk density; DEM, Digital elevation model; DSM, Digital surface model; LU, Livestock unit; P, Phosphorus; SCP, Sustainable Catchments Pilot; SRP, Soluble reactive phosphorus; TLS, Terrestrial laser scanning; TP, Total phosphorus.

* Corresponding author.
E-mail address: scott-a28@ulster.ac.uk (A. Scott).

https://doi.org/10.1016/j.ecolind.2023.111067
Received 2 August 2023; Received in revised form 5 October 2023; Accepted 8 October 2023
Available online 14 October 2023

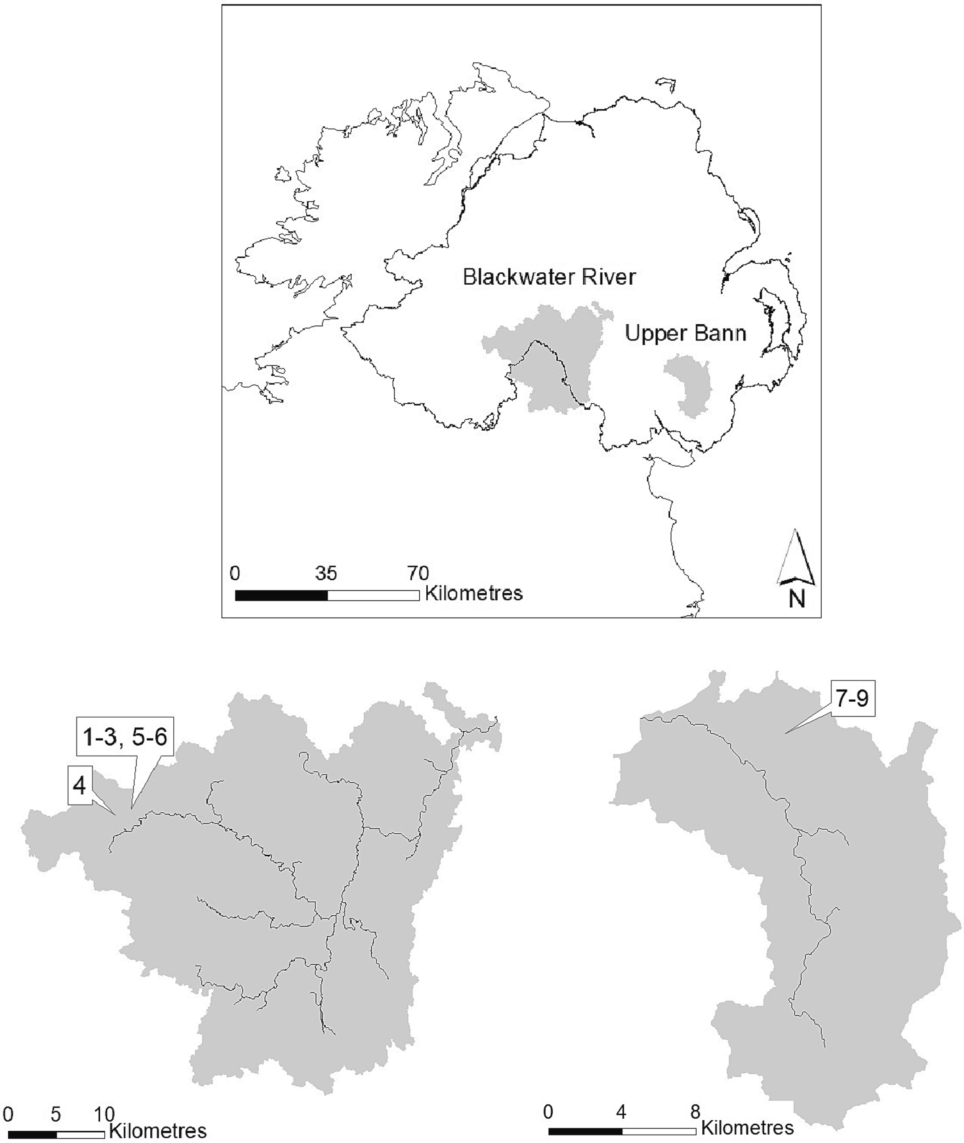

Fig. 1. Two study catchments in Northern Ireland and the locations of nine cattle access points surveyed by photogrammetry and TLS techniques (detailed reach locations are in Supplementary Material Fig. S1).

lake banks can represent a major P source, particularly in riparian margins where P can accumulate (Roberts et al., 2012). For example, in the Chesapeake Bay, USA, 73% of total phosphorus (TP) loads were bound to sediment particles (Noe et al., 2020). After entering the water, tracking the fate of this TP becomes very challenging due to its complex chemical and physical interactions with sediments and other particles. Depending on ambient conditions, sediment bound P may become detached and suspended within the water column until it settles within the bed sediment or is desorbed into soluble reactive phosphorus (SRP). This can have further ecological impacts as SRP is instantly bioavailable for uptake by autotrophs supporting their excessive growth which indicates eutrophication (Roberts et al., 2012).

In grazed grassland catchments, unrestricted cattle access to watercourses (for drinking water) can exacerbate natural bank erosion (Hughes, 2016; O'Callaghan et al., 2019; Rice et al., 2021). Cattle access points can be described as mobile point sources of sediment and nutrients (Rice et al., 2021), as these can be active or recovering at any one time. Areas along field boundaries where livestock gather can also result in soil compaction, creating new diffuse runoff pathways during rainfall events (Georgakakos et al., 2018; Pulley et al., 2021). Impacts of natural erosion processes are magnified as livestock trampling disturbs sediment by increasing hydraulic roughness and shear stress on banks due to greater morphological complexity (Trimble and Mendel, 1995). This, along with removal of riparian vegetation, creates discrete patches of bare earth which can be more susceptible to erosion during times of high streamflow (Evans et al., 2006). There is consensus within the literature that cattle access has a substantial effect on in stream sedimentation (O'Callaghan et al., 2019). Research at catchment scale in the USA (Line et al., 2016), Australia (McKergow et al., 2003), and New Zealand (Holmes et al., 2016) supports this. At river reach scale, Vidon et al. (2008) determined an 11-fold increase in total suspended sediment and a 13-fold increase in turbidity following a 12-month water sampling programme upstream and downstream from 130 m of riverbank impacted by cattle access in Indiana, USA. O'Sullivan et al. (2019) investigated the impact of cattle access in headwater streams in Ireland. While they found that there can be a substantial effect on in-stream deposited sediment with localised impacts, they also demonstrated cumulative downstream impacts of multiple upstream cattle access points. In Northern Ireland, Rice et al. (2021) determined that, during the grazing season, cattle are the primary driver of erosion where they have

Table 1
Summary of characteristics for the Blackwater River and Upper Bann sites.

Site number	Catchment	Bank length (m)	River reach (m)	Slope (°)	Bank height (m)	Soil type	Field size (ha)
1	Blackwater	28.3	229	35	1.5	Organic alluvium	5.68
2	Blackwater	26.3	229	75	1.3	Organic alluvium	5.68
3	Blackwater	14.2	229	40	1.4	Organic alluvium	5.68
4	Blackwater	6.8	482	35	1.7	Alluvium	18.20
5	Blackwater	19.4	229	20	0.9	Organic alluvium	5.68
6	Blackwater	21.5	229	20	0.8	Organic alluvium	5.68
7	Upper Bann	10.5	219	35	0.9	Ground water gley on alluvium	3.20
8	Upper Bann	3.8	166	60	1.0	Brown earth on sandstone till	1.30
9	Upper Bann	7.8	175	55	1.1	Brown earth on sandstone till	2.80
10	Upper Bann	7.5	85	60	1.5	Groundwater gley on alluvium	1.39
11	Upper Bann	2.0	114	45	4.0	Brown earth on sandstone till	0.95
12	Upper Bann	1.0	154	40	1.2	Brown earth on sandstone till	0.76

unrestricted access to bank faces. However, there have still been few investigations with comparisons of seasonal erosion rates of banks impacted by livestock or direct quantification of sediment and nutrient inputs from discrete access points.

In riparian areas where cattle access the watercourse total P can also accumulate and is then readily mobilised due to the disturbed nature of the bank face (O'Callaghan et al., 2019). Despite this, there is uncertainty regarding the water quality impact of cattle access in terms of nutrient contributions. Studies have measured changes in in-stream nutrient concentrations following the implementation of livestock exclusion fencing, with mixed results. McKergow et al. (2003) found only marginal improvements following livestock exclusion while others found significant TP and nitrogen (N) reductions (Line et al., 2016; Georgakakos et al., 2018). Few studies, however, have estimated the direct nutrient contribution from erosion at these sites.

Accurate measurements of bank stability at cattle access points are important to gain an understanding of sediment and nutrient dynamics in areas directly impacted by cattle. Traditionally, invasive methods such as erosion pins have been used to measure bank change in fluvial systems (Tufekcioglu et al., 2012; Kronvang et al., 2013). However, this method is largely unsuitable to quantify erosion at cattle access points due to coarse data resolution and bank disturbance caused by the pins (Myers et al., 2019). Aerial photogrammetry and LiDAR surveys (Thoma et al., 2005; Miller et al., 2010; Gkiatas et al., 2022; Hayes et al., 2023), and terrestrial laser scanning surveys (TLSs—Longoni et al., 2016; Rice et al. 2021) can create high resolution digital elevation models (DEMs) to accurately estimate bank retreat. However, DEM resolution between river reach and catchment scale needs to be considered.

