Rapid Evidence Assessment of the Impacts of Organic Manures on the Water Quality of Rivers

Report No: 774

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Glossary

Adsorption/ sorption	The physical adherence or bonding of ions and molecules onto the surface of another phase/substance.
AD	Anaerobic digestion
Al	Aluminium
AP	Apatite inorganic phosphorus
Broadcasting	Surface spreading method of manure application
Broiler	A chicken bred and raised for meat production.
C	Carbon
Ca	Calcium
$CaCO_3$	Calcium carbonate
CEE	Collaboration for Environmental Evidence
CoAPR	The Control of Agricultural Pollution (Wales) Regulation (Welsh Government, 2021)
Digestate	The by-product of anaerobic digestion
DIP	Dissolved inorganic P
Disking	A sub-surface manure application method
DOP	Dissolved organic P
DRP	Dissolved reactive P
DPS	Degree of P saturation
d.w	Dry weight
EA	Environment Agency
EDTA	Ethylenediaminetetraacetic acid
EU	European Union
Fe	Iron
FGD	Flue gas desulfurization by-product
Fresh manure	Manure that has not been stored prior to land application
f.w	Fresh weight
GHG	Greenhouse gas
HAP	High available P
K	Potassium
Labile	Easily broken down/ bioavailable

Layer chicken	Chicken that are kept and raised for egg production
Mg	Magnesium
Mn	Manganese
MRP	Molybdate-reactive phosphorus
N	Nitrogen
NAIP	Non-apatite inorganic phosphorus
NaOH	Sodium hydroxide
NH_3	Ammonia
NH_4	Ammonium
NO_2	Nitrite
NO_3	Nitrate
NRW	Natural Resources Wales
OP	Orthophosphate (the most bioavailable form of inorganic P)
P	Phosphorus
PAC	Poly-aluminium chloride
Plab	Labile phosphorus forms
PNRP	Particulate non-reactive phosphorus
Poultry litter[1]	Associated with intensive poultry production, is likely to consist of a mixture of bedding material, feathers, poultry manure, urine and food particles.
Poultry manure[1]	Poultry manure is the organic waste material from poultry typically consisting of animal faeces only.
PP	Particulate phosphorus
PRP	Particulate reactive phosphorus
PSR	Soil P saturation rate
REA	Rapid evidence assessment
RQs	Research questions
Ruminant	Herbivores capable of extracting nutrients from plant-based foods by fermenting food via microbial processes in a specialised stomach chamber prior to digestion. This includes cows, sheep and goats.
S	Sulphur
Sludge	The by-product of wastewater treatment processes

[1] In some of the literature poultry litter and poultry manure are not defined or are used interchangeably. In these cases, the terminology used by the author has been reported.

Slurry	A slurry is a manure that has a high enough water content to contain a freely draining liquid that could infiltrate into the soil.
SNRP	Soluble non-reactive phosphorus
Sorption	The process by which a substance (sorbate) is sorbed (adsorbed or absorbed) on or in another substance (sorbent)
SRP	Soluble reactive phosphorus
TDP	Total dissolved phosphorus (dissolved organic and inorganic phosphorus)
TKN	Total Kjeldahl nitrogen
TN	Total nitrogen
TP	Total phosphorus (includes all forms of phosphorus in soluble and sediment bound phases)
TSS	Total suspended solids
TDP	Total dissolved P (dissolved organic and inorganic P)
UAN	Urea ammonium nitrate
UK	United Kingdom
USA	United States of America
WEP	Water extractable phosphorus
WSP	Water soluble phosphorus

Executive summary

The principal aim of this study was to improve Natural Resources Wales' understanding of how organic manures, particularly from the poultry sector, can impact the nutrient status of rivers, with a focus on phosphorus (P). The methodology used to produce this Rapid Evidence Assessment (REA) included a systematic review of relevant literature to establish an evidence base, which was used to address primary and secondary Research Questions (RQs). The scope of the study was restricted to answering the RQs only, and, as a REA, the literature searches and critical appraisal of the literature were not as comprehensive as a full systemic review. The key conclusions relating to each of the RQs are summarised below.

RQ1: Key factors determining the extent to which manures, spread on agricultural land, increase the concentration of bioavailable P in downstream freshwater environments.

Manure type, soil type, manure storage and manure spreading methods are some of the factors that can influence the loss of manure P to surface waters from agricultural land. It is, nevertheless, important to acknowledge that additional site-specific factors (including climate and catchment characteristics) also influence mobilisation and delivery of P to receiving waters.

RQ1a: How readily do the components of P from different manure types become converted to the bioavailable form on land and in water?

The form of P added to land is an important factor controlling mobilisation and bioavailability. Poultry litter/ manure and pig manure/ slurry likely represent a greater potential risk to surface water from leaching than dairy cattle manure and biosolids owing to the higher soluble P/ water extractable P content. Application of soluble P via litter, manure or slurry may directly contribute to soluble P in runoff.

Literature suggests that soil type can also influence both the amount and the form of P entering surface water bodies by directly influencing adsorption, mineral precipitation and leaching from the soil.

RQ1b: If soil type is an important variable, how do the soil types of Wales influence the processing and transport of manure P?

Welsh soils are predominantly podzols, brown soils and gley soils, and each have differing characteristics that may impact the transport and processing of manure P. Podzols typically have a subsurface layer containing iron and aluminium oxides which can contribute to the retention of P, whilst clayey Gley soils have high capacity to adsorb P. Brown soils are highly permeable which may promote the movement of nutrients to deeper soil layers and/or groundwater.

RQ1c: How do different methods of storing and spreading manure affect the rate of P loss from soils?

Wet storage may increase soluble concentrations of P in runoff from land to which manure is applied. The methods employed for spreading manures on land may also affect the extent and rate of P loss from soils. In particular, the type of spreading (surface or sub-surface), application rate and timing of manure application.

RQ2a: How does poultry litter/ manure that is collected from broiler units and then spread on land compare to manure added to land via free-ranging in terms of P processing in soil and loading to freshwaters?

Chemical composition of litter varies between free-ranging and intensive systems; higher total P has been reported in chicken litter from intensive systems which could therefore pose a higher risk to water quality downstream. Nevertheless, virtually no studies have investigated the impact of free-range poultry farms on downstream water quality.

RQ2b: How does digestate from anaerobic digestion (AD) plants using a poultry litter/manure feedstock compare to raw poultry manure in terms of P processing in soil and loading to freshwaters once it is spread on land?

The P in digestate (from AD) may be more accessible to plants, however, if not taken up quickly may be more susceptible to loss in runoff or leachate. Using digestate over raw manure/ litter offers the potential benefit of being more economically viable to transport.

Crynodeb gweithredol

Prif nod yr astudiaeth hon oedd gwella dealltwriaeth Cyfoeth Naturiol Cymru o sut y gall tail organig, yn enwedig o'r sector dofednod, effeithio ar statws maetholion afonydd, gan ganolbwyntio ar ffosfforws. Roedd y fethodoleg a ddefnyddiwyd i gynhyrchu'r asesiad cyflym hwn o'r dystiolaeth yn cynnwys adolygiad systematig o lenyddiaeth berthnasol i sefydlu sylfaen dystiolaeth, a ddefnyddiwyd i fynd i'r afael â phrif gwestiynau ymchwil a chwestiynau ymchwil eilaidd. Cyfyngwyd cwmpas yr astudiaeth i ateb y cwestiynau ymchwil yn unig, ac, fel asesiad cyflym o'r dystiolaeth, nid oedd y chwiliadau llenyddiaeth a'r gwerthusiad beirniadol o'r lenyddiaeth mor gynhwysfawr ag adolygiad systemig llawn. Mae'r casgliadau allweddol sy'n ymwneud â phob un o'r cwestiynau ymchwil wedi'u crynhoi isod.

Cwestiwn ymchwil 1: Beth yw'r prif ffactorau sy'n pennu i ba raddau y mae tail, sy'n cael ei daenu ar dir amaethyddol, yn cynyddu crynodiad ffosfforws sydd ar gael yn fiolegol mewn amgylcheddau dŵr croyw i lawr yr afon?

Ymhlith y ffactorau a all ddylanwadu ar ollwng ffosfforws tail i ddyfroedd wyneb o dir amaethyddol y mae'r math o dail, y math o bridd, dulliau storio tail a dulliau taenu tail. Serch hynny, mae'n bwysig cydnabod bod ffactorau ychwanegol safle-benodol (gan gynnwys hinsawdd a nodweddion y dalgylch) hefyd yn dylanwadu ar symudiad a danfoniad ffosfforws i ddyfroedd derbyn.

Cwestiwn ymchwil 1a: Pa mor hawdd y mae cyfansoddion ffosfforws o wahanol fathau o dail yn cael eu trosi i'r ffurf sydd ar gael yn fiolegol ar dir ac mewn dŵr?

Mae ffurf ffosfforws sy'n cael ei ychwanegu at dir yn ffactor pwysig sy'n rheoli symudiad a bio-argaeledd. Mae gwasarn / tail dofednod a thail / slyri moch yn debygol o fod yn fwy o risg bosibl i ddŵr wyneb o drwytholchi na thail gwartheg godro a biosolidau oherwydd y cynnwys ffosfforws hydawdd uwch / ffosfforws y gellir ei echdynnu o ddŵr. Gall gwasgaru ffosfforws hydawdd trwy wasarn, tail neu slyri gyfrannu'n uniongyrchol at ffosfforws hydawdd mewn dŵr ffo.

Mae deunydd ysgrifenedig yn awgrymu y gall y math o bridd hefyd ddylanwadu ar faint a ffurf y ffosfforws sy'n mynd i mewn i gyrff dŵr wyneb trwy ddylanwadu'n uniongyrchol ar arsugniad, gwaddodiad mwynau a thrwytholchi o'r pridd.

Cwestiwn ymchwil 1b: Os yw'r math o bridd yn newidyn pwysig, sut mae'r mathau o bridd yng Nghymru yn dylanwadu ar brosesu a chludo tail ffosfforws?

Podsolau, priddoedd brown a chleiau glas yw priddoedd Cymru yn bennaf, ac mae gan bob un ohonynt nodweddion gwahanol a allai effeithio ar gludo a phrosesu ffosfforws tail. Yn nodweddiadol mae gan bodsolau haen o dan yr wyneb sy'n cynnwys ocsidau haearn ac alwminiwm a all gyfrannu at gadw ffosfforws, tra bod cleiau glas yn gallu amsugno ffosfforws yn dda. Mae priddoedd brown yn hynod athraidd a gall hyn hybu mudiad maetholion i haenau pridd dyfnach a/neu ddŵr daear.

Cwestiwn ymchwil 1c: Sut mae gwahanol ddulliau o storio a thaenu tail yn effeithio ar gyfradd colli ffosfforws o briddoedd?

Gall storio gwlyb gynyddu crynodiadau hydawdd o ffosfforws mewn dŵr ffo o dir y rhoddir tail arno. Gall y dulliau a ddefnyddir i daenu tail ar dir hefyd effeithio ar faint o ffosfforws a gollir o briddoedd, ac ar ba raddfa, yn arbennig y math o daenu (ar yr wyneb neu o dan yr wyneb), y gyfradd daenu a'i amseriad.

Cwestiwn ymchwil 2a: Sut mae gwasarn / tail dofednod sy'n cael ei gasglu o unedau brwyliaid ac yna'n cael ei daenu ar dir yn cymharu â thail sy'n cael ei ychwanegu at y tir drwy gadw'r dofednod yn rhydd o ran prosesu ffosfforws mewn pridd a'i lwytho i ddŵr croyw?

Mae cyfansoddiad cemegol gwasarn yn amrywio rhwng systemau rhydd a systemau dwys. Adroddwyd am gyfanswm uwch o ffosfforws mewn gwasarn ieir o systemau dwys a allai felly achosi risg uwch i ansawdd dŵr i lawr yr afon. Serch hynny, nid oes fawr ddim astudiaethau wedi ymchwilio i effaith ffermydd dofednod maes ar ansawdd dŵr i lawr yr afon.

Cwestiwn ymchwil 2b: Sut mae gweddillion o weithfeydd treuliad anaerobig sy'n defnyddio gwasarn dofednod / tail porthiant yn cymharu â thail dofednod amrwd o ran prosesu ffosfforws mewn pridd a'i lwytho i ddŵr croyw unwaith y caiff ei daenu ar y tir?

Mae'n bosibl y bydd y ffosfforws mewn gweddillion treuliad anaerobig yn fwy hygyrch i blanhigion, ond os na chaiff ei amsugno'n gyflym gall gael ei golli'n haws i ddŵr ffo neu drwytholch. Mae'n bosib y byddai'n fwy hyfyw yn economaidd i gludo gweddillion treuliad anaerobig ar draul tail / gwasarn crai.

1. Introduction

Phosphorus (P) is essential for food production and is often a limiting factor in crop yields. Inorganic mineral P is, however, considered a finite resource which is estimated to become depleted over the coming decades (Bloem et al., 2017; Vučić and Muller, 2021). The United Kingdom (UK) is reliant on imported P and does not have indigenous mineral phosphate reserves, which poses a significant risk to food security (O'Donnell et al., 2021; Rothwell et al., 2022). Despite relying on imports, only 43% of imported UK food system P is converted into food and exportable goods, with manure P representing a key driver of system surplus and inefficiency (Rothwell et al., 2022).

The application of organic manures to agricultural soil can provide several benefits by increasing organic matter and valuable nutrients (soil fertility), improving soil structure, increasing water holding capacity and stimulating soil microbial communities (Epelde et al., 2018). However, organic manures may also result in elevated concentrations of P in runoff and leachate following application to agricultural land (diffuse pollution) or from slurry/ manure storage (point source pollution) which contribute to the eutrophication of surface waters (Hooda et al., 2000; Kleinman et al., 2005). P loadings to freshwaters in the UK represents a major water quality issue; in England and Wales, P is the largest single contributor to poor ecological status in rivers and lakes (Environment Agency (EA), 2023). In Wales, 61% of rivers classified as Special Areas of Conservation recently failed their P targets (Natural Resources Wales (NRW), 2022).

Over 90 million tonnes of housed livestock manures are produced every year in the UK (Bateman et al., 2011). Livestock systems in the UK can largely be described as pasture-based systems (cattle and sheep) or indoor systems (pigs and poultry) (Hooda et al., 2000). Storage and weather considerations often determine the timing and rates of application of organic manures, rather than crop-specific nutrient requirements, which at times has resulted in quantities of farmyard manure and slurry being applied to soils that far exceed crop requirements (Hooda et al., 2000). The Control of Agricultural Pollution (Wales) Regulation (CoAPR) (Welsh Government, 2021) was brought in, in part, to help address the issue of over-application.

Agricultural land accounts for approximately 90% of the total land area of Wales (Welsh Government, 2020), with the majority of land dominated by grasslands and rough grazing and used for rearing sheep and cattle (Welsh Government, 2022). Source apportionment modelling on the upper River Wye catchment suggests that rural land use contribution to the average daily P load is 72% (Dŵr Cymru Welsh Water, 2023). Modelling of the Wye catchment found that ~6,500 t/yr of P are imported to the catchment, whereas ~3,100 t/yr of P are exported, providing an overall P catchment use efficiency of 48% (Withers et al., 2022). The largest P import into the catchment was in livestock feed (~5,000 t P/yr) and the largest internal flow of P is in livestock manure (~6,100 t P/y) (Withers et al., 2022).

Increased P entering water bodies via runoff promotes the excessive growth of phytoplankton or macroalgae which can result in eutrophication. However, recent evidence suggests that increased P loadings to surface waters may also impact water quality in other ways. P may stimulate aquatic bacterial growth which may increase the biological oxygen demand contributing to hypoxia. Furthermore, stimulation of faecal bacterial growth may have risks to human health through direct (e.g., bathing water) or indirect (e.g., shellfish consumption) exposure (Malin and Cahoon, 2020).

Objectives

The principal aim of this Rapid Evidence Assessment (REA) was to improve NRW’s understanding of how organic manures, particularly from the poultry sector, can impact the nutrient status of rivers, with a focus on P. A systematic review of relevant literature was undertaken to establish an evidence base, which was used to address primary and secondary Research Questions (RQs).

The primary research question (RQ1) was to identify the key factors determining the extent to which organic manures, spread on agricultural land, increase the concentration of bioavailable P in downstream freshwater environments, addressing several specific points:

- How readily do the components of P from different manure types become converted to the bioavailable form on land and in water? (RQ1a)
- If soil type is an important variable, how do the soil types of Wales influence the processing and transport of manure P? (RQ1b)
- How do different methods of storing and spreading manure affect the rate of P loss from soils? (RQ1c)

The secondary research questions specifically relate to poultry farming:

- How does poultry litter/ manure that is collected from broiler units and then spread on land compare to manure added to land via free-ranging in terms of P processing in soil and loading to freshwaters? (RQ2a)
- How does digestate from anaerobic digestion (AD) plants using a poultry litter/ manure feedstock compare to raw poultry manure in terms of P processing in soil and loading to freshwaters once it is spread on land? (RQ2b)

A comprehensive assessment of factors influencing bioavailable P other than manure type, soil type and manure storage and spreading methods, as well as mitigation measures to reduce P loading to surface waters, was beyond the scope of this review. Whilst P pollution represents the primary focus of the REA, information relevant to the above research questions for nitrogen (N) has also been noted where relevent.

Structure of the report

This report is set out in the following way:

- **Chapter 2** – describes the methodology for this REA.
- **Chapter 3** – presents the results, principally a summary of key information extracted from the literature reviewed for each manure type investigated.
- **Chapter 4** – discusses the literature review findings in the context of the RQs.
- **Chapter 5** – provides a summary and conclusions.
- **Chapter 6** – highlights recommendations.

2. Methodology

Rapid Evidence Assessment

An REA was undertaken following the methods described by Collins et al. (2015). The methodology followed a six-stage process, as shown in Figure 1, with key steps described below.

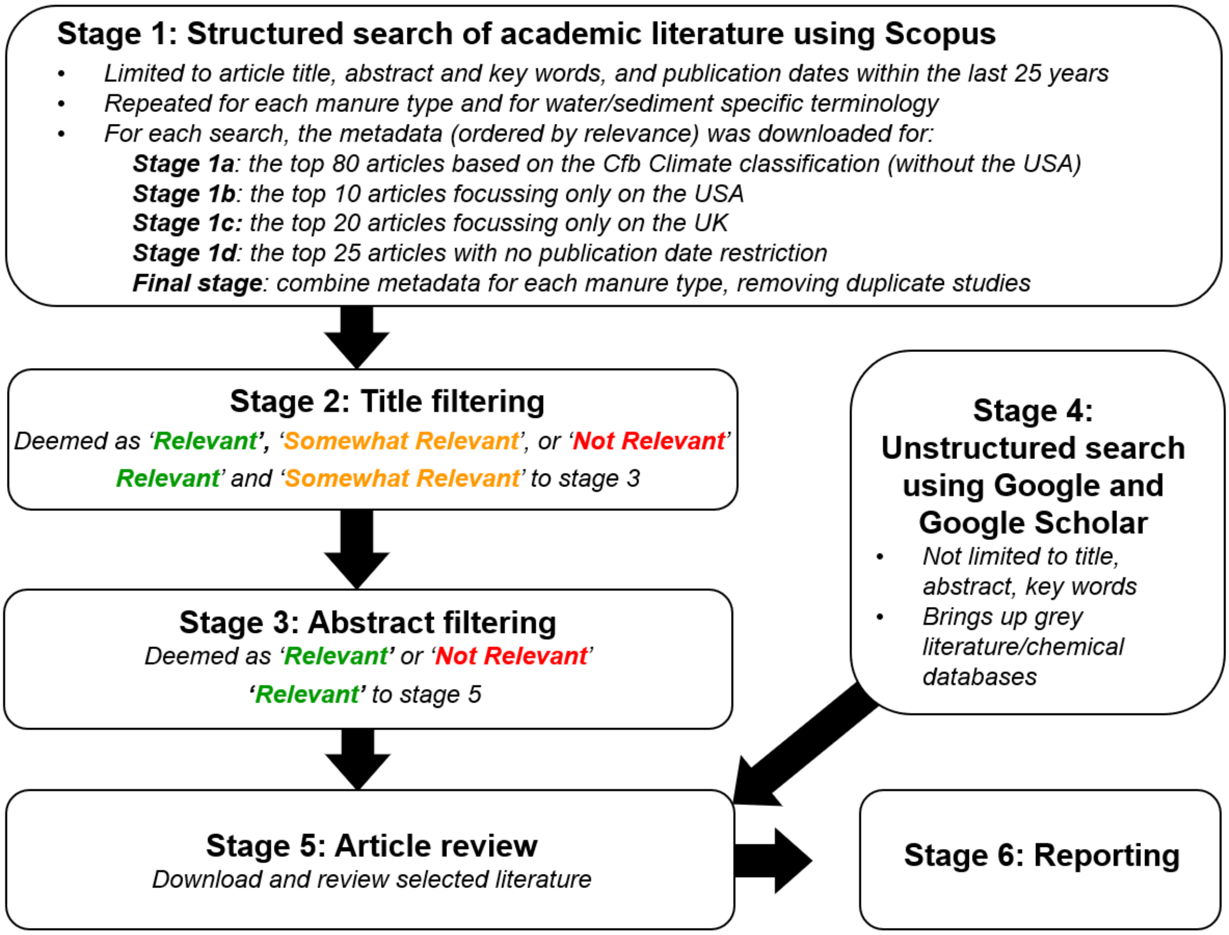

Figure 1. Methodology schematic

Structured searches

Structured searches of the academic literature were conducted using the search engine Scopus. Scopus was selected over other search engines as it provides the option to easily export metadata to csv format and does not limit the number of Boolean commands applied to a search. Search terms were carefully constructed to only obtain the most relevant literature by limiting the breadth, depth and comprehensiveness of the search.

Search strings were designed based on the adapted PICO framework (i.e., including terminology for the Population, Intervention, Comparator, and Outcomes) (Collaboration for Environmental Evidence (CEE), 2013; James et al., 2016), but tailored to meet to the objectives of this project (and without the comparator element).

Two search strings (A and B) were designed to obtain relevant literature to address the primary questions. A summary of the key words/ terminology included within the searches and a description of how this terminology was combined to form the two search strings is presented in Figure 2. The two search strings target different areas:

- (A) Aimed to identify key factors determining the extent to which the manures impact bioavailable P concentrations in downstream freshwater environments (composed of parts 1 to 5, Figure 2).
- (B) Aimed to identify the role of soil types on the fate and behaviour of P (composed of parts 1-4 and 6, Figure 2).

Both search strings were repeated for each manure type/ category. In brief, the search string terminology fits into the PICO framework as follows:

- Population: manure types (and soil-types terminology for search string B).
- Intervention: spreading/ application of manure to agricultural land.
- Comparator: compares results against a baseline/ null hypothesis (not applicable to this research question).
- Outcomes: impacts of manure derived P pollution on water quality (search string A) and the impact of soil processes on the fate and behaviour of manure derived P pollution (search string B).

Searches were limited to the article title, abstract and keywords and restricted to the last 25 years (1999 – 2023) to ensure the most relevant information was captured. Additional inclusion and exclusion criteria were applied to further refine the results based on geographic scope, relevant subject areas and English language, as summarised in Figure 1 (Stage 1).

The geographic scope was defined as countries with the same bioclimatic classification as the UK, as described by the Köppen-Geiger Climate Classification system (Climate Change & Infectious Diseases Group, 2023). The UK is classed as Cfb (C: warm temperate, f: fully humid and b: warm summers). The top 80 most relevant studies were downloaded per search term.

As only very small parts of the United States of America (USA) fall in this bioclimatic category (Cfb; and as the USA produces a disproportionately large number of studies in this field), the USA was not included in the initial search. In order to capture the most relevant studies from the USA, the top 10 USA-specific studies (ordered by relevance) were also included. Additionally, selected key studies from the USA that were referenced within the primary literature were included within the review.

Furthermore, the top 25 most relevant articles (not limited by publication date) were downloaded for each manure type and search term, and duplicates removed with the original list. This step ensures that the most relevant articles are obtained, even if they are older than 25 years.

A final search was conducted that focused on UK-specific literature, as this is the primary focus of the review, following which the top 20 articles (ordered by relevance) were selected.

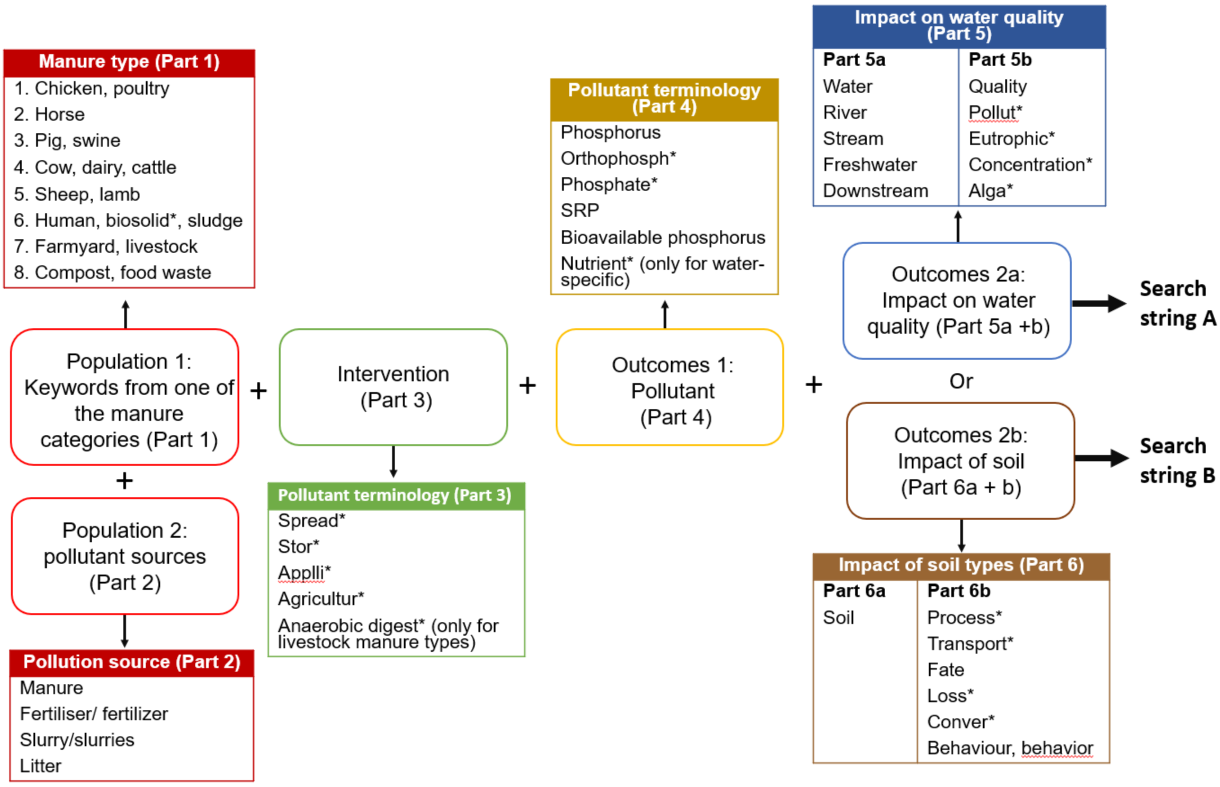

Figure 2. Search string construction

Unstructured searches

Additional unstructured searches were conducted using Google Scholar, providing a sense check that the most relevant literature was obtained in the structured review, and providing an opportunity to identify any relevant academic articles that may have been missed (e.g., where the search terms occur in the main body of the article, but not in the title, abstract or keywords). Searches were also conducted for relevant grey literature using Google.