Quantifying bank erosion from cattle access points is an important issue to address as exclusion fencing is a popular measure within agri-environmental schemes (AES) with water quality objectives (Kilgarriff et al., 2020). For example, according to the European Court of Auditors (2011), payments for AES measures should target areas where the greatest environmental benefits can be achieved (Madden et al., 2019). The EU and affiliated states are an important example as regulators are moving towards payments for environmental works and other results-based schemes. Few studies have considered the environmental benefits of fencing off cattle access points on a nationwide scale and novel geo-spatial methods provide a unique opportunity to achieve this. Therefore, the aim of this research was to quantify nutrient and sediment erosion at unfenced cattle access points and provide an evidence base for justifying future AES.

The objectives were to:

- Determine cumulative erosion from access points and quantify historical transfers of fine sediment and TP using an aerial photogrammetry survey.
- Estimate export coefficients for sediment and TP loss per unit length of streambank with repeated terrestrial laser scanning surveys.
- Upscale the results to provide an assessment of a national agri-environmental scheme.

2. Materials and methods

2.1. Study area and site selection

Six sites were selected in the Blackwater River and three in a sub-catchment of the Upper Bann in Northern Ireland (Fig. 1). Both catchments are characterised by intensive grassland agriculture in drumlin landscapes. The Blackwater River is a large cross border catchment between Co. Tyrone and Co. Monaghan and covers an area of 1,480 km^2 (Campbell et al., 2015). The river is 85 km in length and drains the largest catchment area of all of Lough Neagh's (surface area 382 km^2) six in flowing rivers (Campbell et al., 2015). Selected sites were in close proximity to each other within the Derrymeen sub-catchment at approximately 50 m above Ordinance Datum. Livestock agriculture is the primary land use, 75.9% grassland and 1.2% arable. Long term annual rainfall is 1,000–1200 mm in the lowlands and 1,200—1,600 mm in the uplands, and at the closest downstream river gauging station (183 km^2) long term annual runoff is 846 mm (National River Flow Archive, 2023). Underlying geology consists of Carboniferous sandstones, mudstones, limestones and shale (Campbell et al., 2015), and overlying soil types are primarily comprised of Stagnosols and Cambisols (World Reference Base—UKSO, 2023).

The Upper Bann catchment has an area of 305 km^2 (National River Flow Archive, 2023) and flows north into Lough Neagh. Stagnosols dominate the landscape with areas of Cambisols and Podzols (World Reference Base—UKSO, 2023). Bedrock is classified as Silurian greywacke, sandstones and shales alongside impermeable granites (Cassidy et al., 2019; Rice et al., 2021; National River Flow Archive, 2023). Land use is 95% permanent grassland pasture and 3% arable (Rice et al., 2021). Annual rainfall ranges between 600 and 800 mm in the lowlands and 800–1,200 mm in the uplands, and at the closest downstream river gauging station (305 km^2) long term annual runoff is 595 mm (National River Flow Archive, 2023).

Suitable sites were identified along stream banks with signs of recent sediment disturbance and where cattle had unrestricted access to the channel without an obvious alternative drinking water source. Five Blackwater River sites were located along a single reach of 180 m (sites 1–3, 5–6) and a sixth site was located on a second tributary 3.98 km south-west (site 4).

Using more specific soil type descriptors (1:50,000 General Soil Map of NI—AFBI, 2009) sites 1–3, 5–6 were comprised of organic alluvium while site 4 soils were alluvium. Three sites were also selected in the Upper Bann sub-catchment (Sites 7–9). These were all located along the same tributary. Soils at site 7 comprised of groundwater gley on alluvium, site 8 and 9 comprised of brown earth on a sandstone till. Sites were selected based on management regime, accessibility, and suitability for both aerial photogrammetry and terrestrial laser scanning

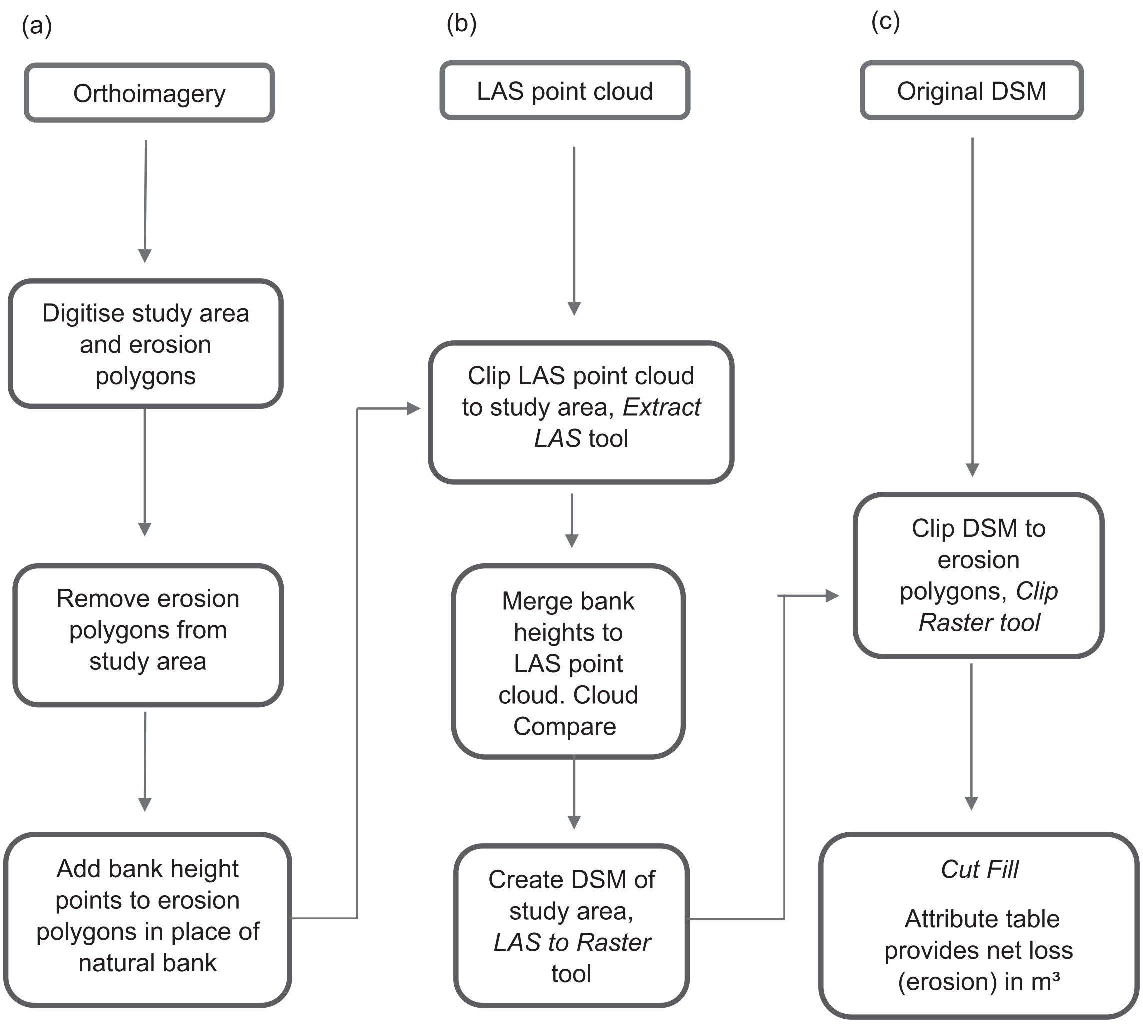

Fig. 2. Workflow chart of cumulative erosion volume calculations using ArcGIS ArcMap 10.8 software. LAS point cloud refers to the file format for points with elevation data obtained through photogrammetry.

surveys. Overhanging vegetation and deep, fast flowing streams would not facilitate these as vegetation would obscure views of the exposed sediment and equipment could not be placed into the stream bed to capture a complete image of the erosion scar (Heritage and Hetherington, 2007).

Cattle were present at each site for the duration of the grazing season; all cattle were from either continental bull beef or suckler enterprises. Herd sizes (in livestock units (LU)) were 1.89 LU ha^{-1} and 1.19 LU ha^{-1} for sites 1–3, 5–6 and 4 respectively. Herd size at site 7 was 1.67 LU ha^{-1} while herd sizes at sites 8 and 9 were 1.52 LU ha^{-1} and 1.67 LU ha^{-1} respectively. As is typical of UK and Irish livestock enterprises (Pulley et al., 2021), cattle are housed indoors during the winter months and banks are only disturbed by rainfall and stream hydraulic processes during this time. Site descriptions are presented in Table 1. There was no evidence of concentrated water runoff areas in the vicinity of the sites and erosion seemed to be reflective of cattle behaviour with some evidence of previous preferred accesses showing signs of recovery.

Orthoimagery of surveyed sites is available in supplementary material (Fig. S1). For context, available data were used from three other sites (Sites 10–12), surveyed using TLS in a previous study in a second subcatchment of the Upper Bann, and detailed site descriptions can be found in Rice et al. (2021).

2.2. *Cumulative erosion volumes*

For context, a single aerial photogrammetry survey was used to estimate cumulative erosion, i.e., the total volume of sediment lost from impacted sites when compared to natural or unimpacted adjacent banks. Blackwater River sites (1–3, 5–6) were captured in April 2021 using a fixed-wing drone (eBee X, AgEagle, USA), flown at 91 m altitude. Conditions were ideal for the survey with windspeeds of < 4 m s^{-1} and orthoimages and DEMs produced (PIX4Dmapper, PIX4D, Switzerland) had a resolution of approximately 2.7 cm. At site 4, a quadcopter drone (Mini 2, DJI, China) was used in November 2021 and at Upper Bann sites (7–9) in March 2022. These sites were adjacent to large trees and required a lower altitude survey, so this drone was flown at 15 m altitude and produced DEMs with a finer resolution of < 1 cm.