Filtering for relevance

Literature identified from the structured searches (see 'Structured searches' section) was filtered for relevance following a two-stage process, as described in Figure 1. The relevance of each article was initially reviewed based on the title, and designated a category of 'relevant', 'somewhat relevant', or 'not relevant'. Following this, the second stage of the filtering process reviewed the abstracts of those articles deemed 'relevant' or 'somewhat relevant' at the title filtering stage into 'relevant' or 'not relevant'. The assessment of relevance was checked by an independent checker. Relevance was determined based on two key criteria:

- Geographical/ climatic relevance: studies in areas that are not geographically/ climatically comparable with the UK were not included (e.g., China, South America and countries with tropical climates)[2].
- Relevance of content to the primary or secondary research questions.

A summary of the number of articles produced per search and those taken forward as part of the filtering process are presented in Table 1. Spreadsheets containing the full article list downloaded from Scopus and the outcomes of the 2-stage filtering process (described above) are provided in Appendix A.

Table 1. The number of articles taken forward at each stage of the filtering process, for each manure type[3]

Organic manure type	Water-specific Total	Soil-specific Total	Combined and duplicates removed	Title filtering	Abstract filtering	Articles reviewed (+ additions from unstructured searches)
1. Poultry	57	25	63	51	43	37[4] (+13)
2. Pig	73	47	101	82	61	46
3. Sheep	62	45	89	55	32	30
4. Cow	99	72	151	121	75	33 (+2)
5. Horse[5]	8	3	9	5	4	4
6. Compost (food waste / green waste)	100	40	128	75	36	30
7. Biosolids	112	84	177	120	69	30
8. Farmyard/ livestock	95	59	139	89	51	30
Total	**606**	**375**	**857**	**598**	**371**	**255**[6]

[2] Note, articles reporting on studies in the USA (and selected others) outside of the Cfb climatic zones were marked as 'relevant' in the abstract filtering stage where deemed of relevance to the objectives of this review.
[3] For the title filtering stage, this number corresponds to articles deemed 'relevant' or 'somewhat relevant', and for the abstract filtering stage this number refers to papers deemed 'relevant'.
[4] 6 articles could not be accessed and thus could not be reviewed.
[5] None of the articles for horse manure, when reviewed, included information deemed relevant for inclusion, except nutrient concentrations (for horse manure) in Pagliari and Laboski (2012) which are included in Appendix B.
[6] Total includes duplicates i.e. some articles were identified for more than one manure type. It was, in fact, relatively common for studies to investigate more than one manure type.

Article review

All articles from the structured searches deemed ‘relevant’ for poultry (and that could be accessed), and a minimum of 30 (as agreed with NRW) of the articles deemed relevant at the abstract filtering stage for the 'other’ manure types were reviewed[7].

Selected information deemed relevant to the primary or secondary research questions was extracted and presented within this document. Typically, information extracted from studies outside of the geographical/ climatic relevance defined (such as the USA) was only included in reporting when novel (as well as relevant to the research question(s)).

In some cases, literature cited (and of potential relevance) in the articles reviewed (and so on) were also reviewed and selected relevant information extracted[8]. Where information associated with a cited source has been included in this report, but the original article has not been reviewed, the citation, for example, Chang et al. (1983) in Quilbé et al. (2005), is included.

Further details of selected key relevant articles referred to in this report have been included in Appendix C.

Limitations of literature review

A limitation of using search terms is that potentially relevant literature may not be picked up since the search terms are strictly defined. However, additional unstructured searches were completed to mitigate this.

Findings included within this report are limited to selected information that was deemed relevant to the primary or secondary research questions. Owing to the breadth of information available and additional complexities to be considered, particularly for biosolids and green waste/ food waste, decisions needed to be made based on professional judgement as to the information reported on. For example, as best as possible, information associated with mixed manures (e.g. biosolids + pig slurry) has not been reported on to better facilitate comparisons between manure types (for RQ1).

A comprehensive assessment of factors influencing bioavailable P other than manure type, soil type and manure storage and spreading methods, as well as mitigation measures to reduce P loading to surface waters was beyond the scope of this review, though select information was extracted and included in ‘supplemental’ results sections where deemed relevant.

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[7] For ‘sheep’ only 29 (of the 32) articles deemed ‘relevant’ at the abstract filtering stage could be accessed, thus one article cited in the literature reviewed (that was deemed relevant) was reviewed to ensure the required minimum (of 30) was met.

[8] These articles were not included in Appendix A, except for ‘sheep’ as one cited article was included in the 30 (for the reason described above).

Academic interviews

Three interviews were held with academic experts working in this field to discuss recent advances in research and to discuss preliminary findings from the review. The choice of academics was informed by the literature review process and input from NRW, and included:

- Professor Philip Haygarth (Lancaster University, UK),
- Dr Shane Rothwell (Lancaster University, UK), and
- Professor Rishi Prasad (Auburn University, USA).

3. Results

Information extracted from the literature reviewed has been summarised by manure type:

- Poultry litter and manure
- Cow manure and slurry
- Pig manure/slurry
- Sheep manure
- Biosolids
- Green waste and food waste

Information associated with each of the following seven categories is included within the results sections. The first five categories represent the focus of the review, and the last two provide supplemental information[9]:

- Introduction
- Typical nutrient concentrations[10]
- Nutrient transformations, accumulation and transportation
- Soil characteristics
- Spreading and storage methods
- Other factors impacting bioavailable P downstream from manure sources
- Mitigations measures to reduce P transport to freshwater

Forms of phosphorous

P naturally exists in a variety of oxidised forms including dissolved and particulate-bound (adsorbed) pools of organic and inorganic P. The river P limits set by regulatory bodies are as concentrations of orthophosphate (OP) which is a bioavailable form of P (thus readily used by algae and microbes). Orthophosphate is analogous to soluble reactive P (SRP), dissolved inorganic P (DIP) and dissolved reactive P (DRP).

Dissolved P also exists in the organic form (dissolved organic P, or DOP), which may or may not be bioavailable (Mallin and Choon, 2020). A substantial proportion of total phosphorus (TP) present in soil solution and leachates can be in the form of DOP. DOP may be more mobile than inorganic OP and thus it may be an important P source for surface water eutrophication (Chardon et al., 1997).

P can bind to soil, and is therefore usually lost via surface runoff during periods of soil saturation and rainfall. P transport in surface runoff occurs in both the soluble and particulate forms (Sharpley and Menzel, 1987). Information about the different forms of P can be used to predict properties such as the movement, bioavailability, and transformations of P in agricultural soils, and the potential impact of P on water quality

[9] Note, a comprehensive assessment of factors influencing bioavailable P other than manure and soil type, as well as mitigation measures to reduce P loading to surface waters was beyond the scope of this review.
[10] A comparison table, which includes selected studies from those reviewed that reported data for more than one of the manure types investigated in this study, is provided in Appendix B.

(Ahmed et al., 2019; Yang et al., 2013). A summary of the manure and soil P pools, and the transformations between these pools is presented in Figure 3.

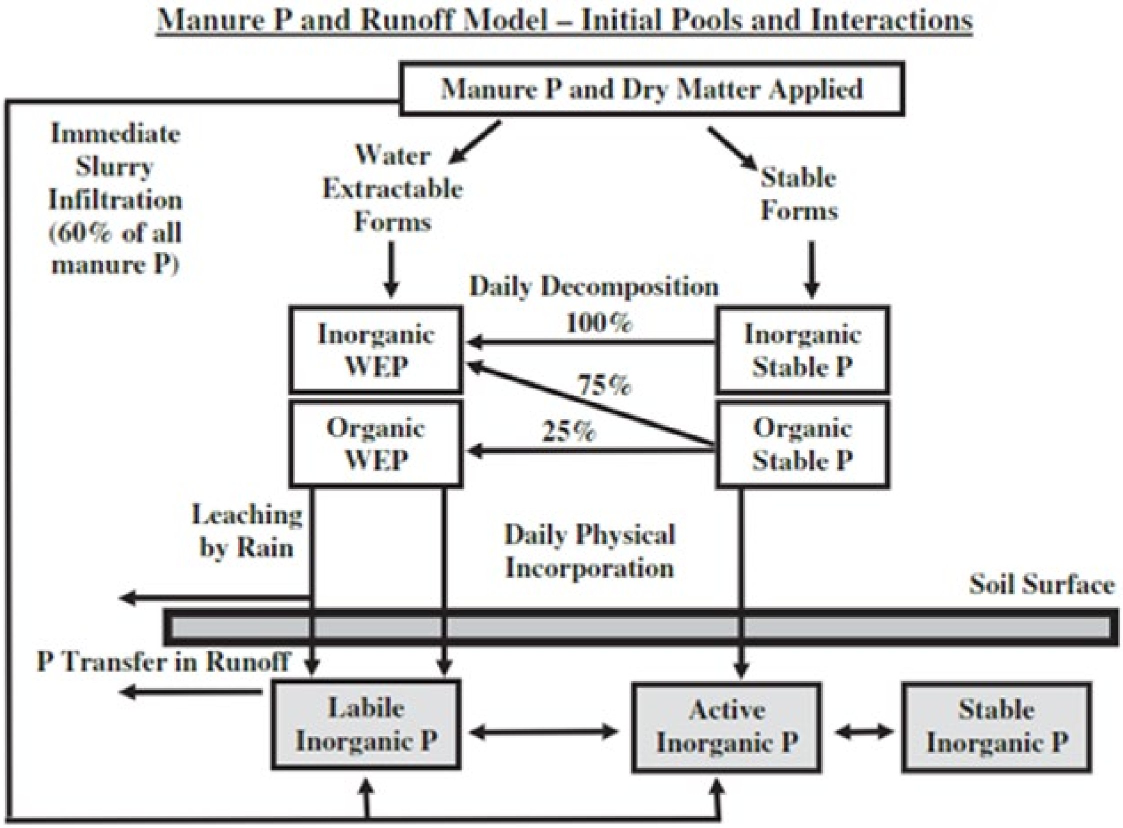


Figure 3. Schematic depiction of manure and soil phosphorus pools, and transformation pathways between these pools (Vadas et al., 2007)

Poultry litter and manure

Introduction

Poultry waste varies in composition due to the varied nature of different farming practices and systems (Stiles, 2017). Poultry manure is the organic waste material from poultry typically consisting of animal faeces only. Poultry litter is associated with intensive poultry production and is likely to consist of a mixture of bedding material (wood shavings, straw or paper), feathers, poultry manure, urine and food particles (Sustainable Energy Ireland, 2003; Stiles, 2017). Poultry litter or manure may be composted before being applied as a fertiliser. This may simply involve creating a compost pile and allowing a period of time for the decay process, or might involve mixing with other materials (such as hay or wood chips) and the addition of a microbial innoculant to the mixture to enhance the composting process (as reported in Sharpley and Moyer, 2000).

Application to land is the primary waste management strategy for poultry litter/ manure. Poultry litter has a relatively low bulk density (< 500 kg/m^3) (Bernhart et al., 2009) making it economically difficult to transport over large distances (albeit easier than pig/ cow slurries with lower dry matter contents).

When poultry litter is spread on agricultural land based on N agronomic rates, P is usually applied in excess of the plants nutrient requirements resulting in P accumulation within the soil (Sims et al., 2000). It is this build-up of P (and subsequent transport to waterbodies) that poses the main environmental risk to surface waters (e.g., eutrophication) associated with the application of poultry litter/ manure (Szogi et al., 2009).

The availability of poultry manure is growing rapidly, particularly in Wales, making its usage in agricultural production more commonplace (Stiles, 2017). Withers et al. (2022) reported that poultry has now overtaken cattle as the main producer of manure P in the Wye catchment as a whole (which spans across parts of England and Wales). Current (2023) estimates for total poultry numbers in Wales stands at over 10.3 million with the majority being broiler chickens (5.1 million) or layers (4.5 million) (Welsh Government, 2023). Daily manure production per animal varies between 0.12 litres (layers) and 0.6 litres (broilers) (CoAPR, 2021). Annual manure production for the total number of poultry in Wales (2023) is therefore around 345 million litres, which equates to around 3,500 tons of P per year and 4,500 tons of N per year.

Typical nutrient concentrations

Tables 2-5 show variations in the concentrations of P, N and dry matter within poultry manure and poultry litter from the literature[11].

Variation may be ascribed to differences in diet, dietary supplements, litter type, handling and storage operations (Sims and Wolf, 1994 in Nicholson et al., 1996). In the study by Nicholson et al. (1996), 121 poultry manure samples were collected from commercial

[11] Only data from selected articles reviewed, and selected forms of P and N, has been included in the tables.

holdings in England and Wales and nutrient contents of the individual manure samples varied widely. Nevertheless, all poultry manure types (classified based on stock type and management system) had similar nutrient concentrations based on dry weights. Linear relationships between poultry manure dry matter contents and P, as well as total nitrogen (TN), content were found in this study; which suggests that dry matter content, rather than the manure type, is the most important factor controlling total nutrient concentrations expressed on a fresh weight basis.

Kacprzak et al. (2023) state that P in poultry manure is mainly inorganic (32 – 84%). Guo et al. (2009) report that 12 – 20% of P in poultry manure is in water-soluble form, which is highly susceptible to removal via runoff following application. Poultry manure also contains more stable P forms (22 – 58% of TP) compared to other animal manures (Dail et al., 2007).

Approximately 60% of the N is in a readily available form (ammonium (NH_4) plus uric N) which can be readily nitrified (Shepherd and Bhogal, 1998). N content of poultry manure is in the form uric acid (40–70%), feed protein (10–40%), urea (4–12%) and NH_4 (4–20%) and varies depending on animal diet, type and age (Dróżdż et al., 2020).

Table 2. Concentrations of P, N and dry matter in poultry manure from the literature[11]

Nutrient	Concentration (g/kg)	Reference
Dry matter (%)	35.00 (layer manure)	Nicholson et al. (1996)
Dry matter (%)	57.90 (free range layer)	Nicholson et al. (1996)
Dry matter (%)	42.00 – 72.00	Zahan and Othman (2018)
Dry matter (%)	40.00 (layers)	Wheeler et al. (2010)
TP	23.60 (dry weight: d.w)	Siddique and Robinson (2003)
TP	27.83	Sharpley and Moyer (2000)
TP	23.60	Montgomery et al. (2005)
TP	55.10 (d.w.)	Wheeler et al. (2010)
TP	9.00 (free range layer fresh weight (f.w))	Nicholson et al. (1996)
TP	7.00 (layer – deep pit collection f.w)	Nicholson et al. (1996)
TP	5.00 (layer – belt scrape system f.w)	Nicholson et al. (1996)
WSP	4.00	Montgomery et al. (2005)
WSP	6.60 (d.w)	Siddique and Robinson (2003)
WSP	7.30	Sharpley and Moyer (2000)
WSP	2.68 – 5.75 (layers)	Brandt et al (2004)
WSP	2.19 – 4.67 (broilers)	Brandt et al (2004)
Inorganic P	24.25	Sharpley and Moyer (2000)
Organic P	2.93	Sharpley and Moyer (2000)
Residual P	0.47	Sharpley and Moyer (2000)
TN	54.80	Montgomery et al. (2005)
TN	50.20	Sharpley and Moyer (2000)
TN	49.30 (d.w)	Wheeler et al. (2010)
TN	52.00 (d.w)	Siddique and Robinson (2003)
TN	34.00 (free range layer f.w)	Nicholson et al. (1996)
TN	21.00 (layer – deep pit collection f.w)	Nicholson et al. (1996)
TN	17.00 (layer – belt scrape system f.w)	Nicholson et al. (1996)
NH_4-N	9.50 (d.w)	Wheeler et al. (2010)
Organic N	39.80 (d.w)	Wheeler et al. (2010)

WSP = Water Soluble Phosphorus

Table 3. Concentrations of P, N and dry matter in poultry litter from the literature[11]

Nutrient	Concentration (g/kg)	Reference
Dry matter (%)	64.20 (broiler litters)	Nicholson et al. (1996)
TP	14.57	Eldridge et al. (2009)
TP	17.60	Spargo et al. (2005)
TP	11.30 (broiler)	Saurer et al. (1999)
TP	19.40	Kibet et al. (2011)
TP	16.25	Sharpley and Moyer (2000)
TP	14.30 ± 0.65 (chicks)	Hill and Cade-Nebum (2009)
TP	17.90 ± 11.2 (chicks)	Hill and Cade-Nebum (2009)
TP	11.00 (broiler f.w)	Nicholson et al. (1996)
TP	13.40 (d.w)	Siddique and Robinson (2003)
Inorganic P	14.60	Sharpley and Moyer (2000)
SRP	1.80 (broiler)	Saurer et al. (1999)
OP[12]	3.87 (d.w) (chicks)	Hill and Cade-Nebum (2009)
Pyrophosphate[Error! Bookmark not defined.]	0.007 (d.w) (chicks)	Hill and Cade-Nebum (2009)
Polyphosphate[Error! Bookmark not defined.]	0.01 (d.w) (chicks)	Hill and Cade-Nebum (2009)
Organic P	1.48	Sharpley and Moyer (2000)
Organic P	6.81 ± 0.74 (chicks)[13]	Hill and Cade-Nebum (2009)
Organic P	3.78 ± 1.17 (chicks)[Error! Bookmark not defined.]	Hill and Cade-Nebum (2009)
Residual P	0.24	Sharpley and Moyer (2000)
WEP	7.20	Kibet et al. (2011)
WEP	4.06	Sharpley and Moyer (2000)
WSP	3.00 (d.w)	Siddique and Robinson (2003)
Bicarbonate ext. P	4.30	Eldridge et al. (2009)
TN	37.50 (broiler)	Saurer et al. (1999)
TN	38.80	Sharpley and Moyer (2000)
TN	33.00 (broiler f.w)	Nicholson et al. (1996)
TN	41.60 (d.w)	Siddique and Robinson (2003)
Nitrite as N	7.30	Eldridge et al. (2009)
Nitrate as N	4.90	Eldridge et al. (2009)

WEP = Water Extractable Phosphorus

Table 4. Concentrations of P and N in poultry manure compost from the literature[11]

Nutrient	Concentration (g/kg)	Reference
TP	9.03	Sharpley and Moyer (2000)
Inorganic P	8.26	Sharpley and Moyer (2000)
Organic P	1.08	Sharpley and Moyer (2000)
WEP	2.11	Sharpley and Moyer (2000)
TN	14.20	Sharpley and Moyer (2000)

[12] Inorganic P compounds

[13] Other organic P constituents are presented in the paper.

Table 5. Concentrations of P in poultry litter compost from the literature[11]

Nutrient	Concentration (g/kg)	Reference
TP TP	12.60 ± 1.80 18.5 ± 0.05	Hill and Cade-Nebum (2009) Hill and Cade-Nebum (2009)
Organic P Organic P	1.94 ± 12.50 2.66 ± 1.09	Hill and Cade-Nebum (2009) Hill and Cade-Nebum (2009)

Nutrient transformations, accumulation and transportation

The application of poultry litter to agricultural land has been recognised as a contributing source of non-point source pollution in the form of P and N to downstream waters through surface pathways (Hoover et al., 2019).

Research suggests that most of the TP in runoff from soils amended with poultry litter is primarily in the dissolved form, and elevated concentrations of dissolved P can persist in runoff for more than a year following applications (Eldridge et al., 2009). A significant positive correlation was found between poultry litter soluble P and SRP concentrations in runoff water (DeLaune et al., 2004).

A laboratory study, looking at leaching of nutrients from different manures and slurries during five simulated rainfall events found the highest DIP and DOP concentrations in the leachate were from poultry manure and swine slurry, compared with dairy cattle manure (Sharpley and Moyer, 2000).

A field-scale plot study (southwest Virginia USA) found that surface-applied poultry litter produced the greatest TP and DRP losses compared to surface applied and incorporated dairy manure[14]. Surface application of dairy manure resulted in 25 – 50% lower runoff volumes when compared to surface applied poultry litter; the mulching effect of the solids in the dairy manure was considered a likely contributing factor (Mishra et al., 2006).

Findings from the study by Bos et al. (2021) suggest that poultry litter application rates in excess of crop P requirements result in accumulation of P within fields and increased DRP in runoff. These authors suggest that the risk of downstream DRP loss in the landscape investigated (Texas Blackland Prairie ecoregion) increased under conditions that promoted enhanced transport (i.e., hydrological connectivity) and source factors (i.e., poultry litter applications, greater area with high rates of poultry litter application).

Nevertheless, not all studies have concluded that poultry litter is a risk. Smith et al. (2007) found that poultry litter amendments represented little risk to water quality and attributed this to the immobilisation of soluble P through the stimulation of microbial biomass.

[14] Following simulated rainfall events representing 2- to 10-year storms in southwest Virginia, occurring 1- and 2-days following manure application. The actual amount of P applied in the poultry litter treatment was about 1.5 times higher than the P applied in dairy manure treatments.

Siddique and Robinson (2003) report that soil solution P concentration following poultry litter and manure application was regulated by large calcium (Ca) inputs via the formation of Ca-P precipitates. This resulted in slower rates of P desorption (compared to cow manure) due to the increased P sorption strength through the formation of Ca-P complexes at the surface of alumino-silicate clays (Siddique and Robinson, 2004).

A potentially significant source of OP in poultry litter is phytate (myo-inositol hexakisphosphate), a heavily phosphorylated organic compound which is largely indigestible by poultry, thus forming a major component of their excreta (Hill and Cade-Menun, 2009). Even after composting, phytate dominates the organic P pool in poultry manure (Turner, 2004). Phytate bonds with soil minerals and the humic and fulvic organic matter fractions, meaning it is not very bioavailable to plants. It is also suggested that phytate and OP compete for the same sorption sites in soils (Anderson et al., 1974), potentially resulting in the release of OP from soils following the addition of materials containing high levels of phytate (Zhang et al., 1994).

Soil characteristics

Using untreated poultry manure as a fertiliser results in excess P accumulating in soils and in groundwaters (Kacprzak et al., 2023). When sampling the soils of agricultural fields that had received poultry litter for 10 ± 2 years, Chakraborty et al., (2021) found that 64 ± 11% of TP was in non-reactive P form, 35 ± 19% in moderately reactive P forms, and < 1% in the highly reactive P form.

A 20-year study that compared the application of poultry manure and urea-ammonium nitrate (UAN) fertiliser to corn fields found that particulate organic matter was significantly higher in the poultry manure plots than the UAN plots. This suggests that long term poultry manure application can potentially stabilise soil particles and increase infiltration and water holding capacity (Hoover et al., 2019).

Nitrate (NO_3) ions are repelled by the clay particles in the soil and generally are not absorbed within the soil matrix (Hubbard et al., 2004).

Spreading and storage methods

Storage method, timing of spreading, application method and rate are important variables influencing P concentrations in runoff following poultry manure/ litter application.

Storage

A long-term study (over 440 days) by McGrath et al. (2005) investigated P speciation of broiler litters stored dry (at their internal moisture content, ~24%) and wet (at a moisture content of 40%[15]). Wet-stored litter increased OP and decreased phytic acid

[15] 40% was chosen as this is represents the highest moisture content of poultry litter typically found in broiler houses in the Delmarva peninsula, USA. Poultry litter temperatures remained below 40 °C, therefore completed composting should not have occurred.

concentrations compared to the fresh and dry-stored poultry litter. Inorganic P comprised 22 – 41% of total extractable P in the fresh litter, which increased to 52 – 66% following wet storage. This translated to increased soluble P in runoff from the wet-stored litter (McGrath et al., 2005).

Kleinman et al. (2005) found dry manures contained significantly lower WEP than manures from liquid storage systems.

Storage time may also impact P speciation. McGrath et al. (2005) found that during storage WSP increased by ~65% (averaged for wet and dry-stored litter), whilst Vadas et al. (2004) reported a 52% increase in WSP for poultry litter that went through freeze-thaw cycles, which may have implications for poultry litter stored over the winter.

Time and extent of spreading

If poultry units have insufficient land area for storage and application, then agronomically and environmentally sensible poultry litter application rates and times can be compromised (Shepherd and Bhogal, 1998).

In the study by DeLaune et al. (2004), P concentrations in runoff showed a positive correlation with application rate, although the magnitude decreased with each runoff event, suggesting the soluble P concentration in the poultry litter is a major contributor of SRP in runoff (Figure 4). Similarly, Smith et al. (2019) found that high poultry litter application rates resulted in increased runoff Redfield P (and decreased Redfield C), compared to inorganic fertiliser application.

Based on the findings from Shepherd and Bhogal (1998), Autumn applications of poultry manures to sandy soils[16] should be avoided as N leaching was much greater compared to applications delayed into December.

[16] Soil in this study was loamy sand to about 1m over sandstone, with a clay content of 4 – 6% w/w (Cuckney series).

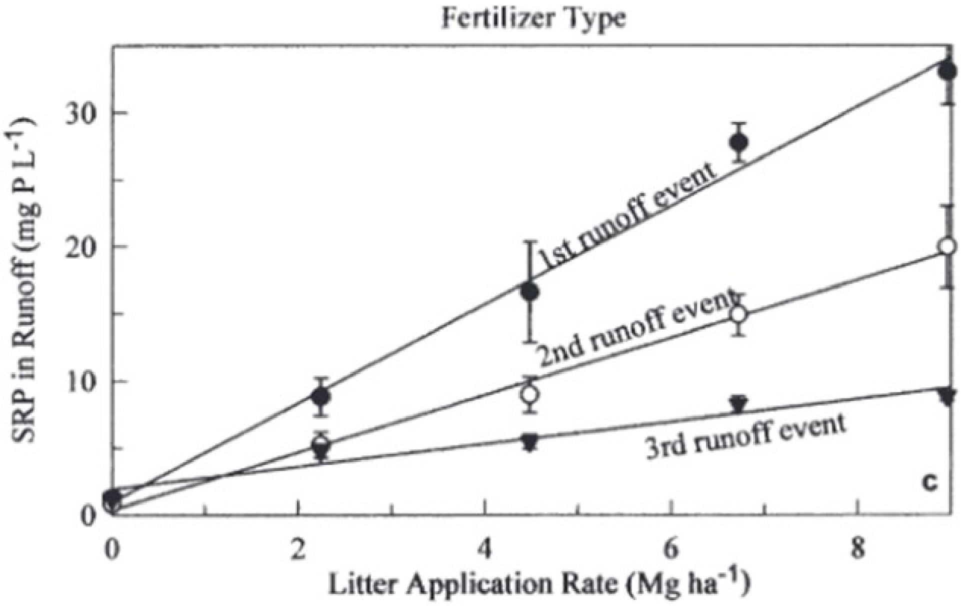


Figure 4. The effect of poultry litter application rate on SRP concentrations in runoff water, across three simulated rainfall events (DeLaune et al., 2004)

Surface versus subsurface application

Kibet et al. (2011) found that total dissolved P was highest in runoff from the surface-applied litter treatment (76% of TP). However, even though cumulative P losses were 1.6 times lower with subsurface application than with surface application over the course of the study, no significant differences were observed between the two treatments. These authors also found no statistical differences between treatments in particulate P (PP) in runoff, despite significant differences in total solids (Figure 5).

Bos et al. (2021) noted that the difference in annual DRP losses between row crops and pastures was the result of P application method. Poultry litter applied to row crops was incorporated with tillage, whilst litter was surface broadcast on pastures. Incorporating applied manure (and fertiliser) has been shown to decrease the concentration of dissolved P in runoff (e.g. Kleinman et al., 2006). Bos et al. (2021) suggest that it is possible that the decomposition of pasture grasses and subsequent release of DRP may also have contributed to the elevated DRP losses from pastures compared with row crops (Guppy et al., 2005; Noack et al., 2012, both in Bos et al., 2021).

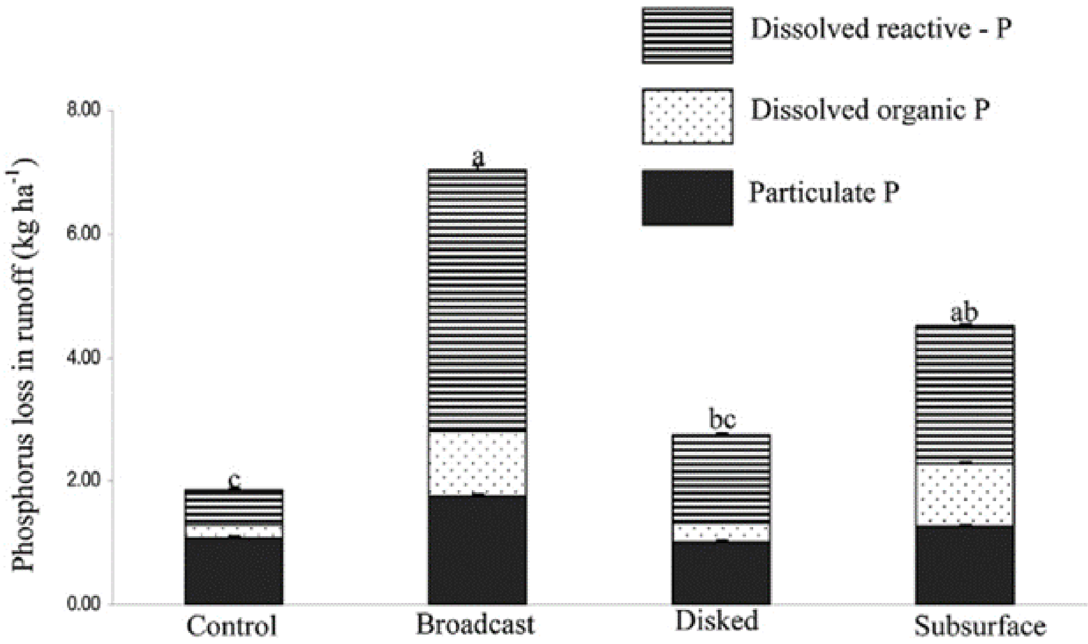


Figure 5. Cumulative P losses in runoff after two rainfall simulations occurring 15 and 42 days after poultry litter application, comparing control (no poultry litter), broadcast and disked (methods of surface application) and subsurface application (Kibet et al., 2011)

Other factors impacting bioavailable phosphorus downstream from manure sources

Several rainfall simulation studies have demonstrated that large increases in runoff P concentrations are observed following the application of poultry litter to agricultural land (Moore et al., 2000; Sauer et al., 2000; DeLaune et al., 2004; Haggard et al., 2005). Studies have found that runoff P, NH_4 and TN concentrations decreased with increased time between poultry litter application and a rainfall/ runoff event, however, no difference was observed for NO_3 (Schroeder et al., 2004).

Cox et al. (2013) explored the impact of poultry litter application (across a range of poultry farming intensities) on downstream water quality at a watershed scale. They reported significant positive correlations between in-stream TP concentrations (spring and summer[17]) and the intensity of poultry farm operations upstream of sample sites, suggesting that the application of poultry litter may be responsible for a five to tenfold increase in mean high flow P concentrations above background levels at the highest density of poultry operations.

[17] The study only sampled during the spring and summer to coincide with the most active period of poultry litter spreading and to capture the majority of the high flow events.

Mitigations measures to reduce phosphorus transport to freshwater

Researchers have suggested several measures to reduce the amount and/or solubility of P in poultry litter before it is applied to agricultural land. These include dietary supplements (Maguire et al., 2005, 2006), the addition of amendments such as iron (Fe), aluminium sulphate or lime (Vadas et al., 2004; Maguire et al., 2006a,b) and treatments such as composting (Stiles, 2017). Atalay et al. (2011) conclude that soil treatment with poultry litter and chemical amendments, especially alum and ferrous sulphate, could result in improved crop yield, soil aggregation, carbon (C) storage, and P immobilisation.

Dietary modifications

Dietary modifications can significantly reduce TP in poultry manure but may also change the speciation/ forms of P. Providing diets closer to broiler chicken nutritional requirements reduced litter TP by 18% compared to those fed normal diets[18], whereas adding phytase[19] reduced P contents by 29% compared to normal diets (McGrath et al., 2005). However, the effect of dietary modification on soluble P is less well understood. DeLaune et al. (2004) found that runoff SRP concentrations were higher for birds fed a phytase and high available P (HAP) corn diet (DeLaune et al., 2004, Figure 6). Maguire et al. (2006) found that combining decreased dietary P and phytase reduced both manure TP and WSP by 42%.

Toor and Sims (2016) compared P in leachate from soils amended with broiler litter from both high and low P diets across three different soil types. They concluded that the lower loss of DRP from lower-diet P broiler litter shows that the addition of phytase to the diet did not increase P leaching from the soil.

[18] Diets recommended by the USA National Research Council (NRC)

[19] Poultry and pigs lack the phytase enzyme needed to utilise P from phytic acid. As a result only 10-20% of the available P in corn is available to poultry and pigs. Because of this, farmers may add inorganic P to animal diets, however an alternative is to add phytase, which increases the availability of P in the food source.

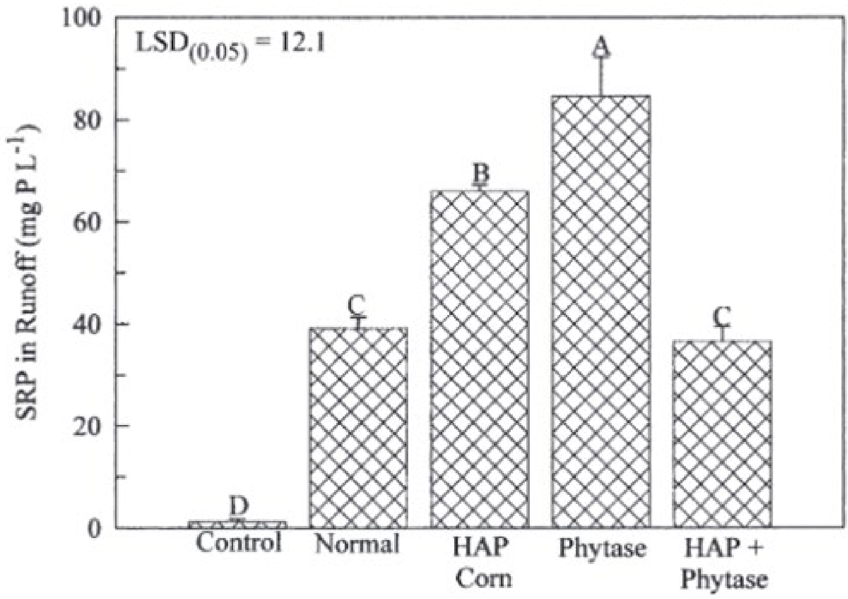


Figure 6. The effect of poultry diet type on SRP concentrations in runoff, comparing a normal diet, HAP corn diet, addition of the phytase (enzyme) and HAP plus phytase (DeLaune et al., 2004)

Alum application

Alum is added to poultry litter for various benefits such as reducing P and ammonia, increasing N and improving bird health. Concentrations of P in runoff were approximately three times lower when poultry litter was treated with alum to reduce the water-soluble content of P in the poultry litter (Moore et al., 2000).

A significant negative relationship was found between the amount of alum present in the litter and SRP concentrations in the runoff water, with SRP concentrations in runoff reduced by 42% with 5% alum, 49% with 10% alum, and 97% with 20% alum (DeLaune et al., 2004, Figure 7). The alum-treated litter produced the lowest SRP runoff concentrations compared to dietary modifications (DeLaune et al., 2004).

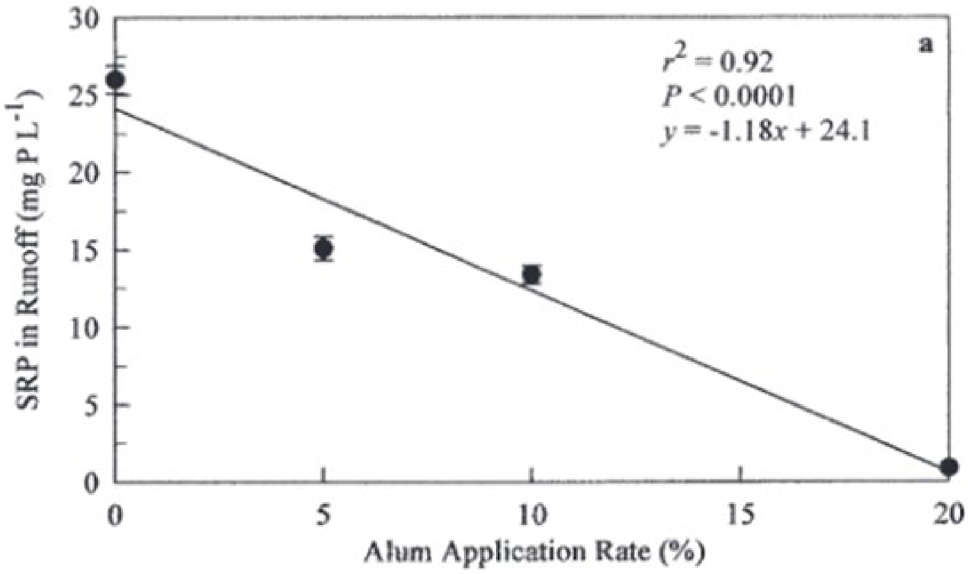


Figure 7. The effect of alum application rate on SRP runoff concentrations (DeLaune et al., 2004)

Cow manure and slurry

Introduction

Cow manure can be applied to land as manure (more solid-form, >20% dry matter) and slurry (more liquid-form, 4-20% dry matter) (European Union (EU), 2023). Cow manure may be composted before being applied as a fertiliser.

There are approximately 1.12 million cattle and calves in Wales (Welsh Government, 2023). Daily manure production per animal varies between 7 litres (calves) and 64 litres (cows with a milk yield >9,000 litres) (CoAPR, 2021). Annual manure production for the total number of cattle and calves in Wales (2023) is around 14 billion litres, which equates to around 27,000 tons of P per year and 68,000 tons of N per year.

Typical nutrient concentrations

Sharpley and Moyer (2000) tested the chemical composition of manures (including dairy cattle manure and dairy cattle manure compost) and found these to be wide ranging; thought to be a function of diet and compost additives. In terms of the composition of dairy manures, they found; for dairy manure 63% is inorganic P; 25% organic P and 12% residual P; for dairy compost 92% inorganic P; 5% organic P and 3% residual P. Further detail of the P fractions showed that in dairy manure, water soluble inorganic P dominates (51%) and in dairy compost bicarbonate inorganic P and acid inorganic P (36% and 33%) dominate.

Sharpley and Moyer (2000) also calculated the C:P ratios of the manures. The dairy manure and dairy compost C:P ratios are below 200:1 (87 and 20, respectively) indicating P mineralisation when applied to land will occur for both, with it being greater for dairy compost compared to dairy manure.

As described in further detail below, the diet of cattle can affect the nutrient composition of the resulting manure. O'Rourke et al. (2010) assessed P concentrations of dairy manure from different P-containing diets and found that as the P-containing diet decreased (from 5.3 – 3.0 g P/kg dm) the TP concentration of the manure decreased (from 13.3 to 4.9 g P/kg dm) and the % WSP of TP decreased also (from 3.6 to 1.4%).

Table 6 shows concentrations of P, N and dry matter in cow manure/ slurry from the literature[11].

Table 6. Concentrations of P, N and dry matter in cow manure/slurry from the literature[11]

Nutrient	Concentration in g/kg unless otherwise specified (type, other details)	Reference
Dry matter (%)	8.00 (dairy)	Vadas et al., (2006)
Dry matter (%)	19.00 ± 4.00 (dairy manure/ slurry)	Kleinman et al. (2005)
Dry matter (%)	30.10 ± 2.40	Sharpley and Moyer (2000)
Dry matter (%)	46.00 ± 37.00 (beef manure/ slurry)	Kleinman et al. (2005)

Nutrient	Concentration in g/kg unless otherwise specified (type, other details)	Reference
WSP	0.20	Montgomery et al. (2005)
WSP	0.75 (cowpie[20])	Soupir et al. (2006)
WSP	0.30 g/l (liquid dairy manure)	Soupir et al. (2006)
WSP	2.50 (slurry, d.w.)	Siddique and Robinson (2003)
WSP	3.41 (fresh)	Elliott et al. (2005)
WSP	1.07 (Ferric chloride treated)	Elliott et al. (2005)
WSP	0.31 (alum treated)	Elliott et al. (2005)
WSP	2.09 (dairy manure)	Sharpley and Moyer (2000)
WSP	2.39 (dairy compost)	Sharpley and Moyer (2000)
WSP	3.16 – 4.74	Brandt et al (2004)
WSP	2.30 (beef catlle, d.w.)	Kleinman et al. (2005)
WSP	4.0 (dairy cattle, d.w.)	Kleinman et al. (2005)
SRP	0.31 (faeces)	Sauer et al. (1999)
TP	4.30	Montgomery et al. (2005)
TP	0.562 (faeces)	Sauer et al. (1999)
TP	5.69, 5.56, 5.74, 5.52 (slurry, d.w.) Dec, Jan, Mar, Apr	McConnell et al. (2016)
TP	7.56	Vadas (2006)
TP	7.10 (slurry, d.w.)	Siddique and Robinson (2003)
TP	3.49 (dairy manure)	Sharpley and Moyer (2000)
TP	16.25 (dairy compost)	Sharpley and Moyer (2000)
TP	0.54 g/l (dairy slurry)	Murnane et al. (2018)
TP	13.30 – 4.90 (varying content of P in diets, d.w.)	O'Rourke et al. (2010)
TP	5.10 (beef cattle, d.w.)	Kleinman et al. (2005)
TP	6.90 (dairy cattle, d.w.)	Kleinman et al. (2005)
TP	1.802 (cattle manure, Aqua regia extractable)	Requejo and Eichler-Löbermann (2014)
TP	12.80 [11.30 – 13.80] (Liquid cattle manure)	Withers et al. (2001)
TN	21.10	Montgomery et al. (2005)
TN	5.50 (faeces)	Sauer et al. (1999)
TN	2.98, 2.99, 3.14, 3.17 (slurry, d.w.) Dec, Jan, Mar, Apr	McConnell et al. (2016)
TN	30.70 (slurry, d.w.)	Siddique and Robinson (2003)
TN	18.90 (dairy manure)	Sharpley and Moyer (2000)
TN	21.50 (dairy compost)	Sharpley and Moyer (2000)
TN	1.16 g/l (dairy slurry)	Murnane et al (2018)
TN	58.00 (55.00 – 61.00) (liquid cattle manure)	Withers et al. (2001)
Ammonium N	1.91, 1.92, 1.99, 2.18 (slurry, d.w.) Dec, Jan, Mar, Apr	McConnell et al. (2016)
WEP (inorganic)	2.71	Vadas (2006)
WEP (organic)	0.35	Vadas (2006)
Inorganic P	1.51 ± 272.00 (Cattle manure, NaOH-EDTA[21] extractable P)	Requejo and Eichler-Löbermann (2014)
Inorganic P	2.52 (dairy manure)	Sharpley and Moyer (2000)

[20] In this study cowpies were constructed from fresh dairy cow deposits scraped from dairy stalls. Cowpies were formed by mixing the fresh manure, then placing it in a mould until a weight was reached. (Soupir et al. 2006)

[21] Sodium hydroxide (NaOH)-Ethylenediaminetetraacetic acid (EDTA)

Nutrient	Concentration in g/kg unless otherwise specified (type, other details)	Reference
Inorganic P	15.25 (dairy compost)	Sharpley and Moyer (2000)
Organic P Organic P Organic P	0.98 (dairy manure) 0.85 (dairy compost) 0.33 ± 0.03 (Cattle manure, NaOH-EDTA extractable P)	Sharpley and Moyer (2000) Sharpley and Moyer (2000) Requejo and Eichler-Löbermann (2014)
Residual Residual	0.49 (dairy manure) 0.43 (dairy manure compost)	Sharpley and Moyer (2000)
DRP	0.34 g/l (dairy slurry)	Murnane et al (2018)
Dry solids Dry solids	46.00 [35.00 – 72.00] 30.20 %	Withers et al. (2001) Requejo and Eichler-Löbermann (2014)

Nutrient transformations, accumulation and transportation

Monitoring of P in surface water and sub-surface runoff from field experiments in Herefordshire where cattle slurry, farmyard manure and inorganic fertiliser were applied, indicated that there were differences in the forms of P in runoff following cattle slurry application. In the second and third monitoring years, the proportion of TP loss as molybdate-reactive phosphorus (MRP) and DRP increased from ~30% and 40 – 50% in the control to 45 – 55% and 55 – 60% in the slurry application for MRP and DRP respectively (Smith et al., 2001).

Brandt et al. (2004) sampled various manures and biosolids, including dairy manure, and reported the percent of WEP. Higher percentages of WEP indicate greater P liability. Dairy manure had a mean WEP of 52%, which was higher than poultry manure, but lower than TSP (triple superphosphate fertiliser) (Figure 8). Elliott et al. (2005) found a strong positive correlation between TDP and TP in runoff and the WEP concentration of the manure applied (including for dairy manures). Both of these studies indicate that the higher the WEP of the manure applied, then the greater P runoff there will be.

Soupir et al. (2006) analysed runoff from both release plots (to measure detachment of nutrients from manures) and transport plots (to measure nutrients in overland flow) with and without different manure applications (including liquid dairy manure and cowpies[20]).

Compared to the control plot, the concentrations of dissolved P, PP and TP were significantly greater in the runoff from plots to which liquid dairy manure and cowpies were applied. There were also differences in the form of P; 65% of TP was dissolved P in runoff from the liquid dairy manure plot, 40% for the cowpie plot and 42% for the control plot (a statistically significant difference between the two cow manures, but not with the control). It was thought that this is due to the liquid manure being able to infiltrate into the soil and had fewer organic matter particles available for P attachment than cowpies. Similarly, 60% of TP was PP in the runoff from the cowpie application runoff plot, but only 35% in the runoff from the liquid dairy application and 58% from the control plot (a statistically significant difference between the two cow manures, but not with the control). In terms of N, the highest concentrations of NH_3 and Total Kjeldahl nitrogen (TKN) were measured in runoff from the liquid dairy (a statistically significant difference between the liquid dairy and the control), and the highest concentrations of nitrite (NO_2)-NO_3 were in runoff from the liquid dairy and control (a statistically significant difference to the cowpie plot).

In the transport plots there were no significant differences in average concentrations of dissolved P, PP or TP between dairy liquid, cowpie and the control plot, but significant differences in proportions of different types of P: 85% of TP in the runoff was dissolved P for liquid dairy and 54% for cowpies and 36% for control (statistically significant difference between liquid dairy and the control plot, but not others); and, 15% of TP in the runoff was PP for liquid dairy and 46% for cowpies and 64% for control (statistically significant difference between liquid dairy and control, but not others). In terms of N, there were no statistically significant differences in average concentrations of NH_3, NO_2-NO_3 and TKN between dairy liquid, cowpie and the control, although runoff from the cowpie had the highest average concentration of NH_3 and TKN, and runoff from the liquid dairy had the highest average concentration of NO_2-NO_3.

Looking at the losses at the edge of the field, there were no statistical differences in P losses between the different manures applied. This is thought to be due to the manure applications being made to pastureland, and therefore the grass being able to retain nutrients and reduce transport.

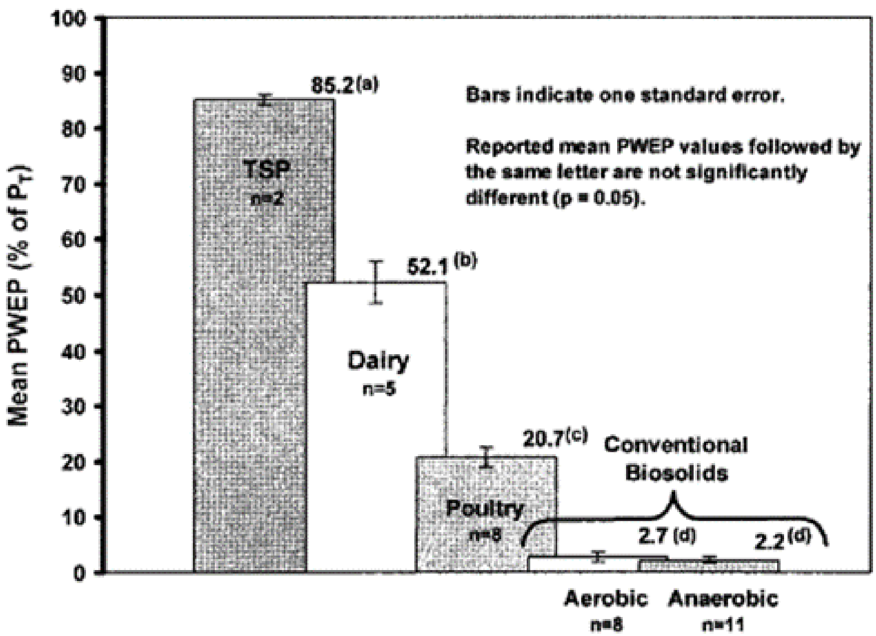


Figure 8. Comparison of WEP for TSP (triple superphosphate fertiliser), manures and conventionally treated biosolids (Brandt et al. 2004)

Soil characteristics

Toor and Sims (2016) compared P in leachate from three different soils (Matapeake - well-drained silt loam, Pocomoke - moderately well-drained sandy loam, and Woodstown - very poorly drained sandy loam) amended with dairy manures from both high and low P diets.

The proportions of the different P fractions were similar for the Pocomoke sandy loam and Woodstown sandy loam, but for the Matapeake silt loam the high-P diet manure had greater DRP and dissolved unreactive P compared to the low-P diet manure.

Murnane et al. (2018) tested the leaching of nutrients from soil samples taken from grass plots in County Cork, Ireland after the application of different manures. They found that no soil P losses were recorded in leachate after the application of all treatments (including dairy slurry); thought to be due to the low proportions of P in the applications compared to the P storage capacity in the soil (i.e., low soil P concentrations, high adsorption capacity). Also, as the testing was undertaken within a laboratory macropores in the soil were destroyed from the packing of the columns.

In their study, McConnell et al. (2016) found that higher soil moisture contents increased runoff and therefore will increase the risk of nutrient loss following manure application. Soupir et al. (2006) found that soils which have received previous manure applications had a larger proportion of PP in their runoff. This was thought due to the higher Total Suspended Solids (TSS) concentrations in runoff, and thus more particles available for nutrient transport.

Chichester et al. (1979) studied a beef cattle-pasturing system on sloping upland watersheds in Ohio (USA). No measurable sediment was lost from pastures used only for summer grazing. Soil and plant-cover disturbance on the pasture used for winter-feeding, however, resulted in some surface erosion, increased run-off and more chemical movement compared with pastures only grazed during the summer. These authors also report that considerably more chemicals moved in subsurface than in surface flow from the summer pastures, whilst the levels of chemicals transported from the winter-feeding pasture were equally as great in subsurface and surface flow.