Cumulative erosion was estimated using GIS software (ArcMap 10.8, ArcGIS, USA) and the workflow stages are shown in Fig. 2. From the orthoimagery (Fig. 2a), polygons of the eroded areas were digitised, and these were removed from a larger study area polygon. Bank height points were created following the curve of the natural bank profile and were given elevation data by taking an average of bank heights from upstream and downstream using the 3D LAS-formatted point cloud along adjacent unimpacted river sections. The point cloud (Fig. 2b) was then clipped to the study area polygon to remove elevation data from inside the erosion scar and manually created bank heights were merged to the clipped point cloud. From this a digital surface model (DSM) was

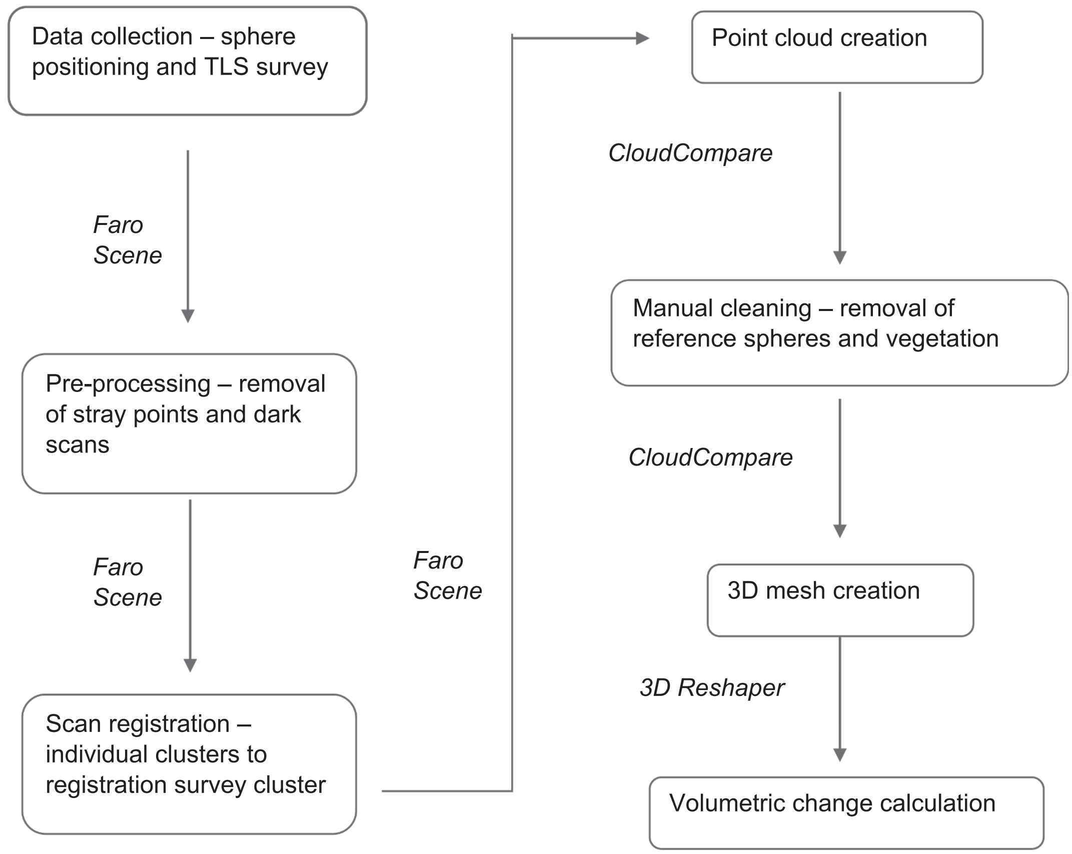

Fig. 3. Workflow chart of seasonal and annual erosion volume calculations from TLS surveys using Faro Scene, CloudCompare and 3D Reshaper software.

created using the linear interpolation method (Schwendel et al., 2012) to recreate an assumed natural surface prior to erosion. Both this and the original DSM (Fig. 2c) with post erosion elevation data were clipped to the erosion polygon extent and the volumetric difference was calculated using the Cut Fill tool in ArcGIS Spatial Analyst toolbox. Net loss (erosion) and net gain (accretion) volumes (m^3) were determined using the Cut Fill attribute table, providing the cumulative loss over an unknown period.

2.3. Seasonal and annual erosion rates

To determine seasonal and annual erosion rates, TLS surveys were undertaken on 5th May 2021, and repeated on 18th November 2021 and 8th March 2022 at the Blackwater River sites. Upper Bann sites were surveyed on 22nd March 2022, 1st December 2022 and 28th February 2023. The May to November study period covers the time sites were likely to be impacted by cattle access to the stream, i.e., the grazing season. November to March allows for the erosional impact of high river flows in the winter period to be quantified and compared. In total, three topographic surveys were completed for each site.

During each survey occasion, a single return TLS (Focus^{3D} X330, FARO, USA) was used to collect the topographic data (following Rice et al. (2021)). Depending on the size and complexity of the erosion scar, between four and six posts were fixed adjacent to each site at various heights to attach ground control reference spheres (Supplementary Material Fig. S3). This ensured ground control points were not impacted or moved by cattle interference and that multiple scans could be stitched together accurately. This enabled an accurate quantification of volumetric change between subsequent scans.

To create point clouds of each survey, individual scans were pre-processed, filtered, and registered (stitched together) using specialised software (SCENE 6.2.3.9, FARO, USA). Point clouds for each survey occasion were then manually cleaned to remove vegetation and anything obscuring bare sediment which would impact erosion volume calculations (Rice et al., 2021). Once cleaned, surface normals were calculated and corrected before 3D meshes were created. Cleaning, and mesh creation were completed using open-source software (CloudCompare V2.10, https://www.cloudcompare.org). The total erosion volumes (m^3) were then calculated by determining the difference between each mesh (3DReshaper, Leica Geosystems, Switzerland) (Rice et al., 2021) (Fig. 3). Erosion volumes were validated by using the point clouds to create DEMs and calculating Cut Fill models in ArcMap 10.8.

Riverbank erosion can be magnified by high runoff rates and flashy river discharge dynamics. To account for this influence on seasonal and annual erosion, daily discharge rates (mm day^{-1}) and flow duration curves (showing percentage flow exceedance) for the study year and previous four years were compared for each catchment. Data was obtained from the nearest downstream monitoring station for Blackwater sites (Derrymeen, station number 203022—NRFA, 2023) and Upper Bann sites (Moyallen, station number 203097—NRFA, 2023).

2.4. Streambank sediment and phosphorus characteristics

The proportion of sediment and TP available for export was seasonally and cumulatively quantified following three soil sampling campaigns. The first campaign determined the total sediment and proportion in the < 2 mm fraction. Four bulk density (BD) samples were collected from the top and bottom 50 cm of each exposed bank face using aluminium BD rings with a volume of 222 cm^3. These samples were dried at 105 °C for 24 h, weighed (for BD) and then disaggregated to pass through a 2 mm sieve for the proportion of fine sediment (Pulley et al., 2021). A comparative analysis was conducted using published BD values for each soil series according to the 1:50,000 General Soil Map of Northern Ireland (AFBI, 2009).

Table 2
Records of measured bulk density (BD), % fine sediment and TP in surface (T) and subsurface soils (B). Sites 1L and 1R (see Table 3) have been averaged as site 1 for this comparison.

Blackwater sites	1 T	1B	2 T	2B	3 T	3B	4 T	4B	5 T	5B	6 T	6B
Measured BD, (kg m^{-3})	915	1064	1015	1013	983	986	1095	1237	838	1010	918	1285
Fine sediment, (%)	100	98	100	95	99	98	100	100	98	97	99	98
TP, (g kg^{-1})	0.45	0.22	0.39	0.16	0.41	0.19	0.32	0.17	0.48	0.22	0.31	0.18
Upper Bann sites	7 T	7B	8 T	8B	9 T	9B						
Measured BD, (kg m^{-3})	1094	1010	790	890	1080	1000						
Fine sediment, (%)	100	100	100	100	98	98						
TP, (g kg^{-1})	0.79	0.63	0.56	0.47	0.72	0.71						

The second soil sampling campaign determined soil TP content. Four 1 m deep soil cores were collected adjacent to each site and composited into top 50 cm and bottom 50 cm samples. These were then air-dried and ground to pass through a 2 mm sieve. A subsample from each site was then ball milled to pass through a 150 μm sieve. This sample was then analysed following the sequential acid digest method (Bock, 1979). Soils were dried at 105 °C for 1 h and a 0.5000 g sub-sample was digested in hydrofluoric acid to remove silicates. Perchloric and nitric acid digests then removed organics, and dilute hydrochloric acid removed the remaining residue into solution with ultra-pure water. This process rendered any P within the sample soluble which was then analysed with spectrophotometry using the Murphy and Riley (1962) molybdate antimony method. Phosphorus concentrations were recorded as mass per unit mass of soil (g kg^{-1}). Subsequent data analysis assumed that each undisturbed soil profile adjacent to an erosion scar had similar TP characteristics to material that had been eroded already.

Combining cumulative and seasonal erosion volumes (m^3) with BD (kg m^{-3}) gave an estimate of eroded sediment (t) and TP (kg) which were also normalised by streambank length (m) of eroded areas (Tufekcioglu et al., 2012).