Spreading and storage methods

Application method

Heinonem-Tanski and Uusi-Kämppä (2001), when comparing surface-spreading and injection application methods of cattle slurry, found there were higher losses of TP and TN from surface-spreading than injection (Surface-spreading: TP loss 1.3 and 1.4 kg/ha; TN loss 2.2 and 5.5 kg/ha. Injection: TP loss 0.2 and 0.3 kg/ha, TN loss 0.7 and 1.4 kg/ha). Additionally, the WSP, NO_3 and NH_4 concentrations within the soil were significantly greater for broadcast (surface applied) plots than plots where slurry was injected.

McConnell et al. (2016) compared two slurry application methods; splash-plate and trailing-shoe. It was found that the trailing-shoe application method reduced P in runoff compared to the traditional splash-plate application method regardless of the weather and soil conditions. For example, DRP, PP and TP reduced by 41, 25 and 32% during the rainfall event two days after application. It was thought this occurred due to the smaller slurry-rainfall contact area and slurry being placed at base of sward allowing plants to intercept rainfall from the trailing-shoe method.

Application rate

In analysis of soils from plots where long-term (since 1970) application of fertiliser has occurred, P adsorption to soils was higher when cow manure was applied at a lower rate compared to a higher rate (Figure 9; Anderson and Wu, 2001).

A study by Smith et al. (2001) found that increasing slurry application rate and slurry solids loading increases P (and solids) loss via surface runoff. They found that an application loading of around 2.5 – 3 t/ha slurry solids appear to be the threshold above which the risk of P loss in surface runoff losses appears to greatly increase.

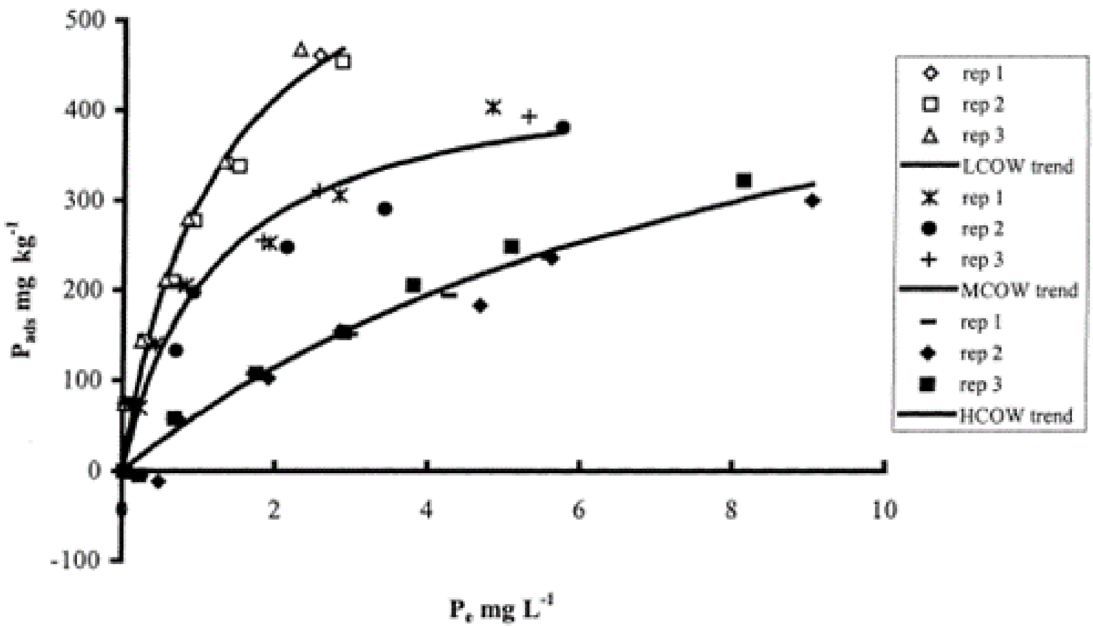


Figure 9. Phosphorus adsorption isotherms for cow slurry treatments at three application rates: low (LCOW: 50 m^3/ha/yr), medium (MCOW: 100 m^3/ha/yr), and high (HCOW 200 m^3/ha/yr) (Anderson and Wu, 2001)

Time following application

McConnell et al. (2016) compared a control plot to plots with cow slurry applied via the trailing-shoe application method and the splash-plate application method, and simulated rainfall 2, 9 and 16-days after manure application. Mean concentrations of DRP, PP and TP were all significantly different when measured in the runoff on day 2 between the application methods and the control plot, with concentrations increasing control < trailing-shoe < splash-plate. However, in the runoff in day 9 and 16, there were not significant differences in the mean DRP, PP and TP concentrations between trailing-shoe and splash-plate runoff (apart from DRP in the day 16 runoff; Figure 10).

O’Rourke et al. (2010) calculated the length of the P signal in overland flow, giving an indication of the time during which elevated P concentrations above those in the control persist after manure application. DRP declined to the threshold (defined as 1 mg/l) after 9 days in both summer and winter, and 28 days in the spring.

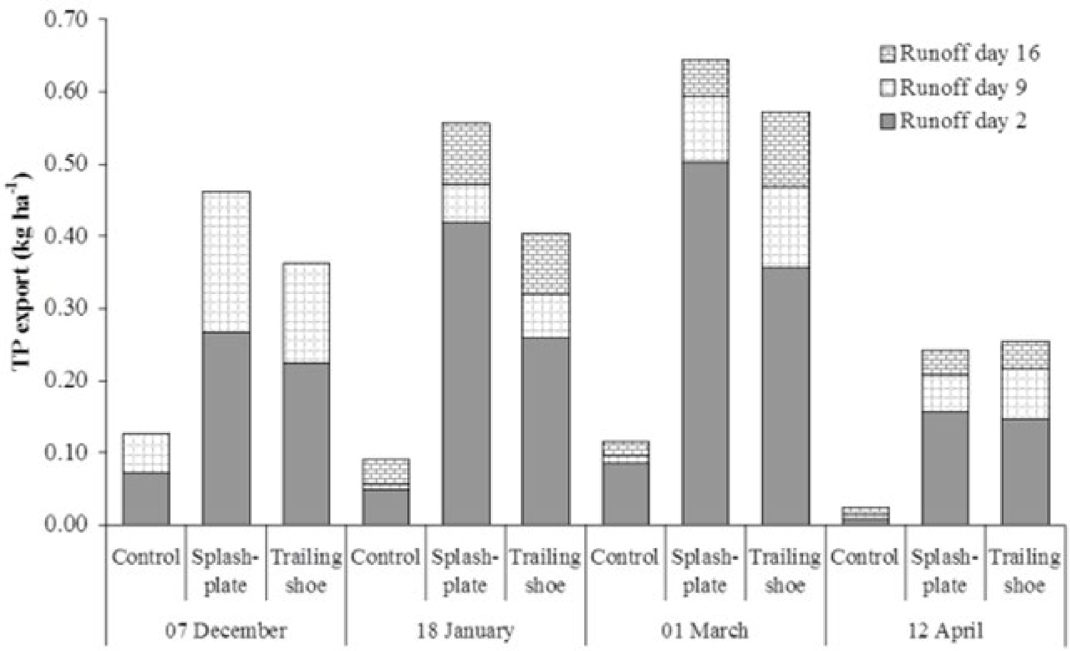


Figure 10. The effect of application date and slurry treatment on TP export rates in runoff, collected 2, 9 and 16 days post slurry application (McConnell et al., 2016)

Timing of application

O’Rourke et al. (2010) found the highest TP concentrations in overland flow were from winter applications of manure, thought to be due to high rainfall, high soil moisture content and low soil temperatures. Additionally, there was an increased proportion of PP in runoff during winter months, thought due to erosion of soil particles. However, the greatest TP loss was in summer applications of manure.

Other factors impacting bioavailable phosphorus downstream from manure sources

In their study, Smith et al. (2001) noted that sealing of the soil surface by slurry solids is a major mechanism contributing to the pollution of surface runoff following slurry application on susceptible soils. The sealing results in a decrease in surface infiltration capacity, and if heavy rainfall occurs within a few days of application of the slurry, then there is a high risk of polluting surface runoff.

Toth et al. (2006) investigated differences in nutrient leaching with four different manure applications applied to three different crops (alfafa, corn and orchard grass). Overall, there were no significant differences in the mean flow-weighted concentrations of NO_3-N or TP in the leachate, or mass loss in leachate, between the crops and manure types.

Mitigations measures to reduce phosphorus transport to freshwater

A number of methods are applied to reduce the potential P loss from soils amended with cow manure/ slurry. These include manure amendments, dietary changes and application timing considerations.

Manure amendments

Amending the composition of manure prior to application can reduce nutrient concentrations in runoff. Application of cow slurry amended with poly-aluminium chloride hydroxide, alum, lime and ferric chloride was found to reduce DRP in runoff by 86%, 83%, 69% and 67% respectively by Brennan et al. (2011). Similarly, Elliott et al. (2005) set up run off plots and applied different forms of cow manure (fresh, ferric chloride treated and alum-amended). It was found that amending the dairy manure with ferric chloride or alum reduced the P in the runoff by about half. This was thought to be due to lower WEP concentrations in the un-amended manure. There was a statistical difference in terms of TP and TDP in the runoff between the raw manure applied and the amended-manure.

Dietary changes

Reductions in P-based diets for cows, has been shown to reduce P in the resultant manure. O'Rourke et al. (2010) tested manure from cattle with different diets and found that decreasing the P content of the cattle's diet from 5.3 to 3.0 g/kg resulted in a 63% reduction in the TP content of manures produced. However, there was not a consistent change in WSP with the change in diet (WSP accounted for 27 – 29%). In terms of runoff, the reduced P in the cattle's manure reduced both DRP and TP concentrations in overland flow but had the greatest effect on reducing DRP.

Toor and Sims (2016) compared P in leachate from soils amended with dairy manures from both high and low P diets. Interestingly, slightly higher concentrations of P were found in the leachate from the low-P diet manure, although this was not found to be statistically significant. It was thought this difference probably relates to the different amounts of manure added which resulted in variable application of C which effects P sorption and desorption in soils. This shows there may not be a simple relationship between P-diets, P in manure and its effect on nutrient runoff.

Other

Smith et al. (2001) list a range of strategies which should be employed to reduce the risk of nutrient loss from manure application: (1) restricting application rates, especially of slurries, to within those consistent with good agronomic practice; (2) avoiding applications to excessively wet soils and during periods of heavy rainfall, or when heavy rainfall is forecast within 48 hours; and, (3) careful soil management practice aimed at encouraging a well-structured, permeable surface which will allow rapid surface infiltration by slurries, manures and effluents.

Pig manure/ slurry

Introduction

The composition of pig (or swine) manure varies widely depending on the management system and diet, but is typically characterised by high N and C levels with variable total solids (Williams, 2020). Pig manure is predominantly liquid, with a low dry matter content (typically <10%; Figure 4) and therefore can be considered a slurry[22]. Pig manure may be composted before being applied as a fertiliser.

There are approximately 24,800 pigs in Wales (Welsh Government, 2023). Daily manure production per animal varies between 1.3 litres (weights 7 – 13 kg) and 10.9 litres (sows including their litter up to 7 kg/piglet) (CoAPR, 2021). Annual manure production for the total number of pigs in Wales (2023) is therefore around 46 million litres, which equates to around 17 tons of P per year and 300 tons of N per year[23].

Pig farming systems may contribute to the acidification and eutrophication of the environment due to emissions of P and N from manure storage and spreading (de Vries and de Boer, 2010).

Typical nutrient concentrations

Pig slurry has particularly high concentrations of P which may lead to a high accumulation of P in soils compared to other organic manures (Anderson and Wu, 2001). In the swine slurry collected by Vadas (2006), approximately 15% of the TP was WEP (a highly mobile form of P). Around 88% of the WEP was in the inorganic (and most bioavailable) form and 12% in the organic form.

Roughly 50% of N in the swine manure/ slurry is present in NH_4 form and the remaining 50% is present as organic N (Sutton et al., 1978, 1982, 1978; Burns et al., 1987). The dry matter content of liquid swine manure has shown weak negative correlations with WEP suggesting the possible dilution effect of manure water on increasing P solubility (Kleinman et al., 2005).

Tables 7-9 show concentrations of P, N and dry matter in pig manure/ slurry from the literature[11] (separated by the description and units used in the original reference).

[22] Defined by Kleinman et al. (2005) as manures with a dry matter content of <10%.

[23] Based on a daily manure production of 5.1 litres/animal which is applicable to pigs over 66 kg that are intended for slaughter and dry fed (CoAPR, 2021).

Table 7. Concentrations of P, N and dry matter in pig manure/slurry from the literature[11]

Nutrient	Concentration (g/kg)	Reference
Dry matter (%) Dry matter (%) Dry matter (%) Dry matter (%) Dry matter (%)	5.20 6.30 8.00 ±9.00 10.70 ± 1.60 11.00	Marshal and Laboski (2005) Vadas (2006) Kleinman et al. (2005) Sharpley and Moyer (2000) DeLaune et al. (2010)
WSP	4.20	Montgomery et al. (2005)
WEP (slurry)	9.20	Kleinman et al. (2005)
WEP inorganic	6.49	Vadas (2006)
WEP organic	0.91	Vadas (2006)
TP TP TP TP	24.70 28.80 ± 10.40 (d.w) 47.37 32.60 ± 1.90	Montgomery et al. (2005) Kleinman et al. (2005) Vadas (2006) Sharpley and Moyer (2000)
Inorganic P	30.10	Sharpley and Moyer (2000)
Organic P	2.46	Sharpley and Moyer (2000)
Residual P	0.36	Sharpley and Moyer (2000)
Water-P	6.08	Sharpley and Moyer (2000)
TN TN TN	50.60 89.00 ± 56.90 (d.w) 63.70 ± 5.70	Montgomery et al. (2005) Kleinman et al. (2005) Sharpley and Moyer (2000)

Table 8. Concentrations of P and N in pig slurry from the literature[11]

Nutrient	Concentration (g/l)	Reference
TP TP TP TP	0.56 0.53 0.20 12.07 ± 0.87	O' Flynn et al. (2012) Marshall and Laboski (2006) Murnane et al (2018) García-Albacete et al. (2012)
Inorganic P Inorganic P	0.42 11.50 ± 0.04	Marshall and Laboski (2006) García-Albacete et al. (2012)
Soluble P	0.15	DeLaune et al. (2010)
TN TN TN	2.15 ±0.21 3.27 3.70	O' Flynn et al. (2012) Marshall and Laboski (2006) Murnane et al (2018)
Ammonium (NH_4-N)	1.25 ± 0.04	O' Flynn et al. (2012)
Dissolved reactive P	0.14	Murnane et al (2018)
OP	0.07 ± 0.01	García-Albacete et al. (2012)
NAIP	4.25 ± 0.22	García-Albacete et al. (2012)
AP	6.17 ± 0.36	García-Albacete et al. (2012)
WSP inorganic	10.01 ± 0.65	García-Albacete et al. (2012)
WSP total	10.06 ± 0.71	García-Albacete et al. (2012)

Table 9. Concentrations of P in pig slurry compost from the literature[11]

Nutrient	Concentration (g/kg)	Reference
TP	53.40 ± 4.10	García-Albacete et al. (2012)
Inorganic P	49.42 ± 2.00	García-Albacete et al. (2012)
WSP inorganic	4.70 ± 0.20	García-Albacete et al. (2012)
WSP total	4.80 ± 0.30	García-Albacete et al. (2012)

Nutrient transformations, accumulation and transportation

The long-term application of pig slurry to land was found to significantly increase soil P reserves (TP and agronomic P (Olsen P)) and N down to a layer of 30 – 40 cm (silty loam and clay loam soils) (Hountin et al., 1997; Anderson and Xia, 2001). Shappell et al. (2016) found that the application of swine manure resulted in runoff with 14 times more P and three to four times more total organic carbon and total N compared to a control (no manure) plot. Studies also suggest that more P accumulated in soils treated with pig manure compared to cow manure.

García-Albacete et al. (2012) report that composting of pig slurry may potentially reduce the loss of P in the first few days following application by approximately ten-fold compared to raw pig slurry. Suggesting that this may occur due to the high amount of calcium in the slurry compost compared with pig slurry; Ca can form Ca-P complexes which reduces P solubility (Kleinman et al., 2005).

Soil characteristics

The application of liquid swine manure can result in changes to the soil chemical composition. Changes are influenced by factors such as soil texture, rate, time and method of manure application, amount of precipitation, crops grown and time of sampling (Choudhary et al., 1996).

Swine manure has one of the lowest dry matter contents and the highest WEP compared to other livestock manures. Liquid manures have the potential to penetrate soil macropores below plough depth and remain easily mobile within the soil. This acts as a potential source of P loss, particularly for manures with high WEP (Hodgkinson et al., 2002).

Spreading and storage methods

Application method, application rate and the amount of time following slurry application are important variables influencing P concentrations in runoff following swine manure application (Schuster et al., 2017). In contrast, runoff loads of NO_3 and TN were not significantly affected by slurry application method or time following slurry addition (Schuster et al., 2017).

Application method

The dissolved P, TP, NO_3 and NH_4 concentrations within the soil were significantly greater for broadcast (surface applied) plots than plots where slurry was injected (Schuster et al., 2017). Soon after swine slurry application, dissolved P and TP loads in runoff were reduced by 60% and 47% respectively when slurry was injected instead of broadcast (Gilley et al., 2013). Daverde et al. (2004) found that injecting swine slurry resulted in DRP and TP runoff load reductions of 99% and 94% respectively, one month and six months after application.

Application rate

P adsorption to soils was higher when pig manure was applied at a lower rate compared to a higher rate (Figure 11; Anderson and Wu, 2001).

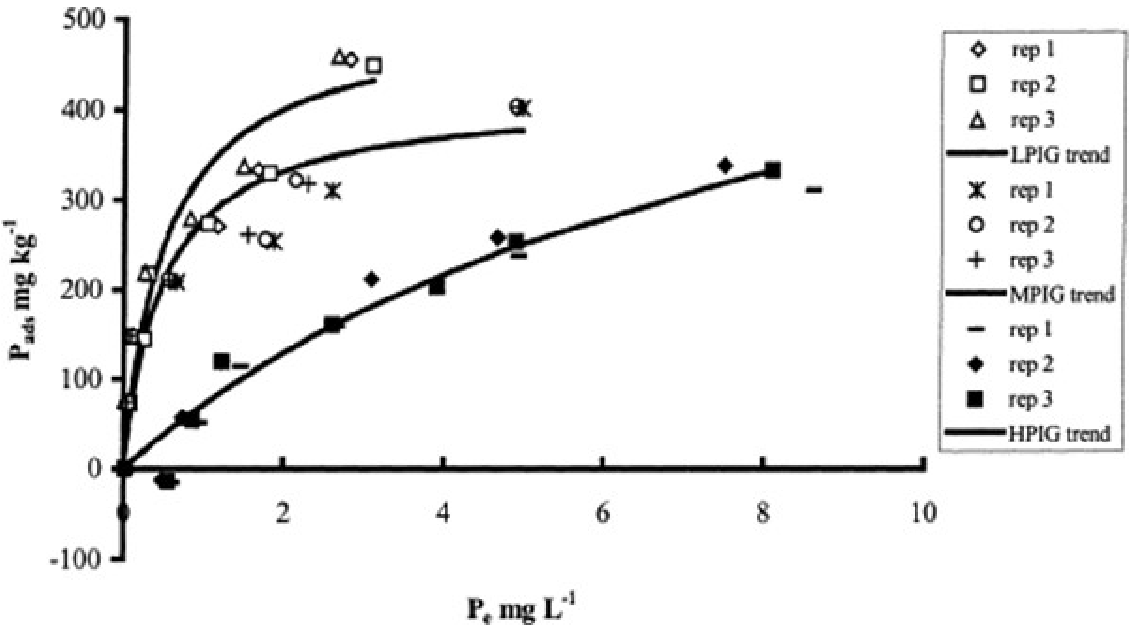


Figure 11. Phosphorus adsorption isotherms for pig slurry treatments at three application rates: low (LPIG: 50 m^3/ha/yr), medium (MPIG: 100 m^3/ha/yr), and high (HPIG 200 m^3/ha/yr) (Anderson and Wu, 2001)

Time following application

Smith et al. (2007) found that pig manure posed the greatest risk to water quality one day after application due to elevated P and NH_4 in runoff, however this risk decreased rapidly with increased time since application. Similarly, Schuster et al. (2017) found that the length of time following application significantly impacted soil concentrations of WSP and NH_4, with the largest values occurring in the first eight days following application. Therefore, P losses in runoff can be significantly reduced if slurry is broadcast when rainfall is not expected during the 15 to 17 days following application (Schuster et al., 2017).

Storage methods

Swine waste is often stored in waste retention pits which then discharge into a lagoon system for stabilisation (digestion of settled solids) (Walker et al., 2003) or into an above-ground slurry storage tank. Pig slurry must be collected and stored in tanks/ pits with a sufficiently large capacity before it is applied to agricultural land. If pits and tanks are not well maintained, or do not meet regulatory standards then overflow and/or leakage may pollute the water environment (Marszałek et al., 2019).

Smith et al. (2017) investigated the impact of manure storage types (deep pits, manure vats[24] and anaerobic lagoons) on nutrient concentrations. They found that both deep pits and lagoons had significant increasing trends in N concentrations, with similar results for P. Average nutrient concentrations (2001– 2015) for different swine slurry storage methods (Iowa, USA) are presented in Table 10.

Table 10. Swine slurry nutrient concentrations for different storage methods (values are in g/l) (Smith et al., 2017)

Nutrient	Deep pit	Vat	Lagoon
N	5.81	3.55	1.72
P	3.39	1.66	1.37

Other factors impacting bioavailable phosphorus downstream from manure sources

Studies have shown that the greatest potential for nutrient transport results from a rainfall event occurring soon after application of swine slurry (Schuster et al., 2017), however, the rate of runoff was also found to influence runoff water quality characteristics following swine manure application, with increased runoff rate resulting in increased TP (Figure 12), dissolved P and NH_4 (Schuster et al., 2017).

[24] Vats were uncovered storage tanks either above or below ground.

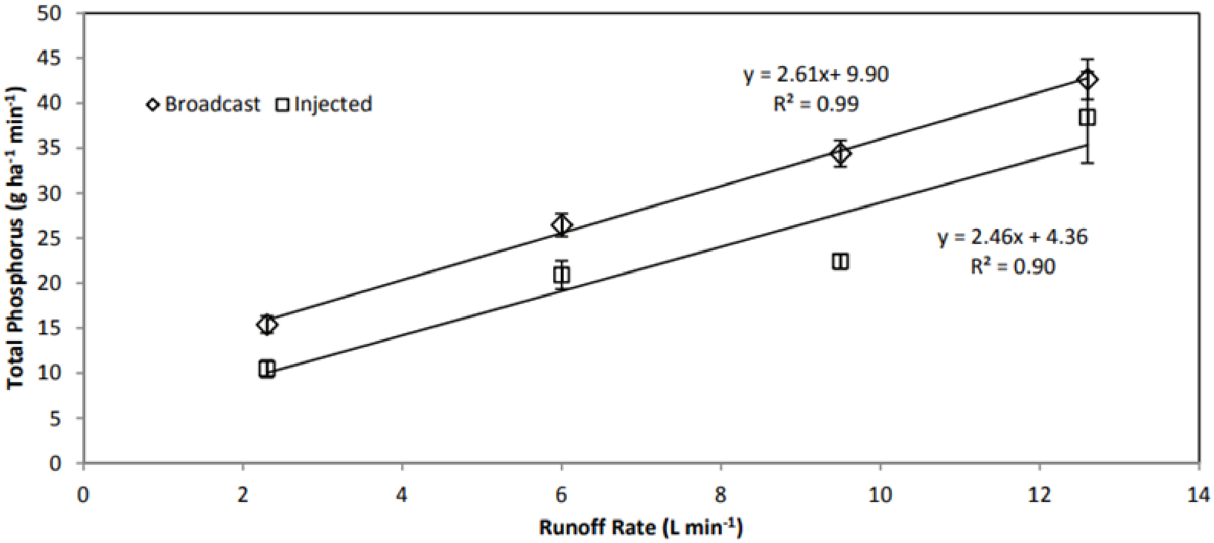


Figure 12. TP transport rate as affected by runoff rate from broadcast (surface applied) and injected experimental treatment plots (USA) (Schuster et al., 2017)

Mitigations measures to reduce phosphorus transport to freshwater

It has been suggested that currently available technologies should be able to achieve a 50% reduction in the TP content of swine manure (Maguire et al., 2006). For example, chemical amendment of pig slurry can reduce P in runoff without any negative impact on nutrient leaching and Greenhouse Gas (GHG) emissions.

O'Flynn et al. (2012) investigated the effectiveness of swine manure amendments at reducing DRP in surface runoff. These were (in decreasing order of effectiveness): alum (86%); flue gas desulfurization by-product (FGD) (74%); poly-aluminium chloride (PAC) (73%); ferric chloride (71%); fly ash (58%); and lime (54%). When these treatments were ranked by feasibility, accounting for effectiveness, cost, and other potential impediments to use, they were ranked: alum, ferric chloride, PAC, fly ash, lime, and FGD.

Whilst aluminium chloride additions did not impact manure TP concentrations, soluble P concentrations were significantly affected. DeLaune et al. (2010) found that the addition of aluminium chloride plus lime was the most effective treatment, resulting in a reduction of SRP concentrations by 69% compared to untreated manure. Similarly, Smith et al. (2001) found that aluminium (Al)-based swine manure amendments reduced P runoff by up to 84%.

Sheep manure

Introduction

There are currently 8.7 million sheep and lambs in Wales, which are mostly situated in the upland areas (Welsh Government, 2023). Sheep graze all year round in the UK although they may be housed indoors during periods of very cold weather, or prior to lambing (mid-March to early April) (Wu et al., 2022; Orr et al., 2016).

Daily manure production per animal varies between 1.8 litres (from 6 – 9 months old) and 5 litres (Weights over 60 kg) (CoAPR, 2021). Annual manure production for the total number of sheep and lambs in Wales (2023) is therefore around 5.5 billion litres, which equates to around 14,500 tons of P per year and 35,000 tons of N per year.

Typical nutrient concentrations

The dry matter content of sheep manure reported in the literature is variable, ranging from 11 – 73% (Domburg et al., 1998; Pagliari and Laboski, 2012) and it has been reported that inorganic P accounts for ~56% of TP (by dry weight) (Nguyen and Goh, 1991).

Table 11 shows concentrations of P, N and dry matter in sheep manure from the literature[11].

Table 11. Concentrations of P, N and dry matter in sheep manure from the literature[11]

Nutrient	Concentration	Reference
Dry matter Dry matter Dry matter	386.00 ± 16.00 g/kg 11.00% 73.30%	Pagliari and Laboski, (2012) Domburg et al., (1998) Mahmood et al. (2017)
TP TP TP	1.30 ± 0.30 g/kg 0.52% d.w 0.45% d.w	Jalali et al. (2022) (Iran) Nguyen and Goh (1991) Nguyen and Goh (1991)
Inorganic P	0.29% d.w (56.00% of TP)	Nguyen and Goh (1991)
TN TN	24.00 ± 7.00 g/kg 1.5% d.w	Pagliari and Laboski (2012) Nguyen and Goh (1991)
Ammonium-N	5.00 ± 5.00 g/kg	Pagliari and Laboski (2012)
Total inorganic P	4.40 ± 1.30 g/kg	Pagliari and Laboski (2012)
WEP	1.96 g/kg	Rowarth et al. (1985)

Nutrient transformations, accumulation and transportation

Well-managed permanent grasslands used for grazing of sheep/ lambs is recognised as a carbon sink under appropriate forage management through the lack of cultivation and recycling of carbon and nutrients directly and indirectly through inputs of manure (Orr et al., 2016)

In a field-based study in New Zealand, Rowarth et al. (1985) found that the rate of physical manure breakdown was the major mechanism controlling the movement of P from sheep manure into the soil rather than the leaching of P from the manure. However, this contrasted with laboratory studies which demonstrated high levels of WEP in sheep manure.

In a wetland mesocosm study (with sediment from Moroccan agricultural ponds), the addition of sheep manure did not increase nutrient concentrations in the mesocosm, which was suggested to be linked to the increases in pH with manure treatment which can result in the chemical precipitation of N and P (Van den Broeck et al., 2019).

Direct P loss from sheep manure deposited either directly in the streams, or in feeding areas close to the streams, contributed to the occasional high daily concentrations of dissolved and particulate P measured under stormflow conditions and sometimes during baseflow in an upland catchment in Northern England (Withers et al., 2007). However, there was no evidence to suggest that higher sheep stocking rates increased the transfer of manure P (or sediment), since grazing alone did not impact the P concentrations of the streams (Withers et al., 2007). This supports the findings of Cooke and Williams (1973), who suggest that sheep manure poses no significant risk for water quality in the UK, unless manure is being directly deposited within the water body.

Soil characteristics

No information deemed relevant for inclusion under this category was found in the literature reviewed.

Spreading and storage methods

Sheep manure is mostly applied directly to land by free-ranging animals. No information on alternative spreading or storage methods deemed relevant for inclusion was found in the literature reviewed.

Other factors impacting bioavailable phosphorus downstream from manure sources

Inorganic P fertilisers such as superphosphate are often applied to pastures to improve the quality of the vegetation on which sheep graze. The proportion of inorganic P in the sheep manure has been found to increase with increasing inputs of superphosphate (Nguyen and Goh, 1991).

The importance of sheep manure as a pathway for P return to the soil is affected greatly by topography. Rowarth et al. (1985) found that the rate of manure breakdown was most rapid on the flat areas and slowest on steep slopes.

In the Pennines (UK), faecal material deposited on moorland decomposed less rapidly than that deposited on grassland, and at each site, samples deposited in summer remained longer than those deposited in winter (White, 1960). Breakdown of manure in summer months was attributed almost entirely to earthworm activity, whereas in winter wind and precipitation were responsible from removing 30 – 50% of the manure (White, 1960).

Similarly, a study from New Zealand found that sheep manure samples had fully decomposed within 28 days in winter but remained for 75 days in summer (Rowarth et al., 1985). This was attributed to the increased moisture content in winter promoting decomposition, whereas once the manure had dried out, it became more resistant to physical breakdown (Rowarth et al., 1985).

Mitigations measures to reduce phosphorus transport to freshwater

Evidence suggests that direct deposits of sheep manure into water bodies pose the most significant risk to water quality. Reducing the contact between sheep and water bodies should therefore provide effective mitigation to reduce the risk to water quality.

Biosolids

Introduction

Sewage sludge is the residual solid waste left over from the treatment of wastewater. Sludge treatment is employed to generate biosolids (treated sewage sludge) that is safe and acceptable to recycle to agriculture. In the UK, AD (employed approximately 73% of the time) is the most commonly used treatment technology to produce biosolids. Lime stabilisation is employed to a lesser degree (approximately 22%), whilst small quantities of sludge are also treated by thermal drying and composting[25] (Assured Biosolids Limited, 2024).

According to Assured Biosolids Limited (2021), 87% of biosolids in the UK are recycled to agriculture (the remainder is used in land restoration or incineration) with biosolids applied to about 1.3% of the UK's agricultural land per annum.

Typical nutrient concentrations

Sewage sludges, and ultimately biosolids, are highly variable in composition and do not form a group of fertilisers with a well-known nutrient content (Quilbé et al., 2005). Table 12[26] below shows a large range in the concentrations of P and N fractions in biosolids. Shepherd and Withers (2001) found that P content, and the fractions of P in biosolids, was even variable between years when collected from the same works.

P content as well as speciation in biosolids vary depending upon the characteristics of the wastewater, the type (e.g. lime, ferric chloride, aluminium sulfate) and quantity of chemical added, and the place of addition of the chemical during the wastewater treatment process (Kirkham, 1982; Quilbé et al., 2005). Differences in the form and availability of P in different types of biosolids are also dependant on wastewater and sludge treatment type. Hinedi et al. (1989) cited in Montgomery et al. (2005), for example, found that waste-activated and aerobically-digested biosolids had higher amounts of organic than inorganic P, while anaerobically-digested biosolids contained mostly inorganic phosphate.

Concentrations of N in biosolids are also recorded in Table 12. In biosolids, N is mainly under organic forms, while mineral forms are generally in relatively low concentrations and are mostly represented by NH_4-N (Quilbé et al., 2005).

[25] Sometimes with green waste.

[26] Only data from selected articles reviewed, and selected forms of P and N, has been included in the table. In all instances it could be established from the articles that biosolids (treated sewage sludge) as opposed to 'raw' sewage sludge had been sampled.

Table 12. Concentrations of P, N and dry matter in biosolids from the literature

Sample Description Code	Nutrient	Concentration	Statistic	Units*	Ref.
A	TP	21.10	Mean	g/kg (d.w)	Siddique and Robinson (2003)
A	TN	35.80	Mean	g/kg (d.w)	Siddique and Robinson (2003)
A	WSP	1.70	Mean	g/kg (d.w)	Siddique and Robinson (2003)
A	Dry Matter	280.00	Mean	g/kg (d.w)	Siddique and Robinson (2003)
B	TN	1.30 – 1.70	Range	g/l	Shepard and Withers (2001)†
B	NH_4-N	0.43 – 0.86	Range	g/l	Shepard and Withers (2001)†
B	TP	0.40 – 0.56	Range	g/l	Shepard and Withers (2001)†
B	P (HCl)	0.17 – 0.47	Range	g/l	Shepard and Withers (2001)†
B	P ($NaHCO_3$)	0.16 – 0.18	Range	g/l	Shepard and Withers (2001)†
B	P (NaOH)	0.08 – 0.13	Range	g/l	Shepard and Withers (2001)†
B	P (H_2O)	0.05 – 0.11	Range	g/l	Shepard and Withers (2001)†
C	TP	26.50	-	g/kg (d.w)	Vanden Bossche et al. (2000)
C	Dry matter	23.00	-	g/l	Vanden Bossche et al. (2000)
D	TP	0.71	-	g/kg	Heathwaite et al. (2006)
D	Water-extractable PO_4–P	0.17 (0.06 – 0.22)	- Range	g/kg	Heathwaite et al. (2006)
D	TN	2.14	-	g/kg	Heathwaite et al. (2006)
D	Water-extractable NO_3–N	0.02 (0.009 – 0.013)	- Range	g/kg	Heathwaite et al. (2006)
D	NH_4–N	2.04 (1.75 – 3.92)	- Range	g/kg	Heathwaite et al. (2006)
E	TN	15.00	-	g N/kg (d.w)	Quilbé et al., (2005)
E	NH_4-N	2.40	-	gN-NH_4/kg (d.w)	Quilbé et al., (2005)
E	TP	78.00	-	gP_2O_5/kg (d.w)	Quilbé et al., (2005)
E	Dry matter	60.40	-	(%)	Quilbé et al., (2005)
F	TN	19.00	-	gN/kg (d.w)	Quilbé et al., (2005)
F	NH_4-N	0.20	-	gN-NH_4/kg (d.w)	Quilbé et al., (2005)
F	TP	22.00	-	gP_2O_5/kg (d.w)	Quilbé et al., (2005)
F	Dry matter	50.50	-	(%)	Quilbé et al., (2005)
G	TP	38.30 (± 0.003)	Mean (SD)	g/kg	Akhtar et al. (2002)
G	Extractable P	1.26 (± 3.00)	Mean (SD)	g/kg	Akhtar et al. (2002)
G	NO_3-N	0.0043 (± 0.0002)	Mean (SD)	g/kg	Akhtar et al. (2002)
G	NH_4-N	2.20 (± 0.17)	Mean (SD)	g/kg	Akhtar et al. (2002)
H	TP	19.50 – 59.00	Range	g/kg (d.w)	Tuszynska et al. (2021)
H	TN	10.10 – 17.10	Range	g/kg (d.w)	Tuszynska et al. (2021)
H	Organic P	9.10 (± 2.40)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
H	NAIP[Ø]	17.10 (± 5.50)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
H	AP[Ø]	8.00 (± 1.10)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
H	Labile P contribution	0.52 – 0.69	Range	Plab[Ø]: TP	Tuszynska et al. (2021)
H	Dry matter	14.40 – 19.50	Range	%	Tuszynska et al. (2021)
I	TP	0.10 – 0.30	Range	g/l	Tuszynska et al. (2021)
I	TN	1.10 – 1.50	Range	g/l	Tuszynska et al. (2021)
I	NH_4	0.90 – 1.40	Range	g/l	Tuszynska et al. (2021)
I	PRP[Ø]	0.10 (± 0.02)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
I	PNRP[Ø]	0.08 (± 0.01)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
I	SRP[Ø]	2.57 (± 0.33)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
I	SNRP[Ø]	0.17 (± 0.04)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
I	Labile P contribution	0.82 – 0.89	Range	Plab: TP	Tuszynska et al. (2021)
J	Dry solids	227.00 (183.00 – 272.00)	Mean Range	g/kg	Withers et al. (2001)
J	TN	32.00 (32.00 – 33.00)	Mean Range	g/kg (d.w)	Withers et al. (2001)

Sample Description Code	Nutrient	Concentration	Statistic	Units*	Ref.
J	TP	10.60 (10.20 – 13.10)	Mean Range	g/kg (d.w)	Withers et al. (2001)
K	Dry solids	71.00 (24.00 – 126.00)	Mean Range	g/kg	Withers et al. (2001)
K	TN	43.00 (29.00 – 58.00)	Mean Range	g/kg (d.w)	Withers et al. (2001)
K	TP	16.50 (13.10 – 21.40)	Mean Range	g/kg (d.w)	Withers et al. (2001)
L	TP	19.90	-	g/kg (d.w)	Chambers et al. (2002)
L	TN	48.90	-	g/kg (d.w)	Chambers et al. (2002)
L	Dry matter	17.60	-	%	Chambers et al. (2002)
M	IP	3.59 (± 0.19)	Mean (SD)	g/kg	García-Albacete et al. (2012)
M	Organic P	0.68 (± 0.05)	Mean (SD)	g/kg	García-Albacete et al. (2012)
M	NAIP	3.53 (± 0.29)	Mean (SD)	g/kg	García-Albacete et al. (2012)
M	AP	0.19 (± 0.02)	Mean (SD)	g/kg	García-Albacete et al. (2012)
M	TP	4.72 (± 0.32)	Mean (SD)	g/kg	García-Albacete et al. (2012)
M	WSP (inorganic)	0.94 (± 0.07)	Mean (SD)	g/kg	García-Albacete et al. (2012)
M	WSP (total)	1.20 (± 0.08)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	IP	27.38 (± 1.04)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	Organic P	0.80 (± 0.05)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	NAIP	8.77 (± 0.45)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	AP	14.10 (± 0.96)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	TP	31.25 (± 2.00)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	WSP (inorganic)	1.24 (± 0.09)	Mean (SD)	g/kg	García-Albacete et al. (2012)
N	WSP (total)	1.50 (± 0.10)	Mean (SD)	g/kg	García-Albacete et al. (2012)
O	TP	49.00	-	g/kg	Montgomery et al. (2005)
O	TN	23.40	-	g/kg	Montgomery et al. (2005)
O	WSP	0.10	-	g/kg	Montgomery et al. (2005)
P	TP	33.50	-	g/kg	Montgomery et al. (2005)
P	TN	7.70	-	g/kg	Montgomery et al. (2005)
P	WSP	0.20	-	g/kg	Montgomery et al. (2005)
Q	TP	16.00	-	g/kg	Montgomery et al. (2005)
Q	TN	10.20	-	g/kg	Montgomery et al. (2005)
Q	WSP	0.20	-	g/kg	Montgomery et al. (2005)
R	TP	23.51 (0.27)	Mean (SD)	g/kg	Peyton et al. (2016)
R	WEP (dry)	16.00 (8.00)	Mean (SD)	g/kg (dry)	Peyton et al. (2016)
R	TN	43.22 (1.67)	Mean (SD)	g/kg	Peyton et al. (2016)
R	NO3-N	3.98 (0.01)	Mean (SD)	g/kg	Peyton et al. (2016)
R	NH4-N	3.85 (0.29)	Mean (SD)	g/kg	Peyton et al. (2016)
R	Organic-N	39.37 (1.96)	Mean (SD)	g/kg	Peyton et al. (2016)
R	Dry matter	25.00 (0.10)	Mean (SD)	%	Peyton et al. (2016)
S	TP	25.19 (0.61)	Mean (SD)	g/kg	Peyton et al. (2016)
S	WEP (dry)	302.00 (1.00)	Mean (SD)	g/kg (dry)	Peyton et al. (2016)
S	TN	54.58 (1.53)	Mean (SD)	g/kg	Peyton et al. (2016)
S	NO3-N	4.24 (0.04)	Mean (SD)	g/kg	Peyton et al. (2016)
S	NH4-N	3.43 (0.24)	Mean (SD)	g/kg	Peyton et al. (2016)
S	Organic-N	51.15 (1.78)	Mean (SD)	g/kg	Peyton et al. (2016)
S	Dry matter	24.00 (0.20)	Mean (SD)	%	Peyton et al. (2016)
T	TP	17.11 (0.19)	Mean (SD)	g/kg	Peyton et al. (2016)
T	WEP (dry)	493.00 (26.00)	Mean (SD)	g/kg (dry)	Peyton et al. (2016)
T	TN	51.45 (2.90)	Mean (SD)	g/kg	Peyton et al. (2016)
T	NO3-N	1.15 (1.00)	Mean (SD)	g/kg	Peyton et al. (2016)
T	NH4-N	0.57 (0.03)	Mean (SD)	g/kg	Peyton et al. (2016)
T	Organic-N	50.87 (2.88)	Mean (SD)	g/kg	Peyton et al. (2016)
T	Dry matter	87.00 (0.10)	Mean (SD)	%	Peyton et al. (2016)
U	TP	3.94 (0.40)	Mean (SD)	g/kg	Peyton et al. (2016)

Sample Description Code	Nutrient	Concentration	Statistic	Units*	Ref.
U	WEP (dry)	9.00 (0.30)	Mean (SD)	g/kg (dry)	Peyton et al. (2016)
U	TN	17.62 (0.40)	Mean (SD)	g/kg	Peyton et al. (2016)
U	NO3-N	2.92 (0.01)	Mean (SD)	g/kg	Peyton et al. (2016)
U	NH4-N	0.45 (0.03)	Mean (SD)	g/kg	Peyton et al. (2016)
U	Organic-N	17.17 (0.40)	Mean (SD)	g/kg	Peyton et al. (2016)
U	Dry matter	34.00 (0.20)	Mean (SD)	%	Peyton et al. (2016)

*Converted where possible to facilitate comparison. †Range of mean concentrations reported in the article included in the table.
ØNAIP – Non-apatite inorganic phosphorus, AP – Apatite inorganic phosphorus, PRP – Particulate reactive phosphorus, PNRP – Particulate non-reactive phosphorus, SRP – Soluble reactive phosphorus, SNRP – Soluble non-reactive phosphorus, Plab - labile P forms.

Sample Description Code	Sample Description
A	Anaerobically digested biosolids (processed for the agricultural and horticultural markets by dewatering, centrifugation, and compression)
B	Liquid digested biosolids
C	Liquid thickened biosolids
D	Anaerobically digested biosolids
E	Anaerobically digested and thermically stabilised biosolids
F	Thickened, filtered and limed biosolids
G	Anaerobically-digested biosolids
H	Anaerobically-digested biosolids – Solid fraction (sewage sludge as mono-substrate or dominant co-substrate)
I	Anaerobically-digested biosolids – Liquid fraction (sewage sludge as mono-substrate or dominant co-substrate)
J	Dewatered biosolids cake
K	Liquid anaerobically-digested biosolids
L	Biosolids (digested cake)
M	Anaerobically-digested dewatered biosolids
N	Anaerobically-digested, dewatered, and thermally treated biosolids
O	Anaerobically-digested biosolids
P	Lime stabilised biosolids
Q	Composted biosolids
R	Anaerobically-digested biosolids
S	Anaerobically-digested biosolids
T	Anaerobically-digested and thermally dried biosolids
U	Lime stabilised biosolids

Nutrient transformations, accumulation and transportation

Phosphorous

Results from P fractionation completed by Nicholson et al. (2018) showed that biosolids additions significantly increased the amount of inorganic P readily available for plant uptake, and the amount of moderately available organic P (i.e. extractable with sodium hydroxide, NaOH) which can be mobilised by soil microbes over time. Thus, demonstrating that when applied to soil biosolids can provide short-term and long-term sources of soil P for uptake by plants.

However, due to the effect of wastewater treatment processes (especially the addition of metal salts and lime), P in biosolids are less plant-available and less mobile than fertiliser or manure P (García-Albacete et al. 2012). Al and Fe in particular have a decisive influence on WSP and subsequently the runoff behaviour of nutrient sources applied to land (García-Albacete et al., 2012). Penn and Sims (2002), for example, found biosolids containing large amounts of Fe had the lowest amounts of WSP and produced the lowest concentrations of dissolved P in runoff when applied to soils. García-Albacete et al. (2012) found that the amount of Non-apatite inorganic phosphorus (NAIP) – considered to be the most labile form of P - depended directly on the Fe content in all biowastes, including biosolids, analysed (except for pig slurry).

Shober and Sims (2007), cited in García-Albacete et al. (2012), suggest that lower P solubility in biosolids is due to the precipitation of metal hydroxides and the sorption of phosphate, or the co-precipitation of both. Other authors have concluded that the solubility of biosolids P was mainly controlled by the amount and type (Al, zinc and manganese, Mn) of metal present (He et al., 2010 in García-Albacete et al., 2012).

Organic ligands from partial decomposition of biosolids may form complexes with Al, Fe, and Ca, making P more available (Stevenson, 1982 in Akhtar et al. 2002). García-Albacete et al. (2012) found a clear relationship between WSP and the total concentration of Al + Fe + Ca in biowastes.

Though much of the P within biosolids may be strongly bonded to Al or Fe (Eldridge et al. 2009), after being spread, mineral forms of P are also likely to link to the Al or Fe present in soil (Chang et al. 1983 in Quilbé et al. 2005) whilst organic forms are rapidly hydrolysed, both in basic or acid soils (Hinedi et al. 1989 in Quilbé et al. 2005).

Heathwaite et al. (2006) report that the majority (66 – 75%) of the P mobilised in drain flow from the biosolids-amended field investigated was transported in the unreactive fraction in association with soil particles. Similar subsurface flux pathways for agricultural soils receiving livestock manures and slurries have also been reported elsewhere (Heathwaite and Dils, 2000; Heathwaite et al., 2005[27]).

A comparison of P loss in surface runoff following application of different P amendments to a dispersive silty clay loam soil (arable land), by Withers et al. (2001), indicated that the risk of P transfer to watercourses from agricultural land when biosolids (liquid or cake form) are applied is less than when cattle manure (liquid) is applied, owing to biosolids lower P solubility. Results from this study indicate negligible release of dissolved P from biosolids, with no increase in the transfer of dissolved P following their application. P was mainly lost under particulate form in runoff, with similar findings reported at sites elsewhere with different soil cover (grassland; Quilbé et al., 2005).

Nevertheless, Vanden Bossche et al. (2000) report that application of biosolids had no effect on the particulate-bound P content in runoff water, but induced a significant augmentation of dissolved P concentrations in the runoff. These authors indicate that dissolved P is released from biosolids as well as soil particles, thus biosolids enhance lability of soil-bound P.

Elliott et al. (2005) compared P levels in runoff losses from soils amended with different biosolids; concluding that the P concentration in runoff from biosolids-amended soils depends on the type of wastewater and solids processing methods used to generate the biosolids. Owing to the addition of Al and/or Fe during wastewater treatment or through solids processes like heat-drying, runoff P losses were not statistically different from unamended soils, supporting the notion that the solubility of P in the organic amendment has an important influence on the potential for P migration. Nevertheless, it is recognised that for biosolids (and treated manures) the susceptibility of P leaching and runoff is so variable that the ability to accurately predict site vulnerability risk is compromised (Elliott et al., 2005).

[27] The significance of the colloidal fraction as a vehicle for P transport in subsurface pathways in grassland soils is reported in this study.

Nitrogen

The amount of 'mineralisable N' (biological measure of soils' capacity to supply N[28]) in biosolids depends on the treatment process (i.e. extent of biological stabilisation) and can vary across climatic regions (Rigby et al., 2016). After the application of biosolids, N can be volatilised or rapidly mobilised by runoff and leaching (Gangbazo et al., 1995 in Quilbé et al., 2005). Whereas NO_3 transfers occur only after long-term nitrification process in soil (Serna and Pomares, 1992 in Quilbé et al., 2005).

Nicholson et al. (2018), found that concentrations of potentially mineralisable N increased at some of the sites investigated following the application of biosolids. Quilbé et al. (2005) observed that biosolids had no significant effect on the N concentration in runoff water, reasoning that this occurred because the N in the biosolids was only under organic form, which is less available for runoff than NH_4 or NO_3.

Peyton et al. (2016) found that runoff from biosolids-amended plots had the same NO_3–N concentrations as the study control, whilst elevated concentrations of NH_4-N were recorded for biosolids-amended plots. Compared with the four different biosolids investigated, runoff from the dairy cattle slurry amended-plot had the highest losses of NH_4-N and DRP. These authors (Peyton et al., 2016) suggest this is because nutrients were more easily mobilised following an episodic rainfall event on plots amended with dairy cattle slurry compared with biosolids.

Soil characteristics

The texture of the soil to which biosolids is applied can strongly influence the fate of P. Akhtar et al. (2002) report that the sandy Thurman soil used in their experiments maintained a significantly higher concentration of soluble P owing to its coarse texture and relative lack of retention sites for P. Blume et al. (2010), cited in Vogel et al. (2017), noted that sandy soils are at higher risk of P losses owing to their low silt and clay content.

Soil pH is also influential. Gustafsson et al. (2012), cited in Vogel et al. (2017), found that in neutral to acidic soils, P is mainly adsorbed to Al- and Fe-oxide, whilst, according to Psenner et al. (1984), cited in Tuszynska et al. (2021), the greatest mobility of P with the highest bioavailability potential in soil (P in organic compounds and associated with Al, Fe, magnesium (Mg) and Mn oxides and hydroxides) occurs in soils with a pH in the range 5–7. Nevertheless, other authors have stated that strong fixation to Al- and Fe-oxides/ hydroxides can decrease the bioavailability of P (Vogel et al. 2017).

Spreading and storage methods

Deeper incorporation of biosolids into the soil may reduce exports by limiting erosion and favouring stabilisation of P in soil (Kladivko and Nelson, 1979 in Quilbé et al., 2005).

[28] Through mineralisation of soil organic N reserves to ammonium-N, which can subsequently be converted to nitrate-N (by nitrification processes) (Nicholson et al. (2018)

However, it is likely that such practices would increase sediment, and subsequently particulate P losses in runoff (Mostaghimi et al., 1992 in Quilbé et al., 2005).

Akhtar et al. (2002) found lower concentrations of biologically available P in sludge-amended soil when temperature was increased (from 25 to 37 degrees Celsius). The observed effect suggests that mid-season sludge applications, once the soil has warmed, may reduce the potential for the undesirable loss of available P (Akhtar et al., 2002).

Other factors impacting bioavailable phosphorus downstream from manure sources

Peyton et al. (2016) found that despite substantial differences in the initial NH_4-N concentrations between two of the biosolids investigated, similar losses were recorded. These authors suggest this occurred as a result of differing consistency and subsequently surface area exposure to rainfall, such that biosolids with finer particle granulated consistency could possibly be easier diluted and transported in runoff.

Mitigations measures to reduce phosphorus transport to freshwater

The release of P to runoff water depends on both biosolids type and the soil. Withers et al. (2016) suggest that biosolids could be more sustainably managed by matching applications to soil type and P fertility status. Incubation studies are an example of a suitable tool to obtain robust information on plant P availability and the risks of P losses (Nanzer et al., 2014).

Results reported by Akhtar et al. (2002) suggest that bioavailability is maximized soon after sludge application and the threat of biologically available P in runoff decreases with time such that ‘short-term’ efforts to keep amended soil from entering surface watercourses are most appropriate.

Hodgkinson et al. (2002) suggested that an additional factor which may have reduced losses from the solid manures investigated was that they were ploughed in after application. Noting that the mixing of manures with the soil provides some protection against mobilisation and disrupted continuity of macropores and fissures created by drainage installation.

Green waste and food waste

Introduction

Biowaste is defined in the EU Landfill Directive as waste capable of undergoing anaerobic or aerobic decomposition such as food and garden waste, and paper and cardboard. The application of biowastes to land has become an attractive alternative to disposal via landfill/ incineration, as they can be a good source of organic matter and nutrients such as N and P (García-Albacete et al., 2012).