The third soil sampling campaign determined the agronomic soil test P status of the fields adjacent to the streambanks. In-field samples were collected on either side of the stream at the Blackwater sites (1–3, 5–6). Samples were collected to 7.5 cm depth following the standard W sampling method (40 sub-samples) and composited. Organic nutrients had already been applied at site 4 and so sampling this site in this manner would not have yielded meaningful results. These in-field samples were also analysed for Olsen plant-available P (Olsen et al., 1954) and collated to an index system for context. For sites 7–9 in the Upper Bann, Olsen P data were already available.

2.5. Data analysis and national scaling

Export coefficient ranges of sediment and TP transfers at cattle access points were estimated based on the range of data obtained at sites 1–9 and the two BD estimates (measured at each site and published according to soil type). Values for sites (10–12) surveyed in Rice et al. (2021) were included in these analyses for grazing season values of total and fine sediment. This enabled a first scaling of estimated losses that were potentially saved in an AES of fields that had previously been fenced to avoid further erosion by cattle access. For example, under a recent Northern Ireland Environmental Farming Scheme (EFS), 13,052 fields had boundaries adjacent to water courses and were fenced to prevent cattle access (DAERA, 2023). The mean length of these fenced boundaries was 191 m. In the current study, using all 12 study sites, approximately 1.9% of such boundaries had cattle access impact. Assuming a similar percentage for all unfenced riparian fields, this scales to an approximate accumulated 48 km of cattle access impact over 13,052 fields. The export coefficient ranges were then applied to this total length to determine how much sediment and TP were prevented from entering water courses.

However, it is rare to find any cost-benefit or cost avoidance data associated with mitigation measures in farmed landscapes. Here, an exploratory analysis was undertaken on the direct consequences of preventing sediment and TP from entering rivers and the method is appended in Supplementary Material.

Table 3
Summary of streambank soil characteristics. Soil series specific BDs are included for comparative purposes. Site 1 is split into right and left bank (1R and 1L) respectively as the site was split for TLS analysis.

Site	Bulk density (Measured) (kg m^{-3})	Bulk density (Soil series) (kg m^{-3})	Fine sediment (%)	TP concentration (g kg^{-1}) soil
1R	1055	1150	98	0.26
1L	924	1150	100	0.42
2	1014	1150	97	0.27
3	984	1150	99	0.30
4	1170	1360	100	0.25
5	924	1150	97	0.35
6	1102	1150	99	0.24
7	1053	1120	100	0.71
8	842	1120	100	0.52
9	1040	1120	98	0.71
10	–	993	38	–
11	–	1350	27	–
12	–	1350	41	–

3. Results

3.1. Streambank fine sediment and phosphorus (bank and field)

Individual records of measured BD, percentage fine sediment and TP concentrations in surface (top 50 cm of exposed stream bank) and subsurface soils (bottom 50 cm of exposed stream bank) are shown in Table 2. Subsurface soil TP was on average 34% less than surface soil TP; the exception was site 9 where TP concentration was uniform through the soil profile (Table 2).

Data from Table 2 were collated into full bank profiles (0–1 m depth) and are presented in Table 3 and bulk density (measured and published data) was used as the metric for bank erodibility. Through this analysis all soils were found to comprise almost entirely of fine sediment. Available bank characteristics of sites surveyed in Rice et al. (2021) are

Table 4
Cumulative erosion volumes expressed in m^3 mass total sediment (not limited to the fine sediment fraction) and TP exports expressed in t and kg respectively, also normalised by streambank length (t m^{-1} and kg m^{-1}).

Site	Erosion (m^3)	Erosion (m^3 m^{-1})	Total sediment (t)	Total sediment (t m^{-1})	TP (kg)	TP (kg m^{-1})
1	50.0	1.76	49.50	1.75	16.64	0.59
2	14.0	0.53	14.00	0.53	3.80	0.15
3	4.3	0.30	4.20	0.30	1.27	0.09
4	12.5	1.85	14.60	2.16	3.59	0.53
5	10.2	0.52	9.40	0.48	3.25	0.17
6	15.8	0.73	17.40	0.81	4.19	0.19
7	1.9	0.18	2.00	0.19	1.43	0.14
8	1.2	0.32	1.00	0.26	0.51	0.13
9	3.1	0.40	3.20	0.42	2.32	0.30

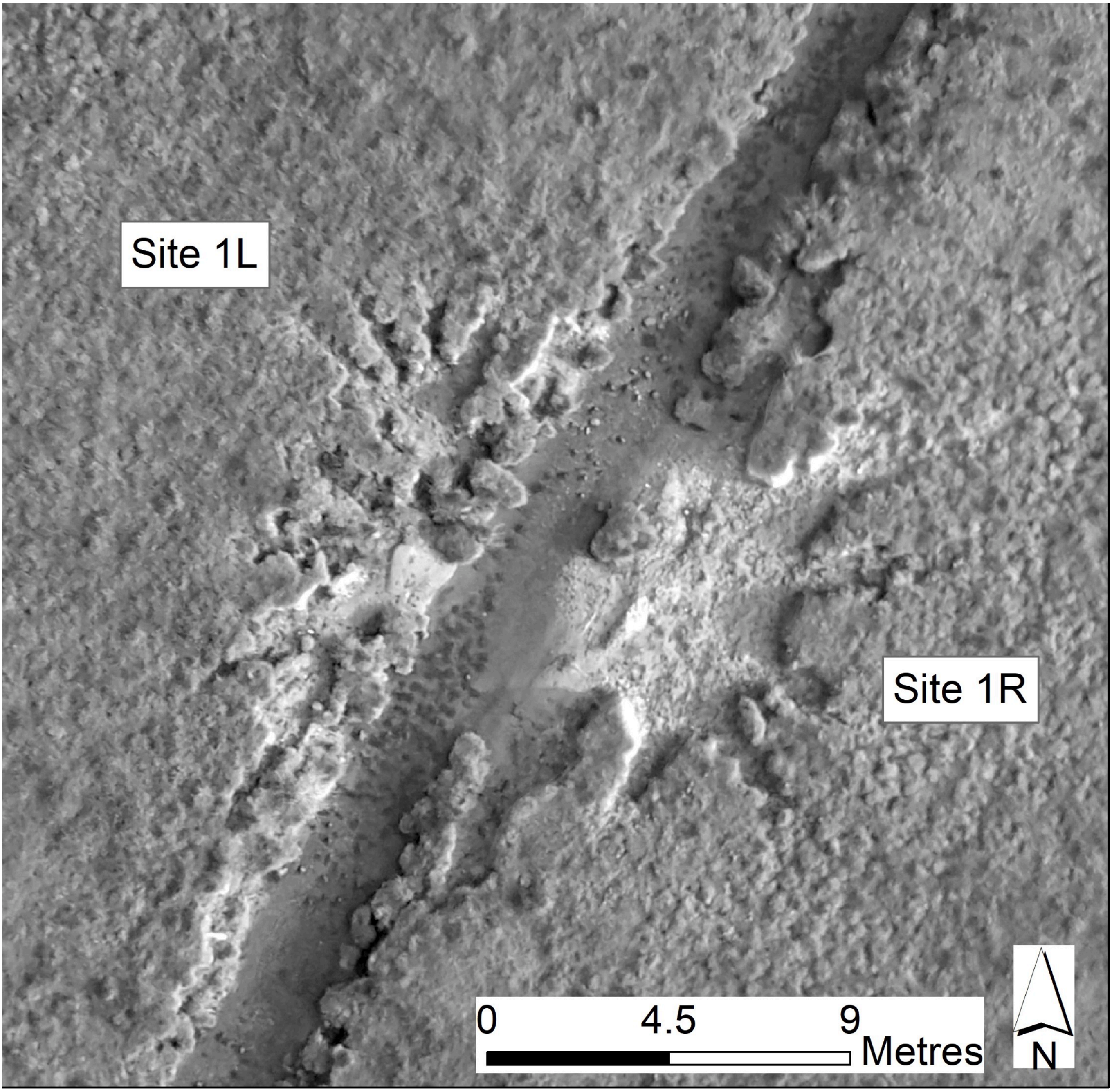

Fig. 4. Example of orthoimagery of a cattle access erosion scar obtained through the aerial photogrammetry survey of site 1L and 1R in the Blackwater catchment (images of all studied sites are in Supplementary Material Fig. S2).

also included in Table 3 and had considerably lower fine sediment percentage contents than sites surveyed in the current study. Sequential acid digests found bank soils to contain high concentrations of TP, ranging between 0.2 and 0.7 g kg^{-1} soil (Table 3), similar to those reported in Granger et al. (2021).

Olsen P concentrations in the fields adjacent to the stream banks were collated to agronomic indices. Sites 1–3 and 5–6 were found to be index 2+ (optimum for intensive grassland). Sites 8 and 9 were index 2- (optimum for low intensity grassland), and site 7 was index 3 (above optimum for grassland).

3.2. Cumulative erosion

Hindcasted erosion volumes varied, ranging between 1.2 – 50 m^3. Cumulative total sediment and TP losses ranged from 1.0 to 49.5 t and 0.51 – 16.64 kg, respectively (Table 4). This large range can be attributed to the variation in the length and 2D area of each erosion scar; site 1 was a large area and accounted for both sides of the bank at the impacted stream section (i.e., 1R and 1L in Table 3 and Fig. 4). Exports of total sediment mass and TP were also considerable when normalised by bank length (0.191 – 2.16 t m^{-1}) and (0.134 – 0.587 kg m^{-1}) respectively (Table 4).

3.3. Seasonal and annual erosion rates

To determine seasonal and annual erosion rates using TLS, it was important that mean distance errors remained low during processing, and these were found to be 3.1 mm on average and all < 6 mm

Table 5
Annual and seasonal erosion rates (m^3 m^{-1}), mass total and fine sediment (t m^{-1}) and mass TP (kg m^{-1}) exports from sites surveyed using TLS normalised by streambank length, using measured and published BDs. Site 1 was separated into 1R and 1L for this analysis.

Site and season	Erosion (m^3 m^{-1})	Total sediment (t m^{-1})	Fine sediment (t m^{-1})	TP (kg m^{-1})
1R				
Grazing	0.32	0.34–0.37	0.33–0.36	0.086–0.094
Winter	0.05	0.05–0.05	0.05–0.05	0.012–0.013
Annual	0.36	0.38–0.42	0.38–0.41	0.098–0.107
1L				
Grazing	0.41	0.38–0.47	0.38–0.47	0.159–0.198
Winter	0.08	0.08–0.09	0.08–0.09	0.031–0.039
Annual	0.49	0.46–0.57	0.46–0.57	0.190–0.237
2				
Grazing	0.10	0.10–0.11	0.09–0.11	0.026–0.030
Winter	0.04	0.04–0.04	0.04–0.04	0.010–0.011
Annual	0.13	0.13–0.15	0.13–0.15	0.036–0.041
3				
Grazing	0.12	0.12–0.14	0.12–0.14	0.037–0.043
Winter	0.05	0.05–0.06	0.05–0.06	0.015–0.017
Annual	0.17	0.17–0.20	0.17–0.20	0.052–0.060
4				
Grazing	0.06	0.07–0.08	0.07–0.08	0.017–0.020
Winter	0.10	0.12–0.14	0.12–0.14	0.029–0.033
Annual	0.16	0.19–0.22	0.19–0.22	0.046–0.054
7				
Grazing	0.09	0.09–0.10	0.09–0.10	0.067–0.071
8				
Grazing	0.14	0.12–0.15	0.12–0.15	0.060–0.080
Winter	0.01	0.01–0.01	0.01–0.01	0.005–0.007
Annual	0.15	0.13–0.17	0.13–0.17	0.065–0.087
9				
Grazing	0.16	0.17–0.18	0.17–0.18	0.122–0.132
Winter	0.03	0.03–0.03	0.03–0.03	0.019–0.020
Annual	0.19	0.20–0.21	0.19–0.21	0.141–0.152
10				
Grazing	0.14	0.14	0.05	–
11				
Grazing	0.27	0.36	0.10	–
12				
Grazing	0.14	0.19	0.08	–

(Table S1), giving confidence that individual scans were stitched together accurately.

For the sole analysis of areas of active erosion, sites were clipped within the erosion scars; therefore, bank lengths differ from those used in the cumulative erosion analysis and these differences are outlined in Table S2. Erosion volumes were normalised by streambank length and are presented in Table 5. These varied between sites with annual totals ranging between 0.13 – 0.49 m^3 m^{-1}. In general, elevated erosion rates were observed following the grazing season compared to the winter period (0.06 – 0.41 m^3 m^{-1} and 0.01 – 0.1 m^3 m^{-1}, respectively). Only one of the eight sites (site 4) had greater erosion during the winter than during the grazing season (0.10 m^3 m^{-1} and 0.06 m^3 m^{-1}, respectively). Fill volumes were attributed to sediment eroded upslope and deposited downslope as per Rice et al. (2021) and therefore excluded from analysis. The mass of total and fine sediment, and TP exported during these periods were also normalised by streambank length using two BD values (measured and published—Table 3). As the bank soils predominantly comprised of fine sediment (Table 3) fine sediment losses were considerable, ranging between (0.07 – 0.47 t m^{-1}) during the grazing season. Total P losses during the grazing season ranged between 0.017 and 0.198 kg m^{-1}. Estimates of mass fine sediment loss at sites (10–12) surveyed in Rice et al., (2021) using published bulk densities and reported percentage fine sediment are also included in Table 5 and are within range of sites surveyed in in the current study.

To account for the influence of variable hydrological conditions, timeseries of river discharge levels for the study years and previous four years are shown from the nearest downstream gauging stations (with reference to the grazing periods) in Fig. 5a and 5b. This analysis indicated that runoff during the study periods was not exceptional, and the conditions were either typical or less than typical for each catchment. This is further demonstrated by flow duration curves with exceedance percentiles in Fig. 6a—d. These data indicate that hydrological conditions were not overly influential on seasonal or annual erosion rates during the study year.

3.4. Cattle access point export coefficients and national scaling

The data in Table 5 enables annual export coefficients for total and fine sediment, and TP to be estimated for cattle access points in general, based on medians within a statistical range. For example, using the measured and published BD values, export coefficients are 0.19 – 0.21 t m^{-1} yr^{-1} for sediment (both total and fine due up to 100% fine sediment composition for sites 1–9) with an interquartile range (IQR) of 0.17 t m^{-1} yr^{-1}. For TP the export coefficient is 0.065 – 0.087 kg m^{-1} yr^{-1} (IQR of 0.080 kg m^{-1} yr^{-1}). Similarly, using livestock and field area data combined with annual exports of sediment and TP gave a median livestock population of 4.67 LU (section 2.1.) and export coefficients per livestock unit of 0.34 – 0.40 t LU^{-1} yr^{-1} (IQR 1.09 t LU^{-1} yr^{-1}) and 0.103 – 0.111 kg LU^{-1} yr^{-1} (IQR 0.245 kg LU^{-1} yr^{-1}), respectively.

The conventional measure of protective fencing is likely to prevent such transfers to rivers and this was undertaken in a nationwide AES which installed 2,493 km of riparian fencing in 13,052 fields over six years. In the current study, using all 12 sites, approximately 1.9% of such field boundaries had cattle access. Assuming a similar percentage for all unfenced riparian fields, this scales to an approximate accumulated 48 km of cattle access impact and this fencing potentially saved 9,047–9,999 t yr^{-1} and 3,095 – 4,143 kg yr^{-1} of total sediment and total phosphorus, respectively, from entering water courses.

When placed into context of the most recent EFS (13,052 fields with boundaries adjacent to streams), a first assessment of cost avoidance using two direct metrics (Supplementary Material) was that for every £1 spent on fencing, approximately £0.56 would be avoided from sediment and TP loss over 15 years (not accounting for inflation) but this excludes all other ecosystem service benefits.

4. Discussion

4.1. Research findings context and limitations

Results from this study can be placed in context with previously published literature, primarily in the US as there is a paucity of research into streambank erosion as a result of cattle access in other temperate regions. Tufekcioglu et al., (2012) quantified erosion along thirteen grazed river reaches in Iowa—total soil and TP losses ranged between 0.06 and 0.61 t m^{-1} y^{-1} and 0.02 – 0.183 kg m^{-1} y^{-1} respectively. Also, in Iowa, Zaimes et al., (2008) observed soil and TP losses between 0.20 and 0.27 t m^{-1} y^{-1} and 0.071 – 0.123 kg m^{-1} y^{-1} respectively, in continuously grazed pastures. While both these studies used the erosion pin method to measure bank retreat, the annual coefficient of total sediment and TP loss observed in this study are within these ranges (median 0.19 – 0.21 t m^{-1} yr^{-1} and 0.065 – 0.087 kg m^{-1} yr^{-1}, respectively).

However, Hayes et al. (2023) conducted a similar riverbank erosion survey at sites 1–3, 5–6 in the Blackwater catchment ("Site 4, Riverbank A and B" in Hayes et al., 2023) using LiDAR imagery for 2014–2020 with a DEM resolution of 1000 mm (±150 mm). The average annual sediment erosion rates reported over four years were at least four orders of magnitude lower. It should be noted that their study used data of a coarser resolution and only analysed a change in height (z coordinate) which may not have accounted for lateral bank retreat or undercutting below the exposed bank face, which this study was able to determine with TLS (with a resolution of ± 2 mm and an uncertainty between scans of < 6 mm). It is likely therefore, that the Hayes et al. (2023) study underpredicted erosion rates rather than the rates reported in the