Information relating to 'green waste' (municipal garden waste or crop residue) and selected food wastes are reported on in this section.

Typical nutrient concentrations

Extracted from the literature reviewed in this study, the table below (Table 13) shows concentrations of P, N and dry matter, including fractions of P, in green waste and selected food wastes [29].

Table 13. Concentrations of P, N and dry matter in green waste and food waste from the literature[29]

Sample Description Code	Nutrient	Concentration	Statistic	Units	Ref.
A	TN	0.01	-	g/kg	Malik et al. (2012)
A	TP	0.56	-	g/kg	Malik et al. (2012)
A	WSP	0.18	-	g/kg	Malik et al. (2012)
B	TN	0.02	-	g/kg	Malik et al. (2012)
B	TP	2.09	-	g/kg	Malik et al. (2012)
B	WSP	1.63	-	g/kg	Malik et al. (2012)
C	TP	3.30 – 4.10	Range	g/kg (d.w)	Tuszynska et al. (2021)
C	TN	0.40 – 0.50	Range	g/kg (d.w)	Tuszynska et al. (2021)
C	Organic P	0.10 (± 0.01)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
C	NAIP	1.20 (± 0.03)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
C	AP	1.80 (± 0.07)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
C	Labile P contribution	0.39 – 0.44	Range	Plab: TP	Tuszynska et al. (2021)
C	Dry matter	17.70 – 24.60	Range	%	Tuszynska et al. (2021)
D	TP	0.20 – 0.30	Range	g/l	Tuszynska et al. (2021)
D	TN	1.40 – 1.60	Range	g/l	Tuszynska et al. (2021)
D	NH4	1.30 – 1.60	Range	g/l	Tuszynska et al. (2021)
D	PRP	0.20 (± 0.05)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
D	PNRP	0.03 (± 0.01)	Mean	g/kg (d.w)	Tuszynska et al. (2021)

[29] Only data from selected articles reviewed, and selected forms of P and N, has been included in the table. Only data from articles within which the details of waste type were found (i.e. confirming samples taken were green waste or food waste) are included.

Sample Description Code	Nutrient	Concentration	Statistic	Units	Ref.
D	SRP	0.94 (± 0.08)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
D	SNRP	0.14 (± 0.01)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
D	Labile P contribution	0.87 – 0.91	Range	Plab: TP	Tuszynska et al. (2021)
E	TP	17.10 – 18.00	Range	g/kg (d.w)	Tuszynska et al. (2021)
E	TN	9.60 – 18.80	Range	g/kg (d.w)	Tuszynska et al. (2021)
E	Organic P	0.50 (± 0.05)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
E	NAIP	6.00 (± 0.10)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
E	AP	9.50 (± 0.10)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
E	Labile P contribution	0.34 – 0.41	Range	Plab: TP	Tuszynska et al. (2021)
E	Dry matter	15.50 – 27.20	Range	%	Tuszynska et al. (2021)
F	TP	0.50 – 0.54	Range	g/l	Tuszynska et al. (2021)
F	TN	2.10 – 2.20	Range	g/l	Tuszynska et al. (2021)
F	NH4	1.90 – 2.00	Range	g/l	Tuszynska et al. (2021)
F	PRP	0.14 (± 0.06)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
F	PNRP	0.09 (± 0.01)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
F	SRP	2.36 (± 0.27)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
F	SNRP	0.22 (± 0.03)	Mean	g/kg (d.w)	Tuszynska et al. (2021)
F	Labile P contribution	0.88 – 0.92	Range	Plab: TP	Tuszynska et al. (2021)
G	TP	1.10	-	g/kg	Chambers et al. (2002)
G	TN	7.80	-	g/kg	Chambers et al. (2002)
G	Dry matter	37.40	-	%	Chambers et al. (2002)
H	TP	2.10	-	g/kg	Chambers et al. (2002)
H	TN	13.70	-	g/kg	Chambers et al. (2002)
H	Dry matter	74.40	-	%	Chambers et al. (2002)
I	IP	2.70 (± 0.14)	Mean (SD)	g/kg	García-Albacete et al. (2012)
I	OP	0.23 (± 0.02)	Mean (SD)	g/kg	García-Albacete et al. (2012)
I	NAIP	0.26 (± 0.02)	Mean (SD)	g/kg	García-Albacete et al. (2012)
I	AP	1.61 (± 0.14)	Mean (SD)	g/kg	García-Albacete et al. (2012)
I	TP	3.80 (± 0.31)	Mean (SD)	g/kg	García-Albacete et al. (2012)
I	WSP i	0.32 (± 0.02)	Mean (SD)	g/kg	García-Albacete et al. (2012)
I	WSP t	0.33 (± 0.02)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	IP	5.99 (± 0.33)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	OP	1.677 (± 0.01)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	NAIP	3.88 (± 0.31)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	AP	1.97 (± 0.17)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	TP	8.22 (± 0.62)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	WSP i	0.84 (± 0.06)	Mean (SD)	g/kg	García-Albacete et al. (2012)
J	WSP t	0.90 (± 0.05)	Mean (SD)	g/kg	García-Albacete et al. (2012)
K	Total P*	2.01	Mean	g P/kg	Requejo & Eichler-Löbermann (2014)
K	IP**	1.83 (± 0.18)	Mean (SD)	g P/kg	Requejo & Eichler-Löbermann (2014)
K	OP**	0.25 (± 0.04)	Mean (SD)	g P/kg	Requejo & Eichler-Löbermann (2014)
K	Dry matter	50.10	Mean	%	Requejo & Eichler-Löbermann (2014)
L	Resin P	1239.00	-	mg/kg	Kelley et al. (2020)
L	TN	22.50	-	mg/g (dry)	Kelley et al. (2020)
L	N-NO3	680.00	-	mg/kg	Kelley et al. (2020)
L	N-NO4	20.00	-	mg/kg	Kelley et al. (2020)

*Aqua regia extractable   **NaOH-EDTA extractable P.

The

Sample Description Code	Sample Description
A	White lupin, LP residue
B	Faba bean, HP residue
C	Agricultural lignocellulosic waste – Solid fraction of digestate (corn silage as mono-substrate or dominant co-substrate)
D	Agricultural lignocellulosic waste – Liquid fraction of digestate (corn silage as mono-substrate or dominant co-substrate)
E	Fruit and vegetable waste (as mono-substrate or dominant cosubstrate) – Solid fraction of digestate
F	Fruit and vegetable waste (as mono-substrate or dominant cosubstrate) – Liquid fraction of digestate
G	Waste peat/ compost (peat and peat-based compost from bags which had failed to meet weight specification, and are normally tipped into the landfill)
H	Composted green waste
I	Grape pomace (compost)
J	Food manufacturing dewatered sludge
K	Green garden and landscape residues (Compost)
L	Food waste compost

Nutrient transformations, accumulation and transportation

Organic residues typically contain considerable amounts of soluble inorganic P, which contributes to the fast release of P after incorporation into soil (Requejo and Eichler-Löbermann, 2014).

Malik et al. (2012) found that, compared with inorganic P addition, P added with crop residues (White Lupin, Faba Bean) was less prone to sorption and precipitation owing to lower concentration of WSP and stimulation of microbial activity (by addition of carbon).

Composted green waste and biosolids additions increased topsoil extractable P concentrations on Winterton clay by up to 32 mg/l and 33 mg/l, respectively (Chambers et al. 2002).

Requejo and Eichler-Löbermann (2014) found that application of composted green garden and landscape residues, in combination with crop P uptake, did not affect the double lactate-extractable P concentration in soil (over a 14-year period); concentrations dropped significantly in soil with cattle manure applied. No differences were, however, found between treatments regarding P-sorption capacity.

In a laboratory incubation, Kelley et al. (2020) found that food waste compost increased NO_3 concentrations in both sandy soils investigated, whilst soils amended with food waste compost had significantly higher TN and resin-extractable P relative to the control.

In the study by Chambers et al. (2002), readily mineralisable N concentrations in soil were increased by composted green waste (56 kg/ha N) and waste peat compost (28 kg/ha N), though the addition from biosolids was substantially greater (87 kg/ha N).

Soil characteristics

Soil microorganisms play an important role in transformation of P by uptake into and release from the microbial biomass, solubilisation of inorganic P and formation and mineralisation of organic P (Malik et al., 2012).

Spreading and storage methods

When comparing non-industrial with industrial food waste composts, Kelley et al. (2020) found that less TN was recovered as plant-available N for the latter. The authors suggest that this may be due to the presence of wood residues in the industrial compost, but could alternatively have occurred due to longer composting and/or storage periods for industrial products, especially if piles were not protected from rainfall (Maltais-Landry et al., 2018 in Kelley et al., 2020).

Curtis et al. (2009) examined nutrient losses in surface runoff from potential types of compost (windrowed yard waste) treatment applied on constructed slopes. Losses of P were lowest with grass cover or improved infiltration, whilst highest losses occurred with compost blankets over bare, untilled soil. The presence of improved infiltration (treatments with tillage) or grass cover was also associated with reduced losses of NO_3 and NH_4 in surface runoff. Curtis et al. (2009) also found that treatments that had compost tilled into the soil lost fewer nutrients in surface runoff than treatments where compost was applied as a mulch.

Other factors impacting bioavailable phosphorus downstream from manure sources

Compost processing conditions including moisture, oxygen availability and additional processing (e.g. vermicomposting, pelletisation) may affect nutrient content and other properties (Kelley et al., 2020).

Mitigations measures to reduce phosphorus transport to freshwater

It was concluded by Curtis et al. (2009) that applying compost to constructed slopes as a surface blanket in combination with having compost tilled into the soil, rather than one of these treatments in isolation, would achieve a greater reduction in nutrient (P, NO_3 and NH_4) losses.

Interviews with academics

Notes taken during each of the three interviews held with academic experts working in this field to discuss recent advances in research and to discuss preliminary findings from the review are provided in Appendix D.

4. Discussion

Key factors determining the extent to which manures, spread on agricultural land, increase the concentration of bioavailable P in downstream freshwater environments (RQ1)

Highly variable spatial and temporal patterns of P delivery, retention of different soluble and particulate forms of P by soil, and complex lag patterns of P release from legacy P stores, mean that it is unreasonable to think that a connection can be made between manure applied and the impact on water quality at a particular location (Haygarth, 2024; Appendix D.1). Literature presents studies which have applied manure at a particular time in a particular location, and it is important that this is recognised when interpreting findings (Haygarth, 2024; Appendix D.1).

Manure type, soil type, manure storage and manure spreading methods are discussed in the following sections as some of the factors that can influence the loss of manure P to surface waters from agricultural land. It is, nevertheless, important to acknowledge that additional site-specific factors (including climate and catchment characteristics) also influence mobilisation and delivery of P to receiving waters.

Manure type

Manure production

Latest (2023) estimates of livestock numbers (Welsh Government, 2023) and manure, N and phosphate production figures obtained from Schedule 1 of the CoAPR were used to calculate the amount of manure and manure phosphate/ N produced. As shown in Figure 13, in 2023, livestock manure production in Wales, and manure N and phosphate production, was dominated by cattle (70%, 63% and 60%, respectivly); followed by sheep (28%, 33%, 32%) and poultry (2%, 4%, 8%), with negligible amounts from pigs (Figure 13)[30].

Data on livestock numbers (cattle, sheep and poultry only)[31] were also available for the period 2007 – 2022. However, this data was at a lower resolution than that available for 2023, in that a 'complete' categorised breakdown (e.g. for cattle, by sex and type i.e. dairy/ beef) was not provided. Specific manure, N and phosphate production figures are available for the majority of these categories in Schedule 1 of the CoAPR, such that more refined calculations for manure production could be completed for 2023. Nevetheless, calculated

[30] Assumptions regarding the most representative/ comparable livestock types between the two datasets were made when producing Figure 13.

[31] Received in email from Dave Johnston (NRW, 12/02/2024).

values for the period 2007 – 2022 (using total livestock numbers and average production figures, presented in Appendix E) also show that manure production in Wales, and manure phosphate and N production, was dominated by cattle, followed by sheep and poultry[32].

It should be noted that the figures presented Appendix E (and in Figure 13) are for the whole of Wales, and there may be significant regional and/or catchment-scale differences in manure N and P production.

Withers et al. (2022) reported that in the Wye catchment, poultry has now overtaken cattle as the main producer of manure P (Withers et al., 2022). Powys is the largest county in Wales and covers nearly all of the Welsh part of the Wye catchment; calculated values for Powys (across the period 2007 – 2022, presented in Appendix F) indicate that although the contribution of poultry to manure phosphate production has increased in recent years (to a high of 19% in 2022), contributions from sheep and cattle are still more dominant (52% and 30% in 2022, respectively). It is important to note that calculations in this study used different sources of information for livestock numbers and covered different area extents.

[32] Bearing in mind that pigs have not been accounted for (as data on numbers was not provided for this period).

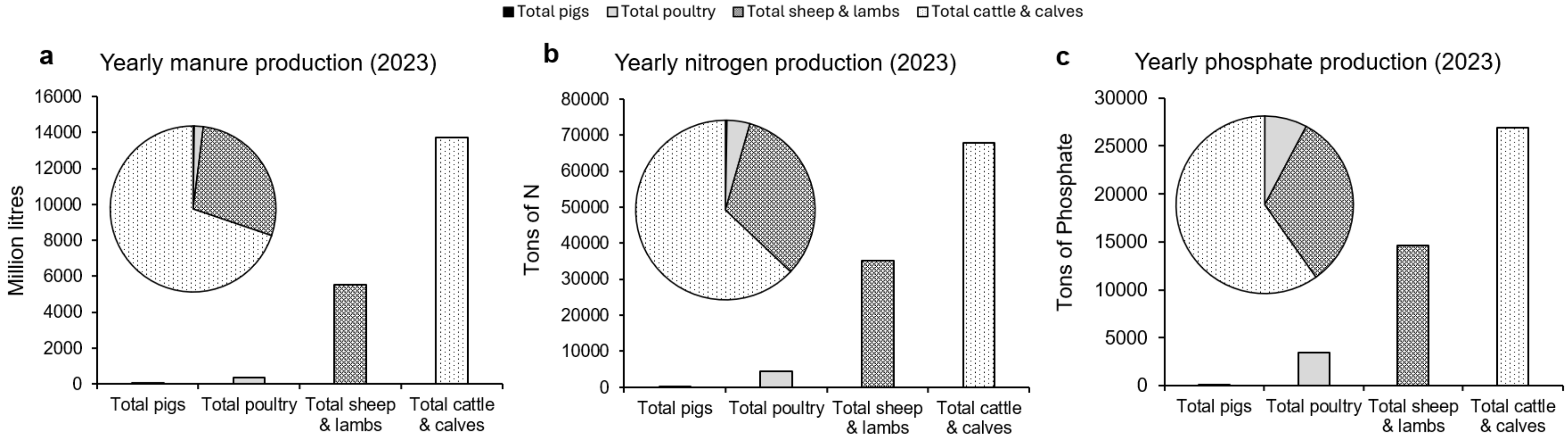


Figure 13. Welsh summary statistics (2023) - (a) yearly manure production, (b) yearly nitrogen production and (c) yearly phosphate production. Livestock numbers were obtained from June 2023 survey of agriculture and horticulture (Welsh Government, 2023) and manure, N and P production figures per animal were obtained from Schedule 1 of the CoAPR. Note, inputs from pigs are negligible and thus are not visible in the figure.

Manure content

Figure 14 summarises the content of TP and WSP/ WEP (as well dry matter and TN) for different manure types from the literature reviewed (whilst the individual numbers may not be directly comparable, owing to differences in methodologies between studies etc., they give a broad indication of variability between manure types).

In general TP is lower in the manure of ruminant animals (cows and sheep) and higher in non-ruminant animals (poultry and swine), as shown in Figure 14. This is because ruminants are able to extract organically-bound P from plant feeds, whereas non-ruminant animals are unable to as they lack the phytase enzyme in their digestive systems (Chowdhary et al., 1996; Sharpley and Moyer, 2000; Pagliari and Lomboski, 2012). As for TP, TN concentrations were typically higher in poultry manure and pig manure/ slurry compared with the manure of ruminant animals (cows and sheep) (Figure 14).

Though biosolids may have higher TP concentrations than other manures, the latter generally have higher WSP/ WEP concentrations, as shown in Figure 14. The lower WEP in biosolids has been attributed to elevated Al and Fe content from chemical additions during wastewater treatment and solids dewatering processes (Brandt et al., 2004).

The form of P in the manure added to land is an important factor controlling mobilisation and bioavailability. When manures are spread on the soil surface, application of soluble P in manure can directly contribute to the soluble P in runoff (Preedy et al., 2001; Kleinman et al., 2002). Differences in P solubility between manure types can, therefore, be an important factor impacting the amount of dissolved P that is lost in runoff when manure is added (DeLaune et al., 2004). As a consequence, manures which contain less WSP/ WEP may represent a lower threat to surface waters from leaching/ runoff (Jalali et al., 2022).

Based on the data reported in Figure 14, poultry litter/ manure and pig manure/ slurry likely represent a greater potential risk to surface water from runoff than dairy manure and biosolids owing to the higher WSP/ WEP content.

Nevertheless, though WSP/ WEP was generally higher in poultry litter/ manure and pig manure/ slurry compared with cow manure/ slurry (see Figure 14), significant variability exists with contrasting trends reported in studies which have investigated multiple manure types. For example, Brandt et al. (2004) found WEP was highest in dairy manure, followed by poultry manure (and biosolids).

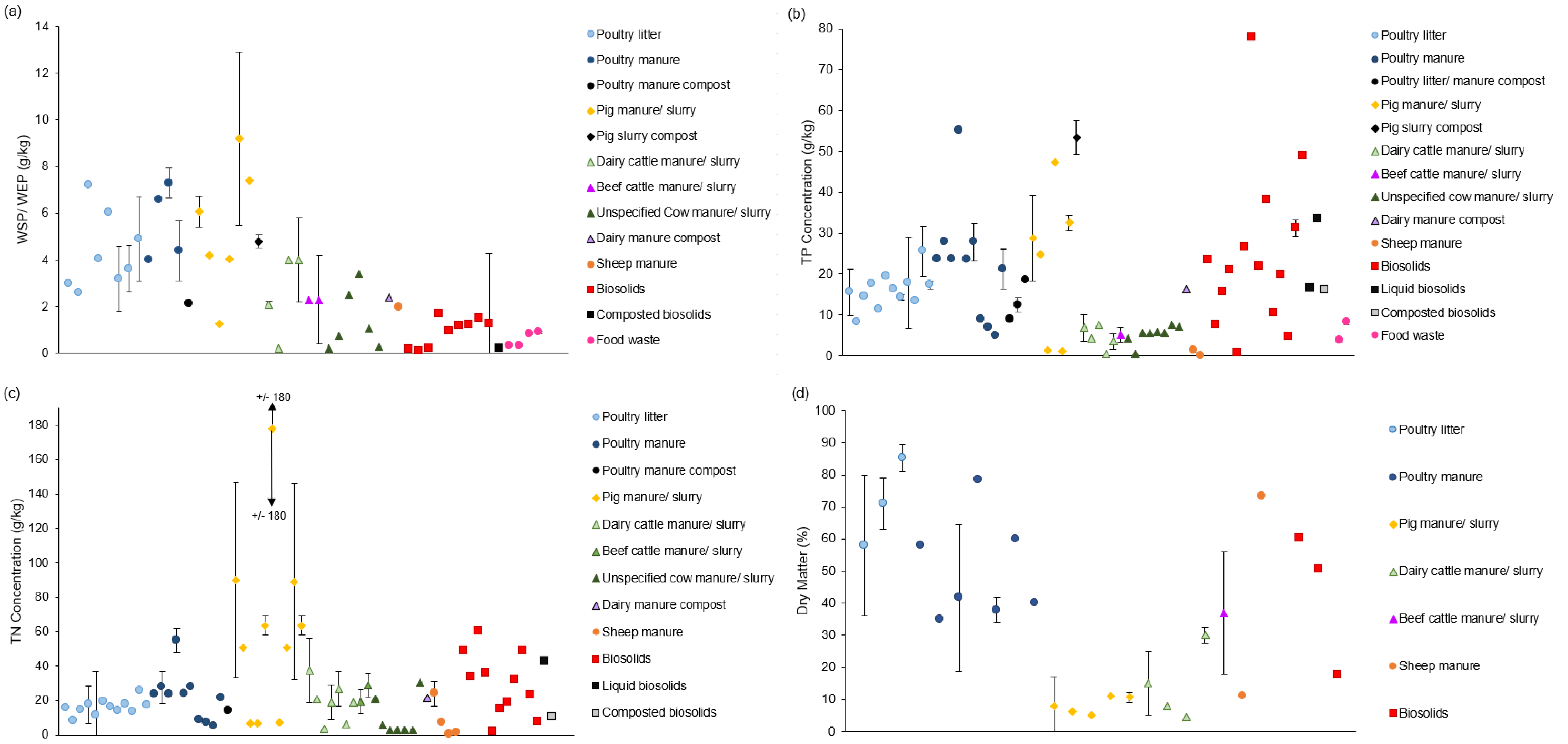


Figure 14. Manure (a) WEP/ WSP, (b) TP, (c) TN and (d) dry matter content[33,34]

[33] Selected studies from the literature are included in the figure. Selected data from these studies has been included. Values may not be directly comparable owing to different methodologies employed by the authors.
[34] Categories of manure types are represented in the legends for the charts.

Soil type

Soil type is determined by physical properties (such as grain size/texture and moisture content) and chemical properties (such as pH) which can influence the transport and ultimately loss of manure P to surface waters.

Chemical properties

In acid soils, P is fixed into largely insoluble forms by precipitation and sorption reactions and with Al and Fe compounds as well as amorphous and crystalline colloids (Pizzeghello et al., 2011). In (alkaline) calcareous soils, surface adsorption and precipitation are the key processes which reduce the availability and mobility of P. For soils rich in carbonates ($CaCO_3$), P solubility may be controlled by the formation of Ca-P surface complexes.

Inorganic P released from organic manures will interact with soil by adsorbing to the surfaces of clays or organic particles, or via precipitation with Al oxides, Fe oxides, or $CaCO_3$ compound (Anderson and Xia, 2001). The addition of organic manures can therefore reduce the soil P sorption capacity. As P loss in runoff or by leaching is higher in soils with low P sorption capacity (Jalali et al., 2022; Hooda et al., 2000), soils amended with organic manures may be more prone to P loss.

It has been reported that the influence on soil P sorption capacity is dependent on the P source. Siddique and Robinson (2004), for example, noted a smaller decline in soil P sorption capacity for biosolids and poultry litter, compared with cattle slurry, attributing this to lower P solubility and the formation of new adsorption sites for P (Siddique and Robinson, 2004). Slower rates of P desorption in the litter and biosolids (compared to cow manure) were attributed to relatively large concentrations of Ca which may have increased P sorption strength through the formation of Ca-P complexes at the surface of alumino-silicate clays (Siddique and Robinson, 2004).

The P sorption sites within soils may also be occluded by organic matter derived from the manures (Anderson and Wu, 2001). When organic manures are added to the soil they may include organic acids, or organic acids may be introduced as the manure is decomposed by microbes within the soil. The anions of organic acids can compete for P sorption sites within the soil, thus decreasing soil P sorption capacity (Marshal and Laboski, 2005). Microbes therefore also play a key role in soil P dynamics by P uptake, solubilisation and mineralisation. It has been suggested that a better understanding of the relationship between type of P amendment, microbial activity and changes in soil P pools is important for a better management of soil P (Malik et al., 2012).

Soil phosphorus saturation rations

Soil P saturation rate (PSR) is a molar ratio of P to Al and Fe, originally determined for sandy soils using oxalate extraction (Breeuwsma et al., 1995), but subsequently extended to other soil types (Kleinman et al., 2003). Studies have identified a PSR threshold, above which added P is typically lost via runoff or leaching (Casson et al., 2006). The PSR threshold depends on soil physicochemical properties, geographic location, land use, and soil management practices (Xu et al., 2020).

The degree of P saturation (DPS) within the soil is closely correlated to the TP concentrations within leachate waters, with studies suggesting a low chance of P loss from soils with a DPS <25%, and increased risk of leaching above this threshold value (Maguire et al., 2002; McDowell and Sharpley, 2001).

Physical properties

Soil texture

The texture of the soil to which organic manures are applied can strongly influence the fate of P. Sorption of P is highly correlated with clay content (Börling et al., 2001 in Pizzeghello et al., 2011). Sandy, well drained soils are particularly prone to P leaching due to limited P retention capacity (Chrysostome et al., 2007; Jalali et al., 2022). Studies suggest that in very sandy soils, P may move through the sediment profile to depths of over 1m (Hubbard et al., 2004 and references therein). For non-sandy soils, the leaching of P with percolating water is much lower. Olsen and Watanabe (1970) suggested there was an eight-times higher risk of P pollution entering groundwaters in sandy compared to clay-rich soils.

A 20-year study on the fine-loamy calcareous soils of the Upper Midwestern USA found that poultry litter application did not increase the subsurface transport of nutrients in these soils (Hoover et al., 2019), which was attributed to the high capacity of this type of soil to adsorb P (Hoover et al., 2015). Similarly, the clay rich (34%) loam soils of Hillsborough (Northern Ireland) had a high capacity to adsorb P, and even in areas with 30 years of heavy fertilisation with organic manures (cow and pig), soils retained substantial capacity to adsorb more P (Anderson and Wu, 2001).

Permeability and soil moisture content

Low permeability soils (with heavy texture) promote low rates of decomposition; therefore, manure application rates should be lower than for high permeability soils (coarse textured soil) which promote rapid decomposition (Xie and MacKenzie, 1986).

Increased runoff is observed from soils with higher moisture contents (McConnell et al., 2016), and the application of manures to drier soil has been shown to help reduce losses in runoff (Torbert et al., 1999).

Though deep seepage of P in soils is relatively negligible in clay rich soils, slurry liquid P may infiltrate deeper into sandier soils or soils with extensively connected, larger pores (Vadas, 2006). Vadas (2006) noted that immediate infiltration of slurry P into the soil following application reduces the availability of P at the surface that can be transported in surface runoff following a rainfall/ runoff event.