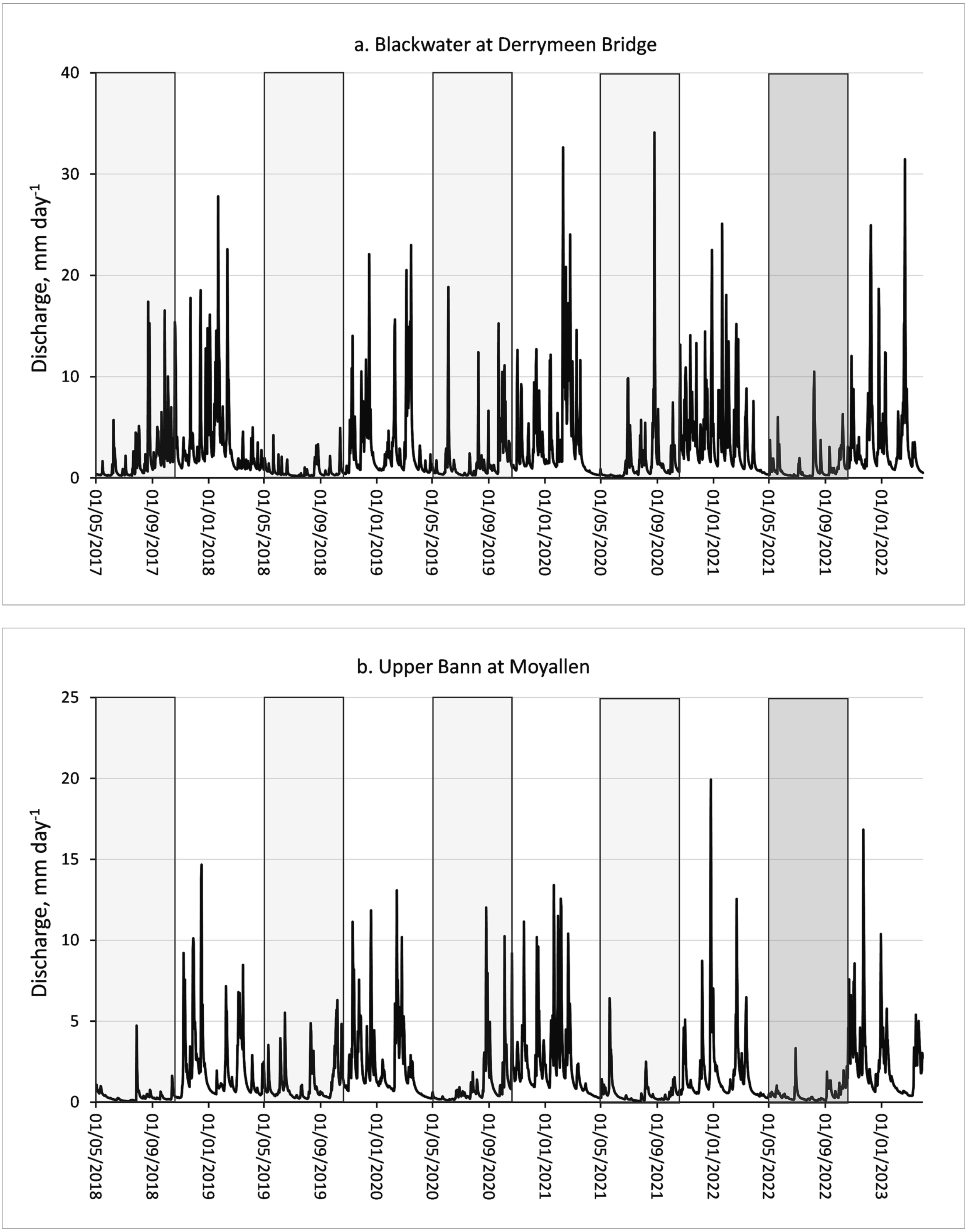

Fig. 5. Timeseries of river discharge in mm day^{-1} for the study periods and previous four years in the two study catchments a) Blackwater (TLS survey dates were 05/05/21, 18/11/21, 08/03/22) and b) Upper Bann (22/03/22, 01/12/22, 28/02/23) at nearest downstream gauging stations. The grazing season for each year is indicated by the grey areas with the studied grazing season a darker grey (and winter season in between).

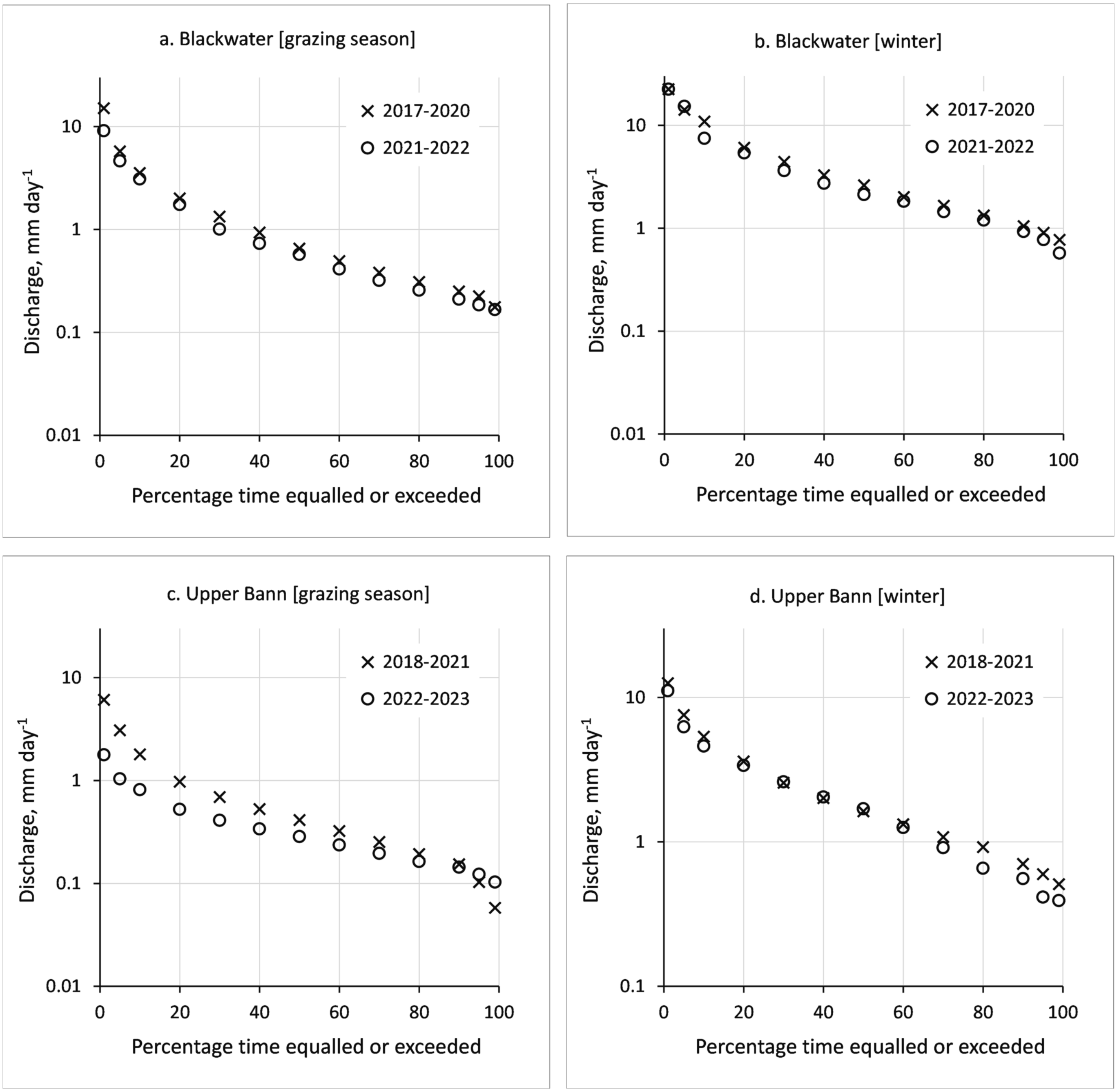

Fig. 6. Flow duration curves for the grazing and winter periods for the two studied catchments Blackwater (a—b, respectively) and the Upper Bann (c—d, respectively), indicating unexceptional hydrological conditions between the study years and previous four years.

current study representing a year of exceptionally high erosion.

Nevertheless, the methods used here also have limitations, particularly the ability of TLS to produce high density point clouds on vegetated bank faces (Myers et al., 2019). The cost and time-consuming nature of the analysis outlined in this study may also limit this method's applicability to larger datasets with multiple sites. While effort is made to select sites with minimal vegetation, it is difficult to find sites which are not impacted by vegetation in some way. Manual removal of pixels attributed to vegetation and other obstructions (Longoni et al., 2016; Rice et al., 2021) is a thorough method of obstruction removal but is time intensive. Plugins are available to automatically remove vegetation, but these can result in data gaps, possibly underestimating bank change (Myers et al., 2019). LiDAR drones are also becoming more available which can capture data of a comparable resolution which can be analysed with greater ease using GIS software (Resop et al., 2019). A further consideration is the inability of all digital surveying techniques being restricted to areas above the water line (Longoni et al., 2016). When measuring bank erosion, the spatial and temporal scales, research aim, and available resources should therefore be considered. This might also include controlling for livestock intensity, and longer hydrometeorological cycles as well as the geomorphic factors listed in Table 1.

4.2. Policy implications

The scaled befits of fencing over a wide region to exclude cattle access has provided the first quantification of sediment and TP prevented from polluting water courses (9,047–9,999 t yr^{-1} and 3,095 – 4,143 kg yr^{-1} of total sediment and total phosphorus, respectively). While high spatial accuracy was gained using photogrammetry for cumulative and TLS for seasonal erosion rates at the 9 study sites, a degree of uncertainty was introduced when scaling the results to a wider region. The percentage of cattle access points per adjacent (to river) field boundary (1.91%) and the annual/seasonal export coefficients estimated for both sediment and TP, even with inclusion of IQRs, need to be considered as uncertain estimates when scaled up. Nevertheless, these rates and scaled estimates can be used as a first assessment and can be improved when further data become available.

While the low cost avoidance ratio estimate is similar to Schulte et al. (2009) who concluded that fencing is an expensive solution for managing water quality, the current study only investigated two metrics (sediment and TP). Other ecosystem services provided by fencing would also need to be taken into account which may increase the environmental and economic value of excluding livestock from riparian zones, and this is a research priority.

For example, as cattle are more likely to defecate within and adjacent to surface waters (Davies-Colley et al., 2004) riverbank access points can also be sources of faecal bacteria and pathogens, which may have negative impacts on the health of both humans and livestock (Kilgarriff et al., 2020). Therefore, there are likely to be biosecurity benefits to riparian fencing, the economics of which are not considered in this study. Furthermore, managing bovine tuberculosis (bTB) is a substantial cost to the farming economies, (approximately £40 million per year) with half of this cost paid as compensation for infected cattle (DAERA, 2021). The disease can be spread via "nose-to-nose" contact between cattle (O'Hagan et al., 2016) or more rarely, between cattle and badgers (Campbell et al., 2019). Natural drinking water sources are often shared between livestock and wildlife, where streams are unfenced, facilitating the environmental spread of bTB pathogens (Barasona et al., 2017). Mass reductions of faecal deposits per day were used by Kilgarriff et al. (2020) to quantify the cost benefits of implementation of a national agri-environmental policy to fence all watercourses on derogated farms in Ireland. The study compared cost benefits between farms with high and low stocking densities, with fencing on high intensity farms being ~ 20% more cost effective than low intensity farms.

There are other benefits which may be associated with riparian fencing as livestock access can cause further water quality pressures due to livestock behaviour and the amount of time they spend in and around watercourses (Davies-Colley et al., 2004; Pulley et al., 2021). Indirect effects of increased water temperatures and turbidity due to changes in stream morphology, suspended sediment levels and riparian vegetation structure can all have negative impacts on stream ecology in areas where cattle have unmitigated access to the riparian zone (O'Callaghan et al., 2019; O'Sullivan et al., 2019). Elevated turbidity has also been reported to negatively impact water treatment costs (Price et al., 2017). Improvements in riparian vegetation have also been observed following livestock exclusion (Miller et al., 2010). These factors all have the capacity to enable a self-sustaining fish population (trout/salmon) which would also need to be economically accounted for.

To maximise cost-benefits, policy makers may choose to take a targeted approach, such as headwater streams where multiple access points can result in both localized and downstream water quality pressures (O'Sullivan, 2019), or areas with greater stocking rates (Kilgariff et al., 2020). Fencing is a popular measure within voluntary AES (Thomas et al., 2019) for multiple reasons and therefore can act as an important incentive for farmers to enter these schemes and possibly take on further environmental measures, which may lead to other unintended benefits (such as biosecurity).

5. Conclusions

To quantify sediment and TP contribution directly from erosion at unmitigated cattle access points, high resolution aerial photogrammetry facilitated the hindcasting of cumulative erosion volumes and repeated TLS surveys determined seasonal and annual erosion rates and export coefficients. Cumulative total sediment and TP loss ranged between 0.19 and 2.16 t m^{-1} and 0.134 – 0.587 kg m^{-1} of streambank, respectively. Median annual export coefficients were determined for sediment (0.19 – 0.21 t m^{-1} yr^{-1}) and TP (0.065 – 0.087 kg m^{-1} yr^{-1}); these were also determined per livestock unit (0.34 – 0.40 t LU^{-1} yr^{-1} for sediment 0.103 – 0.111 kg LU^{-1} yr^{-1} for TP). While these export coefficients are transferable using high resolution geo-spatial methods, more data will improve their utility over wider areas.

Scaled to a national AES where 13,052 fields were fenced at their riparian boundaries and excluding 2,493 km of watercourses from cattle access prevented the transfer of 9,047–9,999 t yr^{-1} and 3,095 – 4,143 kg yr^{-1} of total sediment and total phosphorus, respectively to rivers. A research priority will be the full economic costs and benefits analysis of riparian fencing (more than the direct savings associated with sediment and phosphorus) such as the analysis of other environmental and agricultural goods and services. This information is important to policy makers who may consider or have to justify riparian fencing in future, targeted schemes to improve water quality in agricultural catchments.

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.

Data availability

The data that has been used is confidential.

Acknowledgements

This work was funded by a DAERA PhD Scholarship (A. Scott) with additional resources from DAERA Evidence and Innovation project 16/4/03. We acknowledge and thank Rivers Trust (Dominic McCann), Armagh, Banbridge and Craigavon Borough Council (Tom Woods), AFBI (Brian Wallace, Joe Larkin, Hugh McKeating) and Ulster University (Peter Devlin) staff for their support with site identification, field and laboratory work. We acknowledge hydrometeorological data provided by the Centre for Ecology and Hydrology, Great Britain and the Department for Infrastructure, Northern Ireland. We also thank landowners for access to study sites.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2023.111067.

References

AFBI, 2009. General soil map of Northern Ireland 1:50,000. In: AFBI (Ed.), Northern Ireland Soil Survey. Agri-Food and Biosciences Institute.

Archive, N.R.F., 2023. National water archive. Natural Environment Research Council, CEH Wallingford https://nrfa.ceh.ac.uk/data (Accessed 8 September 2023.

Barasona, J.A., Vicente, J., Díez-Delgado, I., Aznar, J., Gortázar, C., Torres, M.J., 2017. Environmental presence of mycobacterium tuberculosis complex in aggregation points at the wildlife/livestock interface. Transboundary and Emerging Diseases. 64, 1148–1158. https://doi.org/10.1111/tbed.12480.

Bock., R., 1979. A Handbook of Decomposition Methods in Analytical Chemistry. International Textbook Company, Glasgow.

Campbell, E.L., Byrne, A.W., Menzies, F.D., McBride, K.R., McCormick, C.M., Scantlebury, M., Reid, N., 2019. Interspecific visitation of cattle and badgers to fomites: A transmission risk for bovine tuberculosis? Ecology and Evolution 9, 8479–8489. https://doi.org/10.1002/ece3.5282.

Campbell, J.M., Jordan, P., Arnscheidt, J., 2015. Using high-resolution phosphorus data to investigate mitigation measures in headwater river catchments. Hydrology and Earth System Sciences. 19, 453–464. https://doi.org/10.5194/hess-19-453-2015.

Cassidy, R., Thomas, I.A., Higgins, A., Bailey, J.S., Jordan, P., 2019. A carrying capacity framework for soil phosphorus and hydrological sensitivity from farm to catchment scales. Science of the total Environment. 687, 277–286. https://doi.org/10.1016/j.scitotenv.2019.05.453.

Collins, A.L., Walling, D.E., McMellin, G.K., Zhang, Y., Gray, J., McGonigle, D., Cherrington, R., 2010. A preliminary investigation of the efficacy of riparian fencing schemes for reducing contributions from eroding channel banks to the siltation of salmonids pawning gravels across the south west UK. Journal of Environmental Management. 91, 1341–1349. https://doi.org/10.1016/j.jenvman.2010.02.015.

European Court of Auditors, 2011. Is agri-environment support well designed and managed? Special Report No 7, Luxembourg. https://www.eca.europa.eu/Lists/ECADocuments/SR11_07/SR11_07_EN.PDF.

DAERA, 2021. Bovine Tuberculosis (bTB) Consultation on the Department's Proposed Implementation and Next Steps of the bTB Eradication Strategy for Northern Ireland.

Davies-Colley, R.J., Nagels, J.W., Smith, R.A., Young, R.G., Phillips, C.J., 2004. Water quality impact of a dairy cow herd crossing a stream. New Zealand Journal of Marine and Freshwater Research. 38, 569–576. https://doi.org/10.1080/00288330.2004.9517262.

Dorioz, J.M., Wang, D., Poulenard, J., Trévisan, D., 2006. The effect of grass buffer strips on phosphorus dynamics-A critical review and synthesis as a basis for application in agricultural landscapes in France. Agriculture, Ecosystems and Environment. 117, 4-21, https://doi.org/10.1016/j.agee.2006.03.029.

Evans, D.J., Gibson, C.E., Rossell, R.S., 2006. Sediment loads and sources in heavily modified irish catchments: A move towards informed management strategies. Geomorphology 79, 93–113. https://doi.org/10.1016/j.geomorph.2005.09.018.

Fox, G.A., Purvis, R.A., Penn, C.J., 2016. Streambanks: A net source of sediment and phosphorus to streams and rivers. Journal of Environmental Management. 181, 602–614. https://doi.org/10.1016/j.jenvman.2016.06.071.

Georgakakos, C.B., Morris, C.K., Walter, M.T., 2018. Challenges and opportunities with on-farm research: Total and soluble reactive stream phosphorus before and after implementation of a cattle-exclusion, riparian buffer. Frontiers in Environmental Science. 6, 71. https://doi.org/10.3389/fenvs.2018.00071.

Gkiatas, G.T., Koutalakis, P.D., Kasapidis, I.K., Iakovoglou, V., Zaimes, G.N., 2022. Monitoring and quantifying the Fluvio-Geomorphological changes in a torrent channel using images from unmanned aerial vehicles. Hydrology. 9 (10), 184. https://doi.org/10.3390/hydrology9100184.

Granger, S.J., Harris, P., Upadhayay, H.R., Sint, H., Pulley, S., Stone, M., Krishnappan, B. G., Collins, A.L., 2021. Novel approaches to investigating spatial variability in channel bank total phosphorus at the catchment scale. Catena 202, 105223. https://doi.org/10.1016/j.catena.2021.105223.

Hayes, E., Higgins, S., Mullan, D., Geris, J., 2023. High-resolution assessment of riverbank erosion and stabilization techniques with associated water quality implications. International Journal of River Basin Management. 1 https://doi.org/10.1080/15715124.2023.2214866.

Heritage, G., Hetherington, D., 2007. Towards a protocol for laser scanning in fluvial geomorphology. Earth Surface Processes and Landforms. 32, 66–74. https://doi.org/10.1002/esp.1375.

Holland, J.M., 2004. The environmental consequences of adopting conservation tillage in europe: Reviewing The evidence. Agriculture, Ecosystems and Environment. 103, 1–25. https://doi.org/10.1016/j.agee.2003.12.018.