Other authors have suggested that soil macropores may facilitate the preferential flow of soluble P (adsorbed to particulate matter) (Geohring et al., 2001, Sharpley et al., 2003 both cited in Galbally et al., 2013).

Soil moisture was identified as an important factor controlling N mineralisation and denitrification processes and N uptake by plants within the vadose zone (soil above groundwater level) (D'odorico et al., 2003).

Land use management practices

Soil type can dictate the agricultural practices that are undertaken on the land and in turn this may influence P loss. For example, Withers et al. (2000) estimated that ~50% of the agricultural land in England and Wales (at the time of publication) had been under-drained to correct surface wetness problems and/or allow access to the field during autumn. The subsurface flow of P at shallower depths may represent a risk to surface water quality, particularly if this intercepts with agricultural drainage systems (Anderson and Xia, 2001). A UK field-based study concluded that liquid livestock manures should not be applied to recently drained clay soils to avoid excessive P enrichment in drainage waters (Hodgkinson et al., 2002).

Soupir et al., (2006) found that soils which have had previous applications of manure, had high TSS in their runoff which meant greater nutrient transport with the particles.

How do the soil types of Wales influence the processing and transport of manure P? (RQ1b)

Wales has 183 different soil series that vary in their chemistry, biology and physical attributes (Welsh Government, 2022). The most common soil types in Wales are presented in Figure 15 alongside land cover classes from 2019 (Welsh Government, 2022), illustrating that arable farming is typically associated with brown soils in the lowlands, whereas improved grassland is associated with brown soils, podzols and surface-water gley soils. Together these three soil types cover the majority of Wales.

Table 14 provides a description of these soil types, along with their key characteristics and the impacts these may have on the processing and transport of manure P (identified from the literature reviewed).

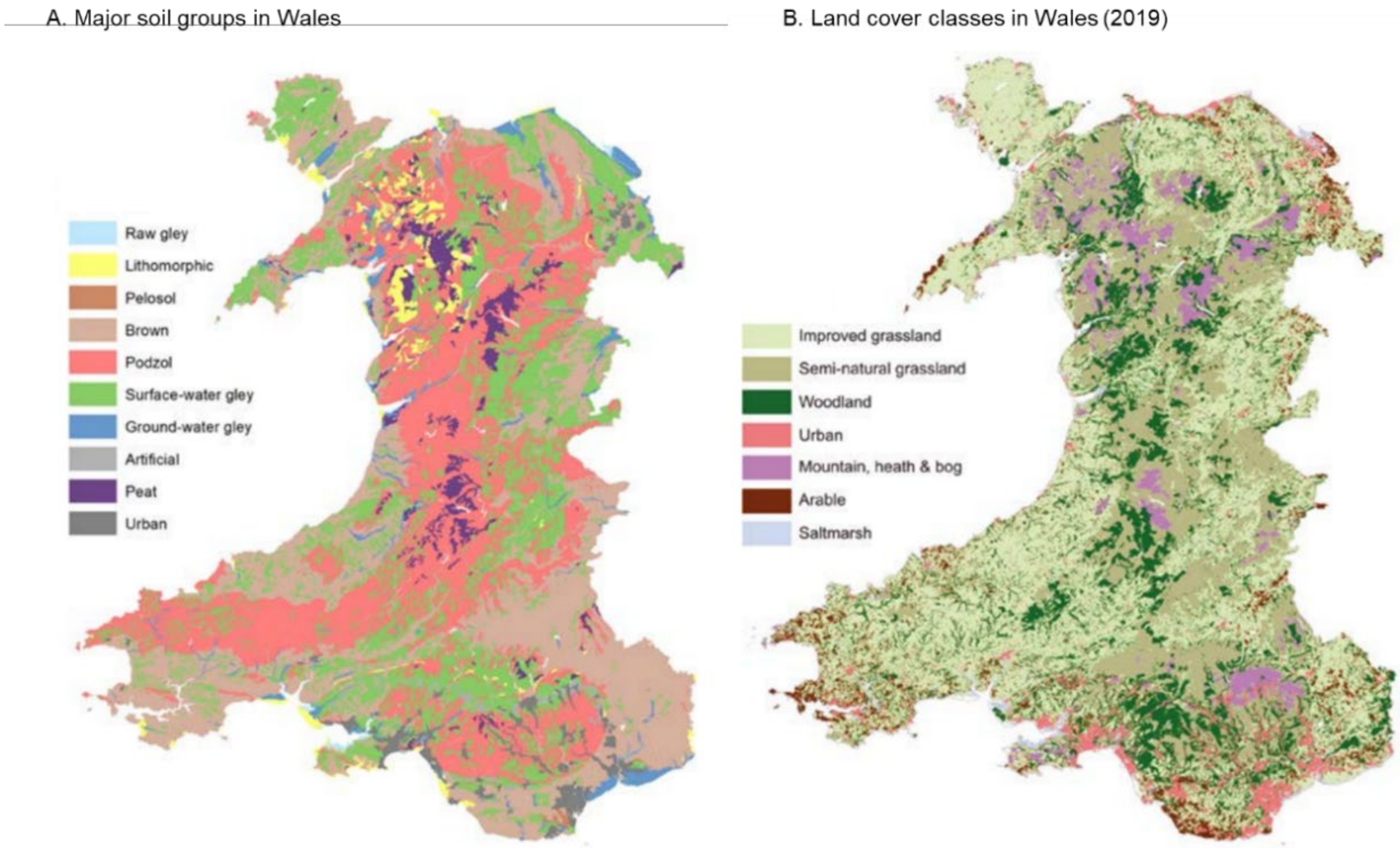


Figure 15. (A) the major soil groups in Wales and (B) the land cover classes in Wales (2019). Both figures were obtained from the Welsh Soil Evidence Review (Welsh Government, 2022).

Table 14. Most common soil types in Wales, their characteristics and potential impacts these may have on the processing and transport of P

Soil type	Main landscape	Key characteristics	Impacts on the processing and transport of P
Podzols (covering ~32% of Wales)	Uplands (> 300 m) and steep areas	Acidic soils with iron-enriched subsoil.	▪ Podzols have a characteristic subsurface layer containing accumulated humus and metal oxides (predominantly iron and aluminium) (Britannica, 2010). The iron-enriched subsoil may contribute to P retention by forming chemical bonds between the manure OPs and iron (Pierzynski et al., 2005). ▪ In acid soils such as podzols, P binding to soils is largely controlled by iron and aluminium. ▪ The Wye catchment has podzols, especially in the uplands (Owens et al., 2008). ▪ In Scotland, P concentrations in streams draining well-drained coarse-textured podzols were over four-times lower than those draining the poorly drained fine-textured gley soils (Hooda et al., 1997).
Brown soils (covering ~30% of Wales)	Lowlands	Loamy and permeable with a weathered subsoil.	▪ High permeability promotes the decomposition of organic matter. ▪ High permeability also promotes the movement of nutrient to deeper soil layers and/or groundwater – particularly for N which is more mobile in soils than P. ▪ Brown earths which are silty or coarse loamy and well-drained are the dominat soils in the Wye catchment (Owens et al., 2008). ▪ Particular properties (high silt content) of the Wye soils mean they have a poor ability to retain applied P (in manures and fertilisers), and therefore pose a high risk of P loss to draining streams (Withers et al., 2022). It is important to note, however, that only a small area of land in Wales (and the Wye catchment) is shown to have soil types which exhibit or are likely to exhibit these properties of P behaviour (see Figure 16).
Surface-water gley soils (covering ~25% of Wales)	Lowlands, uplands and coastal areas	Gley soils have impeded drainage. They are clayey and seasonally waterlogged in the winter due to high groundwater tables of impermeable subsoil.	▪ There is a minimal risk of P entering groundwater in clay-rich soils because clay-rich soils have a high capacity to adsorb P. ▪ Clay-rich soils may promote slower rates of P desorption due to the increased P sorption strength through the formation of Ca-P complexes at the surface of alumino-silicate clays (Siddique and Robinson, 2004). ▪ However, highly impermeable soils may require underdrainage to combat waterlogged surfaces, which could impact water quality if manure is applied soon after the installation of mole and/or tile drains (Anderson and Wu, 2001). ▪ If underdrainage is not installed, high soil moisture content soils can lead to greater runoff (McConnell et al., 2016).

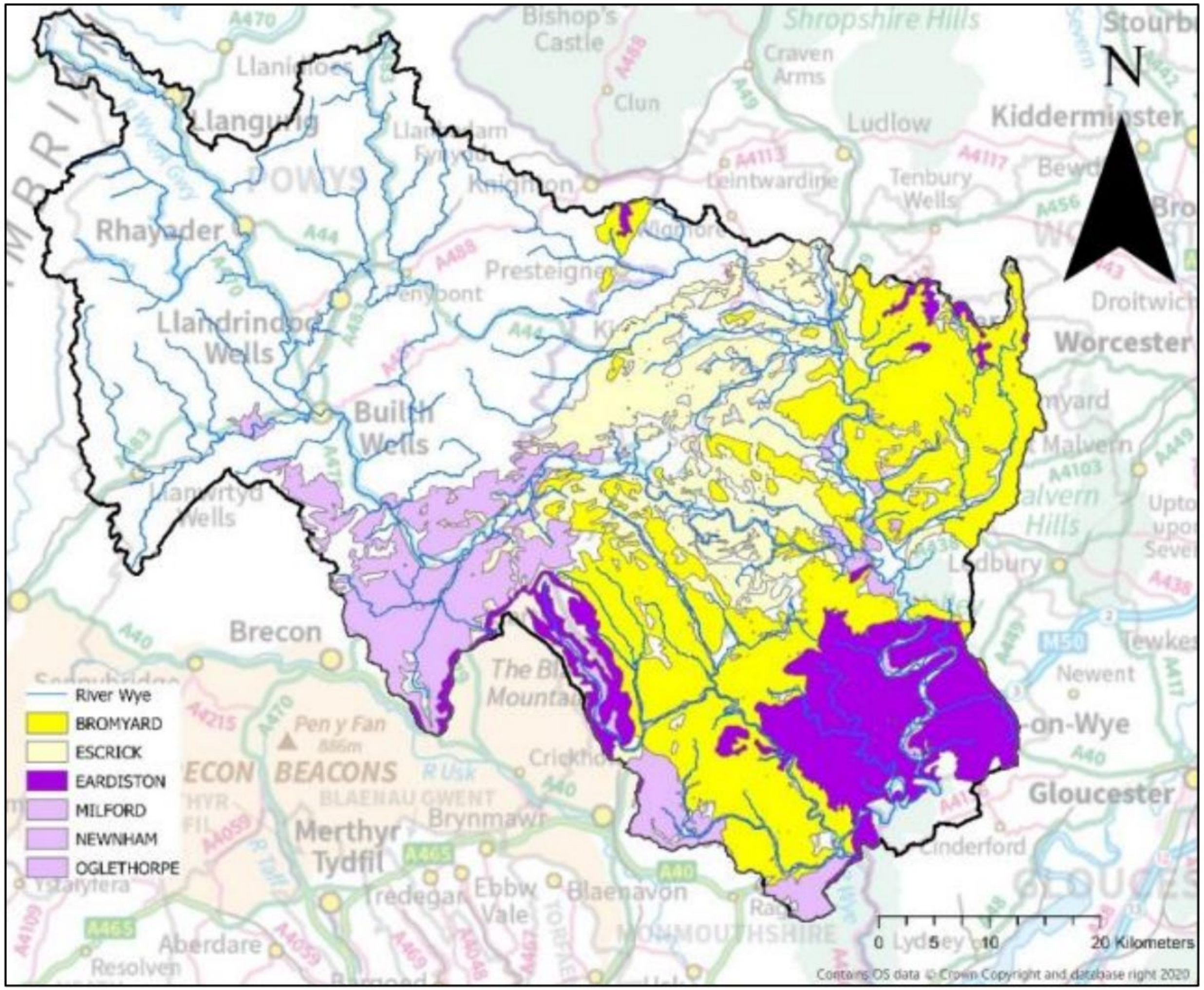


Figure 16. Distribution of the soils sampled from the Wye catchment (used in the trial). Bromyard (yellow) and Eardiston (purple) were the two soils used, the other soil types shown in the lighter shades are from the same soil series and will likely exhibit similar properties of P behaviour. (Withers et al., 2022)

How do different methods of storing and spreading manure affect the rate of P loss from soils? (RQ1c)

Manure storage

There is a concern about the potential for direct P loss to occur from manure during storage, and the effect of storage practices on P loss from manures following application to agricultural land.

The CoAPR, regulatory measures to address agricultural pollution, state that organic manure (other than slurry), including bedding contaminated with any organic manure, must be: stored in a vessel; in a covered building; on an impermeable surface; or in a temporary field site if it can be stacked in a free-standing heap without slumping. Slurry storage has several specific requirements, as detailed in CoAPR Part 6 (2021).

Field heaps

Organic manures stored in field heaps represent a significant threat to water quality if mobilised during rainfall events and washed into nearby water bodies (Doody et al., 2013). A number of storage best management practices have been adopted by farmers and/or incorporated into legislation to help mitigate nutrient losses from organic manure storage heaps.

The Nitrate Directive (91/676/EEC) regulations aim to minimise risks to waterbodies by preventing the mobilisation of contaminants during rainfall events and by providing a buffer zone to waterways. In 2013, regulations in the UK and Northern Ireland allowed for solid manure to be stored in temporary field heaps prior to land application. These regulations have since been replaced in Wales by the CoAPR (April 2021) which state that:

Temporary heaps must not be located within 10 m of a surface water body or land drain; within 30 m of a water course if land has an incline greater than 12° or within 50 m of a spring, borehole or well.

- Temporary heaps must not be located in a field prone to flooding or becoming water logged.
- Poultry manure (i.e., not containing bedding materials) must be covered with impermeable material.

Studies suggest that poultry litter stored in covered heaps pose a negligible risk to water quality if managed correctly, with the main factor controlling P export across sites indicated to be the pre-existing soil P concentration (Doody et al., 2012). Doody et al. (2012, 2013) found that field heaps established on bare soil reduce the risk to water quality compared to those stored on impermeable concrete due to the buffering capacity of the soil, and that the greater the distance a manure heap is stored relative to the water body the lower the risk posed to water quality[35].

Wet and dry storage

A long-term storage study (over 440 days) investigated the P speciation of broiler litters stored dry (at their internal moisture content (~24%)) and wet (at a moisture content of 40%[36]) (McGrath et al., 2005). The TP content and P speciation varied significantly with storage methods, with wet-storage resulting in a shift in P from organic forms to WEP such that the litter contained more than twice the amount of WEP when compared to that 'dry-stored' (McGrath et al., 2005). Similar results were reported by Kleinman et al. (2005) who found dry manures contained significantly lower WEP than manures from liquid storage systems.

Microbial activity within stored litter plays an important role in hydrolysing OP and thus increasing the labile P pool. Microbial activity mineralised significantly more P in litter stored wet than dry resulting in increased concentrations of more labile inorganic forms of P (McGrath et al., 2005). This resulted in significantly increased soil water-soluble P concentrations, and thus increased TDP and DRP in runoff from soil plots amended with wet-stored litter compared to dry-stored litter (McGrath et al., 2005).

Effectively managing stored poultry litter to minimise moisture content increases during storage may therefore significantly reduce dissolved P losses in runoff following the application of manure to agricultural land. For example, Chaump et al. (2019) found that litter stored outdoors in a waste pile had double the moisture content of litter collected from inside the poultry house, therefore covering litter/ manure during storage, or using fresh manure/ litter where possible may reduce nutrient losses following land application.

Manure spreading methods

The methods employed for spreading manures on land may affect the extent and rate of P loss from soils. In particular, the type of spreading (surface or sub-surface), application rate and timing of manure application.

Surface spreading verses sub-surface mixing

Several different methods exist for the application of organic manures to land, such as:

- Broadcasting (surface spreading).
- Broadcasting with incorporation (mixing, or incorporating, the manure into the soil (tillage) immediately or within a few days after broadcasting).
- Band spreading to soil surface (with trailing hoses/shoes).

[35] Study conducted under the Nitrates Action Programme Regulations (Northern Ireland) 2006, which states that manure must be stored in fields in compacted heaps, a minimum distance of 20m from any waterways.
[36] 40% was chosen as this is represents the highest moisture content of poultry litter typically found in broiler houses in the Delmarva peninsula, USA. poultry litter temperatures remained below 40 °C, therefore completed composting should not have occurred.

- Injection (sub-surface application) which can occur through knife injection, sweep injection, disk/coulter injection systems, and slurry precision application systems.

Studies have shown that sub-surface application of organic manures can reduce the loss of TP and dissolved P from agricultural land.

Kleinman et al. (2006) found that the surface application of manures consistently produced elevated concentrations of dissolved P in runoff whereas the incorporation of manures within the soil (e.g., by tillage) reduced runoff dissolved P concentrations. The incorporation of manures within soils removed manure P from the soil surface (the source of P in runoff) and promoted sorption of dissolved P in manure by soil (Kleinman et al., 2006). In an earlier study, Kleinman et al. (2002) reported that ~64% of TP in the runoff from the surface-applied plots was dissolved P, compared with ~9% when manures/ fertilisers were mixed into the soil.

Kibet et al. (2011) found significantly lower TP losses in runoff from soils with subsurface litter application (1.90 kg/ha) compared with surface poultry litter application (4.78 kg/ha) following a (simulated) rainfall event. However, by the second rainfall event TP losses did not differ significantly between surface and subsurface litter treatments. Subsurface application of poultry litter lowered the availability of litter P to runoff water over the short term, however in poorly drained soils subsurface pocket of manure/ litter could eventually act as a source of P due to rising water tables (Kibet et al,. 2011).

Elsewhere, Heinonem-Tanski and Uusi-Kämppä (2001) reported higher losses of TP (and TN) from surface spreading than injection application of cattle slurry.

Application rate

Studies have shown that application rate can influence the loss of P from agricultural land.

DeLaune et al. (2004) reported that P concentrations in runoff from land to which poultry litter was applied showed a positive correlation with application rate, whilst Smith et al. (2001) indicated that an application loading of around 2.5 – 3 t/ha cow slurry solids appears to be the threshold above which the risk of P loss in surface runoff losses greatly increases.

Evidence suggests that the increased potential for P desorption from soil and inclusion within runoff is related to the application rate of organic manure to agricultural land (Kleinman et al., 2006). Holford et al. (1997) found that the magnitude of the decrease in sorption capacity and strength was dependent on the amount of manure applied over time; with increased application rates causing a greater reduction in the sorption capacity and strength.

Timing of manure application

The amount of time between fertiliser application and the first rainfall/ runoff event has been shown to be an important factor in controlling P losses.

In the UK, a significant proportion of livestock manure is applied during autumn (August to October), with around 50% of pig and poultry manure and 25% of cattle manure added during this period (Chambers et al., 1999). However, autumn and winter applications may result in increased P losses within drainage waters, as soils may already be saturated. The application of manures to drier soil has been shown to help reduce P (and N) losses in runoff (Torbert et al., 1999). Manure should not be applied on snow or frozen ground and heavily manured fields should not be summer-fallowed to minimise the risk posed to groundwater N (Larson, 1991).

Whilst additions of manure to soil can have lasting effects on soil properties, the direct contribution of manure P to runoff tends to diminish with time (Kleinman et al., 2006). The potential for P loss peaks immediately after manure is applied to the sediment surface. As time progresses, dissolved P within the manure increasingly interacts with soil microbiota, becoming converted to recalcitrant forms (Edwards and Daniel, 1993).

However, in some cases dissolved P can remain elevated in runoff for a long duration following application. This appears to be particularly true for dry poultry litter application (Kleinman et al., 2006). For example, Pierson et al. (2001) observed concentrations of dissolved P in runoff > 1.0 mg/l (compared to a background concentration of 0.4 mg/l) for 19 months after poultry litter was surface applied to a pasture soil. Several studies have reported highly variable runoff P concentrations resulting from the timing of the last application of poultry litter (Sharpley, 1997; DeLaune et al., 2004; Romeis et al., 2011; Cox et al., 2013).

O'Rourke et al. (2010) calculated the length of the P signal in overland flow, giving an indication of the time during which elevated P concentrations above those in the control persist after manure application. DRP declined to the threshold (defined as 1 mg/l) after 9 days in both summer and winter, and 28 days in the spring, showing the differences in timing of application on runoff concentrations.

Processing and loading of P from broiler manure compared to free-ranging poultry manure/ litter (RQ2a)

Most of the research investigating the environmental impact of poultry production has focussed on the best practices to reduce the loss of P following the application of poultry manure/ litter to agricultural land that was generated in intensive broiler and layer farming systems (O'Bryan et al., 2017). Compared to conventional poultry systems, virtually no studies have investigated the impact of free-range poultry farms (or pastured poultry farms in the USA) on downstream water quality (Rothrock et al., 2019).

In the UK and many other EU countries free-ranging chicken farms have expanded since the 2012 EU ban on keeping egg-laying hens in cages. In the UK, legislation requires free range birds to meet certain requirements, such as the amount of space per bird, access to outdoor pastures and the age of the birds at slaughter. UK free range poultry production accounts for around 3.5% of total UK poultry meat production and is dominated by chickens although also includes turkeys, ducks and geese at Christmas (Griffiths, 2017).

Omeira et al. (2006) found that the chemical composition of litter varied between intensive and free-ranging systems. Chicken litter from intensive systems had a significantly higher TP and N content than litter from free-ranging broiler systems. Whilst the litter from free-range broiler systems displayed the lowest P values compared to other systems. These authors suggest this could be due to the difference in P metabolism within layers and broilers, and the accumulation in the litter of layers as they age (Omeira et al., 2006). Free-ranging birds were, however, found to compact the grass on which the birds forage, which increases the amount of runoff entering nearby watercourses (Rowe, 2017).

The impact of poultry manure/ litter digestate compared to raw poultry manure/ litter (RQ2b)

Poultry manure and poultry litter can be applied directly to agricultural land (land spreading) or may be treated by technologies including biotic based processes such as composting (aerobic microbial breakdown), AD, as well as thermal processes (e.g., pyrolysis to obtain biochar) prior to agricultural application (Figure 17) (Bhatnagar et al., 2022; Kacprzak et al., 2023). This section aims to address RQ2b and focuses on how digestate from AD plant using a poultry manure/ litter feedstock compares to raw (land spreading) manure in terms of P processing in the soil and loading to freshwater following application to agricultural land.

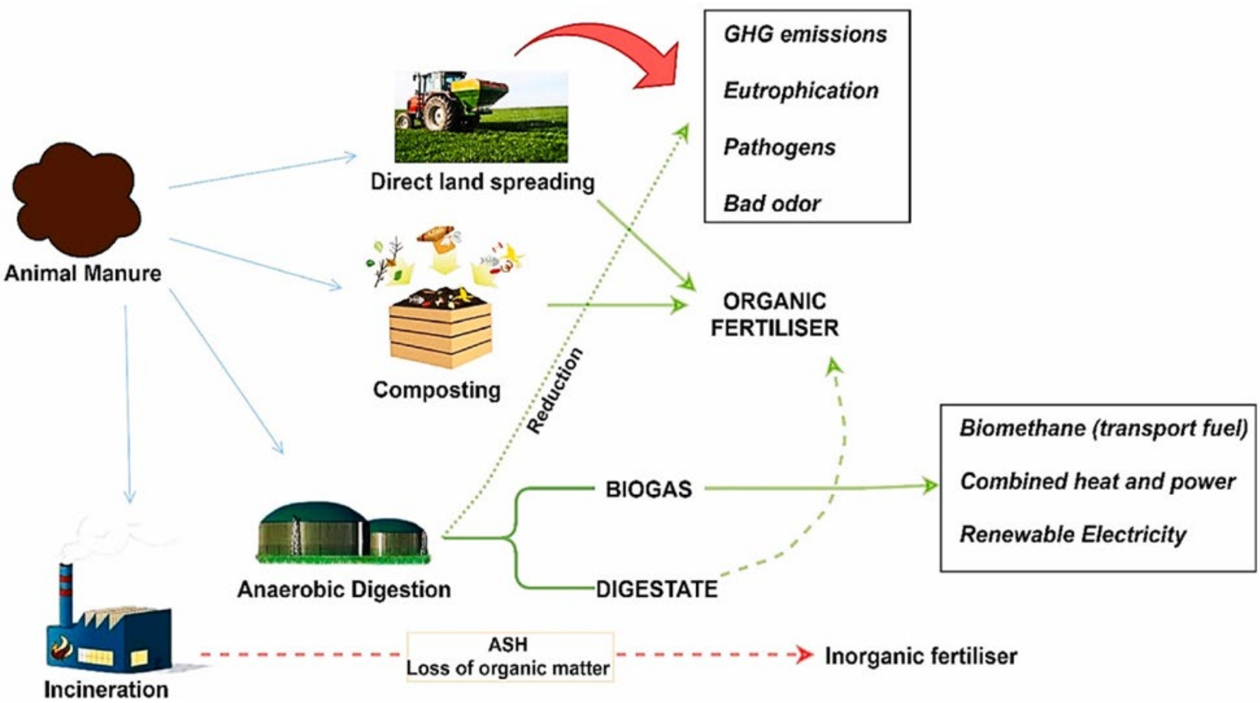


Figure 17. Animal manure disposal options (Bhatnagar et al., 2022)

AD is the breakdown of organic matter through biological processes in the absence of oxygen, producing biogas (methane, carbon dioxide and trace gases) and digestate (solid and liquid fractions) as by-products (Scarlet et al., 2018). Whilst AD has been extensively applied to food waste and dairy and pig manure, it is typically underutilised in the poultry industry owing to the high N, NH_4 and lignocellulose content of poultry litter (Chaump et al., 2019; Beausang et al., 2020; Bhatnagar et al., 2022). However, recent technological advances suggest that AD can be applied to effectively stabilise poultry litter, and recent studies have demonstrated the advantages of using poultry litter in AD plants, including odour and GHG mitigation, production of gaseous biofuel (renewable energy) and avoiding eutrophication of water bodies (Figure 17) (Hassanein et al., 2019; Bhatnagar et al., 2022).