Holmes, R., Hayes, J., Matthaei, C., Closs, G., Williams, M., Goodwin, E., 2016. Riparian management affects instream habitat condition in a dairy stream catchment. New Zealand Journal of Marine and Freshwater Research. 50, 581–599. https://doi.org/10.1080/00288330.2016.1184169.

Hughes, A.O., 2016. Riparian management and stream bank erosion in new zealand. new zealand Journal of Marine and Freshwater Research. 50, 277–290. https://doi.org/10.1080/00288330.2015.1116449.

Izagirre, O., Serra, A., Guasch, H., Elosegi, A., 2009. Effects of sediment deposition on periphytic biomass, photosynthetic activity and algal community structure. Science of the Total Environment. 407, 5694–5700. https://doi.org/10.1016/j.scitotenv.2009.06.049.

Kilgarriff, P., Ryan, M., O'Donoghue, C., Green, S., Ó hUallacháin, D., 2020. Livestock exclusion from watercourses: Policy effectiveness and implications. Environmental Science and Policy. 106, 58-67, https://doi.org/10.1016/j.envsci.2020.01.013.

Kronvang, B., Andersen, H.E., Larsen, S.E., Audet, J., 2013. Importance of bank erosion for sediment input, storage and export at the catchment scale. Journal of Soils and Sediments. 13, 230–241. https://doi.org/10.1007/s11368-012-0597-7.

Line, D.E., Osmond, D.L., Childres, W., 2016. Effectiveness of livestock exclusion in a pasture of central north carolina. Journal of Environmental Quality. 45, 1926–1932. https://doi.org/10.2134/jeq2016.03.0089.

Longoni, L., Papini, M., Brambilla, D., Barazzetti, L., Roncoroni, F., Scaioni, M., Ivanov, V.I., 2016. Monitoring riverbank erosion in mountain catchments using terrestrial laser scanning. Remote Sensing. 8, 241. https://doi.org/10.3390/rs8030241.

Madden, D., Harrison, S., Finn, J.A., Ó hUallacháin, D., 2019. The impact of cattle drinking points on aquatic macroinvertebrates in streams in south-east Ireland. Irish Journal of Agricultural and Food Research. 58, 13-20.

McKergow, L.A., Weaver, D.M., Prosser, I.P., Grayson, R.B., Reed, A.E.G., 2003. Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, western australia. Journal of Hydrology. 270, 253–272. https://doi.org/10.1016/S0022-1694(02)00286-X.

Miller, J.J., Chanasyk, D.S., Curtis, T., Willms, W.D., 2010. Influence of streambank fencing on the environmental quality of Cattle-Excluded pastures. Journal of environmental quality. 39, 991–1000. https://doi.org/10.2134/jeq2009.0233.

Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Analytica Chimica Acta. 27, 31–36.

Myers, D.T., Rediske, R.R., McNair, J.N., 2019. Measuring streambank erosion: A comparison of Erosion pins, total station, and terrestrial laser scanner. Water. 11, 1846. https://doi.org/10.3390/w11091846.

Noe, G.B., Cashman, M.J., Skalak, K., Gellis, A., Hopkins, K.G., Moyer, D., Webber, J., Benthem, A., Maloney, K., Brakebill, J., Sekellick, A., Langland, M., Zhang, Q., Shenk, G., Keisman, J., Hupp, C., 2020. Sediment dynamics and implications for management: State of the science from long-term research in the chesapeake bay watershed, USA. Wiley interdisciplinary Reviews. Water. 7 https://doi.org/10.1002/wat2.1454.

O'Callaghan, P., Kelly-Quinn, M., Jennings, E., Antunes, P., O'Sullivan, M., Fenton, O., hUallacháin, D.Ó, 2019. The Environmental Impact of Cattle Access to Watercourses: A Review. Journal of Environmental Quality. 48, 340-351, **https://doi.org/10.2134/jeq2018.04.0167**.

O'Hagan, M.J.H., Matthews, D.I., Laird, C., McDowell, S.W.J., 2016. Herd-level risk factors for bovine tuberculosis and adoption of related biosecurity measures in northern ireland: A case-control study. Veterinary Journal. 213, 26–32. https://doi.org/10.1016/j.tvjl.2016.03.021.

Olsen, S.R., Cole, C.V., Watanabe, F.S., 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. US Department of Agriculture, Washington.

O'Sullivan, M., Ó hUallacháin, D., Antunes, P.O., Jennings, E., Kelly-Quinn, M., 2019. The impacts of cattle access points on deposited sediment levels in headwater streams in Ireland. River Research and Applications. 35, 146-158, .

Price, J.I., Renzetti, S., Dupont, D., Adamowicz, W., Emelko, M.B., 2017. Production costs, inefficiency, and source water quality: A stochastic Cost frontier analysis of canadian water utilities. Land Economics 93, 1–11. https://doi.org/10.3368/le.93.1.1.

Pulley, S., Cardenas, L.M., Grau, P., Mullan, S., Rivero, M.J., Collins, A.L., 2021. Does cattle and sheep grazing under best management significantly elevate sediment losses? evidence from the north wyke farm platform. UK. Journal of Soils and Sediments. 21, 1875–1889. https://doi.org/10.1007/s11368-021-02909-y.

Resop, J.P., Lehmann, L., Hession, W.C., 2019. Drone laser scanning for modeling riverscape topography and vegetation: Comparison with traditional aerial lidar. Drones 3, 35. https://doi.org/10.5194/hess-19-453-2015.

Rice, A.R., Cassidy, R., Jordan, P., Rogers, D., Arnscheidt, J., 2021. Fine-scale quantification of stream bank geomorphic volume loss caused by cattle access. Science of the Total Environment. 769, 144468 https://doi.org/10.1016/j.scitotenv.2020.144468.

Roberts, W.M., Stutter, M.I., Haygarth, P.M., 2012. Phosphorus retention and remobilization in vegetated buffer strips: A review. Journal of Environmental Quality. 41, 389–399. https://doi.org/10.2134/jeq2010.0543.

Schulte, R.P.O., Doody, D.G., Byrne, P., Cockerill, C., Carton, O.T., 2009. Lough melvin: Developing cost-effective measures to prevent phosphorus enrichment of a unique aquatic habitat. Tearmann: Irish journal of Agri-Environmental. Research 7, 211–228.

Schwendel, A., Fuller, I.C., Death, R.G., 2012. Assessing DEM interpolation methods for effective representation of upland stream morphology for rapid appraisal of bed stability. River Research and Applications. 30, 132–133. https://doi.org/10.1002/rra.1475.

Sherriff, S.C., Rowan, J.S., Melland, A.R., Jordan, P., Fenton, O., Ó hUallacháin, D., 2015. Identifying the controls of soil loss in agricultural catchments using ex situ turbidity-based suspended sediment monitoring. Hydrology and Earth System Sciences Discussions. 12, 2707-2740.

Thoma, D.P., Gupta, S.C., Bauer, M.E., Kirchoff, C.D., 2005. Airborne laser scanning for riverbank erosion assessment. Remote Sensing of Environment. 95, 493–501. https://doi.org/10.1016/j.rse.2005.01.012.

Thomas, E., Riley, M., Spees, J., 2019. Good farming beyond farmland – Riparian environments and the concept of the 'good farmer'. Journal of Rural Studies. 67, 111–119. https://doi.org/10.1016/j.jrurstud.2019.02.015.

Trimble, S.W., Mendel, A.C., 1995. The cow as a geomorphic agent — A critical review. Geomorphology 13, 233–253. https://doi.org/10.1016/0169-555X(95)00028-4.

Tufekcioglu, M., Isenhart, T.M., Schultz, R.C., Bear, D.A., Kovar, J.L., Russell, J.R., 2012. Stream bank erosion as a source of sediment and phosphorus in grazed pastures of the rathbun lake watershed in southern iowa, united states. Journal of Soil and Water Conservation. 67, 545–555. https://doi.org/10.2489/jswc.67.6.545.

Ukso, 2023. Soils of northern ireland - world reference base. accessed 20 June 2023. https://www.ukso.org/static-maps/soils-of-northern-ireland.html.

Van Zanten, B.T., Verburg, P.H., Ferrer Espinosa, M., Gomez-y-Paloma, S., Galimberti, G., Kantelhardt, J., Kapfer, M., Lefebvre, M., Manrique, R., Piorr, A., Raggi, M., Schaller, L., Targetti, S., Zasada, I., Viaggi, D., 2014. European agricultural landscapes, common agricultural policy and ecosystem services: a review. Agronomy for Sustainable Development. 34, 309–325. https://doi.org/10.1007/s13593-013-0183-4.

Vidon, P., Campbell, M.A., Gray, M., 2008. Unrestricted cattle access to streams and water quality in till landscape of the midwest. Agricultural Water Management. 95, 322–330. https://doi.org/10.1016/j.agwat.2007.10.017.

Zaimes, G.N., Schultz, R.C., Isenhart, T.M., 2008. Streambank soil and phosphorus losses under different riparian Land-Uses in iowa. Journal of the American Water Resources Association. 44, 935–947. https://doi.org/10.1111/j.1752-1688.2008.00210.x.

Zaimes, G.N., Tamparopoulos, A.E., Tufekcioglu, M., Schultz, R.C., 2021. Understanding stream bank erosion and deposition in iowa, USA: A seven-year study along streams in different regions with different riparian land-uses. Journal of Environmental Management. 287, 112352 https://doi.org/10.1016/j.jenvman.2021.112352.