AD biotechnologies produce solid and liquid fractions of digestate and biogas (at average rate of 0.48 l/g volatile solids from poultry slurries which is higher than swine and bovine slurries) (Massé et al., 2011a). During the AD treatment of poultry litter, the majority of the P is partitioned into the solid fraction whereas the majority of the N is present in the liquid fraction in the form of NH_4. Liedl et al. (2006) found that the concentration of P in the solid fraction (13 – 20 g/kg) was significantly greater than that in the liquid fraction (~0.33 g/kg) (Liedl et al., 2006), however, the solid fraction P content of the AD digestate does not differ significantly from the variable P content of raw poultry litter found in this study (11 – 19.4 g/kg; Table 3). The solid fraction of the digestate can be used as a peat replacement in horticulture, and the liquid fraction can be used to replace artificial nitrogen fertilisers (Beausang et al., 2020).

Chaump et al. (2019) compared digestate properties of both fresh and outdoor stored 'waste' poultry litter; though there was a higher concentration of soluble phosphate in waste litter (see Table 15) both types of litter experienced reductions (15 – 40%) in soluble phosphate concentration after digestion. Anaerobic processes are known to precipitate phosphate as salts of Mg, Ca, and Fe (Möller and Müller, 2012), leading to reductions in free, soluble phosphate in the digestate. Furthermore, digestate may be chemically treated to concentrate P by precipitation with advanced nanomaterials (Rashid et al., 2017). The concentrated phosphate can then be transported cost-effectively to distant locations and the remaining liquid can be used with reduced concern regarding P accumulation in local soils (Chaump et al., 2019).

Chaump et al., (2019) also found that NO_2 and NO_3 concentrations declined after digestion of both fresh and outdoor stored poultry litter, but declines were greater during stored litter digestion than fresh litter digestion (see Table 15). In contrast, the soluble TN content increased following digestion, with similar results observed from both litter feedstocks (Table 15).

Improvements in AD efficiency can be achieved by feedstock pre-treatment (before introducing feedstock to the digestion chambers) and/or co-digestion of poultry manure with other organic waste (Kacprzak et al., 2023). The Tully biogas plant (Northern Ireland) is one of the first biogas plants in the world to operate on poultry litter mono-digestion (AD with single substrate) combined with a patented nitrogen stripping technology (Bhatnagar et al., 2022).

The nutrients contained in AD digestate are more accessible to plants and have N:P ratios that are more balanced to meet the crop needs than the nutrients contained in other organic fertilisers, thus reducing the need for supplementary chemical N fertilisers (Massé et al., 2011b). Using digestate over raw manure/ litter therefore offers potential benefits for water quality and environmental and human health as it decreases organic pollution potential as well as reducing risk of spreading microbial contamination (IEA, 2019). However, if the nutrients enter water bodies through incidental losses (prior to equilibration with soils and/or uptake by plants), then this may represent a greater risk to water quality, as the nutrients will also be highly available to algae. In this regard, the application of digestate requires careful planning concerning the timing and amount of digestate applied to land.

Potentially one of the most significant advantages of AD is the production of a concentrated P source that is economically viable to transport significant distances. This was noted by Prof. Phil Haygarth (Appendix D.1) as one of the key innovation challenges required in order to fully utilise the nutrients within manure and create a sustainable circular economy. Being able to transport nutrients from areas of intensive poultry farming operations to areas with nutrient deficit will reduce the tendency to over-fertilise land proximal to poultry farming operations, whilst also reducing the need to import inorganic fertiliser to areas further afield, effectively reducing the amount of P entering the agricultural system. This was echoed by Rothwell et al. (2022) who conclude that policies that target the recovery of P from secondary sources (manure, biosolids, food waste) are critical to addressing the inefficient use of P in the agricultural system. However, only by effectively replacing imported inorganic P fertiliser, will the national P surplus decline and efficiency improve.

Table 15. P and N content in poultry litter before and after AD (Chaump et al. 2019)[37]

Treatment	**Phosphate (mg/l)**	**Soluble N (mg/l)**	**NO_2 (mg/l)**	**NO_3 (mg/l)**	**NH_4 (mg/l)**
Fresh poultry litter (*before digestion*)	371	935	0	11	419
Fresh poultry litter leachate (*after digestion*)	266	1185	0	10	895
Waste poultry litter (*before digestion*)	614	600	10	12	414
Waste poultry litter leachate (*after digestion*)	385	863	0	9	671

[37] Values in the table are mean of results reported in Chaump et al. (2019) for the four different solids loading rates investigated.

5. Summary and conclusions

The application of organic manures, including poultry litter/ manure, has been recognised as a source, or potential source, of P in downstream freshwater environments. Understanding the key factors that determine the extent to which organic manures increase the concentration of bioavailable P downstream is nevertheless a complex question. For example, the risk posed by manures to water quality is impacted by manure management (storage and spreading methods), catchment characteristics (soil type, topography, proximity to watercourses) and climate characteristics (rainfall). These in turn influence the mobilisation, delivery and overall impact of manure P on water quality.

Whilst organic fertilisers vary in physical (e.g., dry matter) and chemical composition (e.g., nutrients), they have the potential to provide a valuable source of P fertiliser. If managed efficiently and effectively this resource provides an opportunity to reduce reliance on importation of inorganic P into the agricultural system. This in turn would contribute towards a more sustainable circular economy.

However, this is not obtainable without overcoming a number of barriers and challenges. For example, one of the most significant problems with manure usage in agriculture is that it is not economically viable to transport significant distances from source. Being able to redistribute this P resource to areas further away from livestock operations would represent a significant advancement towards sustainable usage of manure as fertiliser.

Conclusions relating to each research question are presented below.

RQ1a: *How readily do the components of P from different manure types become converted to the bioavailable form on land and in water?*

- TP is generally lower in the manure of ruminant animals (cows, sheep and goats) and higher in non-ruminant animals (poultry and swine).
- The form of P added to land is an important factor controlling mobilisation and bioavailability. Application of manure containing soluble P can directly contribute to soluble P in runoff, such that differences in P solubility between manure types can influence the amount of soluble P lost in runoff once manure is added.
- Poultry litter/ manure and pig manure/ slurry likely represent a greater potential risk to surface water from leaching than dairy manure and biosolids owing to the higher soluble P/ WEP content.
- Soil type is determined by physical properties (such as texture, permeability and moisture content) and chemical properties (such as pH) which can influence the transport and ultimately loss of manure P to surface waters.
- Indirectly, soil type can dictate the agricultural practices that are undertaken on the land and this in turn may influence P loss.
- A limited number of studies were found which assessed the direct effects of poultry manure on water quality in the same climatic zone as the UK.

RQ1b: *If soil type is an important variable, how do the soil types of Wales influence the processing and transport of manure P?*

- Welsh soils are predominantly podzols (32%), brown soils (30%) and gley soils (25%), and have differing characteristics that may impact the transport and processing of manure P:
    - Podzols typically have a subsurface layer containing Fe and Al oxides which can contribute to the retention of P.
    - Brown soils are highly permeable, which may promote the movement of nutrients to deeper soil layers and/or groundwater – particularly for N which is more mobile in soils than P.
    - Gley soils (associated with improved grassland) have a high capacity to adsorb P, which lowers the risk of P entering groundwater. However, they are also highly impermeable, and may require underdrainage to combat waterlogged surfaces, which could impact water quality if manure is applied soon after the installation of mole and/or tiles drains.
- As such, Wales is predominantly covered by soils (podzols and gley soils) which are more likely to retain nutrients. Nevetheless, risk of P loss will vary on a catchment basis owing to the influence of additional characteristics, such as elevation change and proximity to watercourses.

RQ1c: *How do different methods of storing and spreading manure affect the rate of P loss from soils?*

- TP content and P speciation vary significantly with dry verses wet storage of poultry litter. It has been shown that:
    - Dry storage of litter can result in minor changes in P forms.
    - Wet storage can result in a shift in P from organic forms to inorganic WEP, subsequently increasing soluble concentrations of P in runoff from land to which manure applied.
- Studies suggest that poultry litter stored in covered heaps pose a negligible risk to water quality if managed correctly.
- Studies have shown that sub-surface application of organic manures can reduce the loss of TP and dissolved P from agricultural land.
- Studies have shown that application rate can influence the loss of P from agricultural land; possibly influencing P desorption from soil and inclusion within runoff.
- The amount of time between fertiliser application and the first rainfall/ runoff event has been shown to be an important factor in controlling P losses.

RQ2a: *How does poultry litter/ manure that is collected from broiler units and then spread on land compare to manure added to land via free-ranging in terms of P processing in soil and loading to freshwaters?*

- The chemical composition of litter is likely to vary between intensive and free-ranging systems. The only study from the academic literature reviewed which compared these systems found that intensive systems had higher TP (and N) content; which could result in a higher risk to downstream water quality.
- Compared to conventional poultry systems, virtually no studies have investigated the impact of free-range poultry farms (or pastured poultry farms in the USA) on downstream water quality.

RQ2b: *How does digestate from anaerobic digestion (AD) plants using a poultry litter/manure feedstock compare to raw poultry manure in terms of P processing in soil and loading to freshwaters once it is spread on land?*

- Nutrients contained in digestate are more accessible to plants than the nutrients contained in other organic fertilisers. However, if N and P are not taken up quickly by the plants, then nutrients may be more likely to enter water bodies via leachate and/or runoff.
- Using anaerobic digestate offers environmental and human health related benefits by reducing the risk of spreading microbial contamination.
- It is more economically feasible to transport manure digestate (following AD) further from the source, which reduces the likelihood of over fertilisation on agricultural land proximal to poultry farming operations.
- Being able to move manure/ litter further from the source effectively and economically may represent one of the most significant innovation challenges in the field. However, this may also offer the greatest potential benefits to water quality by reducing the amount of inorganic phosphate fertiliser entering the agricultural system and reducing over-application of manure to fields proximal to poultry operations.

It is important to acknowledge the limitations of this study. As a rapid evidence assessment, the breadth and scope of the literature searches used for this report are more limited, and the critical appraisal of the evidence less comprehensive, than a full systematic review. Furthermore, a comprehensive assessment of factors influencing bioavailable P other than manure type, soil type and manure storage and spreading methods, as well as mitigation measures to reduce P loading to surface waters was beyond the scope of this review.

6. Recommendations

- Few recent studies were found which assessed the direct effects of poultry manure on water quality in the same climatic zone as the UK and therefore further research is needed in this area.
- Only one study from the academic literature reviewed compared poultry litter from intensive and free-ranging systems, reporting lower levels of P in the latter. The potential impacts of free-range poultry farms on downstream water quality are however unknown and therefore further research is needed in this area.
- Using digestate (from AD) over raw manure/ litter offers the potential benefit of being more economically viable to transport. The potential for AD to be utilised more readily warrants further investigation. For example, information could be sought from Tully biogas plant (Northern Ireland)[38] to better understand their operation.
- Findings from this study have shown that different methods of spreading and storing manure can affect the rate of P loss when applied to land. Therefore, the importance of spreading technique, application rate and timing, and storage conditions (in particular wet versus dry storage) should be considered in the future management of manure application.
- The discrepancy between values calculated in this study from Welsh Government livestock data and those reported in the RePhoKUs report (Withers et al., 2022) as to the contribution of poultry to overall manure P production should be investigated.
- Future work should investigate potential future trends in agriculture in Wales and assess the potential impact this may have on manure production and the frequency and quantity of manure/ litter spread on land. Specifically for poultry farming, the market for eggs and chickens, and novel alternatives to current bedding materials should be explored.
- Whilst not the focus of this review, it has been reported that composting of poultry manure/ pig slurry can reduce the amount of WEP/ WSP that may be subsequently applied to land. It is therefore plausible that the application of composted rather than fresh manure/ slurry may reduce the amount of P entering waterbodies through incidental losses, though this requires further investigation.

[38] One of the first biogas plants in the world to operate on poultry litter mono-digestion (AD with single substrate) combined with a patented nitrogen stripping technology (Bhatnagar et al., 2022).

References

Ahmed, W., Jing, H., Kaillou, L., Qaswar, M., Khan, M. N., Jin, C., Geng, S., Qinghai, H., Yiren, L., & Guangrong, L. 2019. Changes in phosphorus fractions associated with soil chemical properties under long-term organic and inorganic fertilization in paddy soils of southern China. PloS One, 14(5), e0216881.

Akhtar, M., McCallister, D. L., & Eskridge, K. M. 2002. Availability and fractionation of phosphorus in sewage sludge-amended soils. Communications in Soil Science and Plant Analysis, 33(13–14), 2057–2068.

Anderson, G., Williams, E. G., & Moir, J. O. 1974. A Comparison of the sorption of inorganic orthophosphate and inositol hexaphosphate by six acid soils. Journal of Soil Science, 25(1), 51–62.

Anderson, R., & Wu, Y. 2001. Phosphorus quantity–intensity relationships and agronomic measures of P in surface layers of soil from a long-term slurry experiment. Chemosphere, 42(2), 161–170.

Anderson, R., & Xia, L. 2001. Agronomic measures of P, Q/I parameters and lysimeter-collectable P in subsurface soil horizons of a long-term slurry experiment. Chemosphere, 42(2), 171–178.

ASAE, American Society of Agricultural Engineers. 2005. ASABE Standards Manure Production and Characteristics. ASAE D384.2 MAR2005 (R2014) Available from: https://elibrary.asabe.org/abstract.asp?aid=32018&t=&redir=&redirType=.

Asmare Atalay Carol Bronick, T. M., & Whitehead, B. 2011. Phosphorus Immobilization and Soil Aggregation in Chemically Amended Poultry Litter Used in Corn/Soybean Rotation. Journal of Sustainable Agriculture, 35(3), 260–275.

Assured Biosolids Limited, 2024. About biosolids. Available from: https://assuredbiosolids.co.uk/about-biosolids/ (accessed 26/01/2024)

Bateman, A., van der Horst, D., Boardman, D., Kansal, A., & Carliell-Marquet, C. 2011. Closing the phosphorus loop in England: The spatio-temporal balance of phosphorus capture from manure versus crop demand for fertiliser. Resources, Conservation and Recycling, 55(12), 1146–1153.

Beausang, C., McDonnell, K., & Murphy, F. 2020. Anaerobic digestion of poultry litter – A consequential life cycle assessment. Science of The Total Environment, 735, 139494.

Bernhart, M., & Fasina, O. O. 2009. Moisture effect on the storage, handling and flow properties of poultry litter. Waste Management, 29(4), 1392–1398.

Bhatnagar, N., Ryan, D., Murphy, R., & Enright, A. M. 2022. A comprehensive review of green policy, anaerobic digestion of animal manure and chicken litter feedstock potential – Global and Irish perspective. Renewable and Sustainable Energy Reviews, 154, 111884.

Biosolids, A. 2023. About Biosolids. https://assuredbiosolids.co.uk/about-biosolids/

Bloem, E., Albihn, A., Elving, J., Hermann, L., Lehmann, L., Sarvi, M., Schaaf, T., Schick, J., Turtola, E., & Ylivainio, K. 2017. Contamination of organic nutrient sources with potentially toxic elements, antibiotics and pathogen microorganisms in relation to P fertilizer potential and treatment options for the production of sustainable fertilizers: A review. Science of The Total Environment, 607–608, 225–242.

Blume, H. P., Brümmer, G. W., Horn, R., Kandeler, E., Kögel-Knabner, I., Kretzschmar, R., Stahr, K., & Wilke, B. M. 2010. Textbook of Soil Scinece (Lehrbuch der Bodenkunde) (16th ed.). Spektrum Akademischer Verlag GmbH.

Börling, K., Otabbong, E., & Barberis, E. 2001. Phosphorus sorption in relation to soil properties in some cultivated Swedish soils. Nutrient Cycling in Agroecosystems, 59, 39-46.

Bos, J., Williams, M., Smith, D., Armstrong, S., & Harmel, D. 2021. Long-term effect of poultry litter application on phosphorus balances and runoff losses (Vol. 50, No. 3, pp. 639-652).

Brady, N. C., Weil, R. R., & Weil, R. R. 2008. The nature and properties of soils (Vol. 13). Prentice Hall Upper Saddle River, NJ.

Brandt, R. C., Elliott, H. A., & O'Connor, G. A. 2004. Water-Extractable Phosphorus in Biosolids: Implications for Land-Based Recycling. Water Environment Research, 76(2), 121–129.

Breeuwsma, A., Reijerink, J. G. A., & Schoumans, O. F. 1995. Impact of manure on accumulation and leaching of phosphate in areas of intensive livestock farming. In Animal waste and the land-water interface (pp. 239–249). Lewis.

Brennan, R. B., Fenton, O., Grant, J., & Healy, M. G. 2011. Impact of chemical amendment of dairy cattle slurry on phosphorus, suspended sediment and metal loss to runoff from a grassland soil. Science of The Total Environment, 409(23), 5111–5118.

Britannica, The Editors of Encyclopaedia. "Podzol". Encyclopedia Britannica, 16 Dec. 2010, https://www.britannica.com/science/Podzol. Accessed 8 January 2024.

Burns, J. C., Westerman, P. W., King, L. D., Overcash, M. R., & Cummings, G. A. 1987. Swine Manure and Lagoon Effluent Applied to a Temperate Forage Mixture: I.

Persistence, Yield, Quality, and Elemental Removal. Journal of Environmental Quality, 16(2), 99–105.

Casson, J. P., Bennett, D. R., Nolan, S. C., Olson, B. M., & Ontkean, G. R. 2006. Degree of phosphorus saturation thresholds in manure-amended soils of Alberta. Journal of Environmental Quality, 35(6), 2212–2221.

CEE. 2013. Guidelines for systematic review and evidence synthesis in environmental Management. Environmental Evidence, 4.2. https://www.environmentalevidence.org/wp-content/uploads/2014/06/Review-guidelines-version-4.2-final.pdf

Chakraborty, D., Prasad, R., Bhatta, A., & Torbert, H. A. 2021. Understanding the environmental impact of phosphorus in acidic soils receiving repeated poultry litter applications. Science of The Total Environment, 779, 146267.

Chambers, B., Royle, S., Hadden, S., & Maslen, S. 2002. The use of biosolids and other organic substances in the creation of soil-forming materials. Journal of the Chartered Institution of Water and Environmental Management, 16(1), 34–39.

Chang, A. C., Page, A. L., Sutherland, F. H., & Grgurevic, E. 1983. Fractionation of Phosphorus in Sludge-Affected Soils. Journal of Environmental Quality, 12(2), 286–290.

Chardon, W. J., Oenema, O., del Castilho, P., Vriesema, R., Japenga, J., & Blaauw, D. 1997. Organic Phosphorus in Solutions and Leachates from Soils Treated with Animal Slurries. Journal of Environmental Quality, 26(2), 372–378.

Chaump, K., Preisser, M., Shanmugam, S. R., Prasad, R., Adhikari, S., & Higgins, B. T. 2019. Leaching and anaerobic digestion of poultry litter for biogas production and nutrient transformation. Waste Management, 84, 413–422.

Chichester, F. W., Van Keuren, R. W., & Mc Guinness, J. L. 1979. Hydrology and chemical quality of flow from small pastured watersheds: II. Chemical quality (Vol. 8, No. 2, pp. 167-171). American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America.

Choudhary, M., Bailey, L., & Grant, C. 1996. Review of the Use of Swine Manure in Crop Production: Effects On Yield and Composition and On Soil and Water Quality. Waste Management & Research - Waste Manage Res, 14, 581–595.

Chrysostome, M., Nair, V. D., Harris, W. G., & Rhue, R. D. 2007. Laboratory validation of soil phosphorus storage capacity predictions for use in risk assessment. Soil Science Society of America Journal, 71(5), 1564–1569.

Climate Change & Infectious Diseases Group. 2023. World maps of Köppen-Geiger climate classification. Available from: https://koeppen-geiger.vu-wien.ac.at/ (accessed 30/01/2024)

CoAPR. 2021. The Water Resources (Control of Agricultural Pollution) (Wales) Regulations 2021. Available at: https://www.legislation.gov.uk/wsi/2021/77/contents/made (accessed 31/01/2024).

Collaboration for Environmental Evicnece. 2013. Collaboration for Environmental Evidence. Guidelines for systematic review and evidence synthesis in environmental Management. Version 4.2. Environmental Evidnece. https://www.environmentalevidence.org/wp-content/uploads/2014/06/Review-guidelines-version-4.2-final.pdf

Collins, A., Coughlin, D., Miller, J., & Kirk, S. 2015. The Production of Quick Scoping Reviews and Rapid Evidence Assessments a How to Guide Joint Water Evidence Group.

Cooke, G. W., & Williams, R. J. B. 1973. Significance of man-made sources of phosphorus: Fertilizers and farming. The phosphorus involved in agricultural systems and possibilities of its movement into natural water. Water Research, 7(1), 19–33.

Cox, T. J., Engel, B. A., Olsen, R. L., Fisher, J. B., Santini, A. D., & Bennett, B. J. 2013. Relationships between stream phosphorus concentrations and drainage basin characteristics in a watershed with poultry farming. Nutrient Cycling in Agroecosystems, 95(3), 353–364.

Crandall, P., Gibson, K. E., Jaroni, D., O'Bryan, C. A., & Ricke, S. C. 2017. Assessment of nitrogen and phosphorus loads present in environments impacted by alternative poultry processing operations utilized in pasture-raised poultry production. Renewable Agriculture and Food Systems, 32(1), 33–42.

Curtis, M. J., Rider, D. E., Fristensky, A., & Claassen, V. P. 2009. Cumulative Sediment, Nitrogen and Phosphorus Losses from Bare and Compost-Amended Fill Slopes. Compost Science and Utilization, 17(1), 25–30.

D'odorico, P., Laio, F., Porporato, A., & Rodriguez-Iturbe, I. 2003. Hydrologic controls on soil carbon and nitrogen cycles. II. A case study. Advances in Water Resources, 26(1), 59–70.

Dail, H. W., Zhongqi, H., Erich, M. S., & Honeycutt, C. W. 2007. Effect of Drying on Phosphorus Distribution in Poultry Manure. Communications in Soil Science and Plant Analysis, 38(13–14), 1879–1895.

Daverede, I. C., Kravchenko, A. N., Hoeft, R. G., Nafziger, E. D., Bullock, D. G., Warren, J. J., & Gonzini, L. C. 2004. Phosphorus runoff from incorporated and surface-applied liquid swine manure and phosphorus fertilizer. Journal of Environmental Quality, 33(4), 1535–1544.

DeLaune, P. B., Moore Jr., P. A., Carman, D. K., Sharpley, A. N., Haggard, B. E., & Daniel, T. C. 2004. Development of a Phosphorus Index for Pastures Fertilized with Poultry Litter—Factors Affecting Phosphorus Runoff. Journal of Environmental Quality, 33(6), 2183–2191.

DeLaune, P., Moore, P., & Formica, S. 2010. Reducing Phosphorus in Swine Effluent with Aluminum Chloride Treatment during Lagoon Cleanout. Applied Engineering in Agriculture, 26.

De Vries, M., & de Boer, I. J. M. 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science, 128(1–3), 1–11.

Domburg, P., Edwards, A. C., Sinclair, A. H., Wright, G. G., & Ferrier, R. C. 1998. Changes in fertilizer and manurial practices during 1960–1990: implications for N and P inputs to the Ythan catchment, N.E. Scotland. Nutrient Cycling in Agroecosystems, 52(1), 19–29.

Doody, D. G., Bailey, J. S., & Watson, C. J. 2013. Evaluating the evidence-base for the Nitrate Directive regulations controlling the storage of manure in field heaps. Environmental Science & Policy, 29, 137–146.

Doody, D. G., Foy, R. H., Bailey, J. S., & Matthews, D. 2012. Minimising nutrient transfers from poultry litter field heaps. Nutrient Cycling in Agroecosystems, 92(1), 79–90.

Dróżdż, D., Wystalska, K., Malińska, K., Grosser, A., Grobelak, A., & Kacprzak, M. 2020. Management of poultry manure in Poland – Current state and future perspectives. Journal of Environmental Management, 264, 110327.

Dŵr Cymru Welsh Water. 2023. Phosphorus Source Apportionment Summary: Updating the Sagis Upper Wye Model. Available from: https://corporate.dwrcymru.com/en/community/environment/river-water-quality/sac-rivers (accessed 26/02/2024)

EA. 2023. State of the water environment indicator B3: supporting evidence. https://www.gov.uk/government/publications/state-of-the-water-environment-indicator-b3-supporting-evidence/state-of-the-water-environment-indicator-b3-supporting-evidence

Edwards, R. D., & Daniel, C. T. 1993. Drying Interval Effects on Runoff From Fescue Plots Receiving Swine Manure. Transactions of the ASAE, 36(6), 1673–1678.

Eldridge, S. M., Chan, K. Y., Barchia, I., Pengelly, P. K., Katupitiya, S., & Davis, J. M. 2009. A comparison of surface applied granulated biosolids and poultry litter in terms of risk to runoff water quality on turf farms in Western Sydney, Australia. Agriculture, Ecosystems & Environment, 134(3), 243–250.

Elliott, H. A., Brandt, R. C., & O'Connor, G. A. 2005. Runoff Phosphorus Losses from Surface-Applied Biosolids. Journal of Environmental Quality, 34(5), 1632–1639.

Elliott, H., Brandt, R., Kleinman, P., Sharpley, A., & Beegle, D. 2006. Estimating Source Coefficients for Phosphorus Site Indices. Journal of Environmental Quality, 35, 2195–2201.

Epelde, L., Jauregi, L., Urra, J., Ibarretxe, L., Romo, J., Goikoetxea, I., & Garbisu, C. 2018. Characterization of Composted Organic Amendments for Agricultural Use. Frontiers in Sustainable Food Systems, 2.

European Union. 2023. Glossary: Manure. https://ec.europa.eu/eurostat/statistics-explained/index.php?oldid=476892

Farming and Water Scotland. 2020. How much manure and slurry is produced over the housed period? https://www.farmingandwaterscotland.org/downloads/4pp-section-4-better-nutrient-use-working-it-out/

Galbally, P., Ryan, D., Fagan, C. C., Finnan, J., Grant, J., & McDonnell, K. 2013. Biosolid and distillery effluent amendments to Irish short rotation coppiced willow plantations: Impacts on groundwater quality and soil. Agricultural Water Management, 116, 193-203.

Gangbazo, G., Pesant, A. R., Barnett, G. M., Charuest, J. P., & Cluis, D. 1995. Water Contamination by Ammonium Nitrogen Following the Spreading of Hog Manure and Mineral Fertilizers. Journal of Environmental Quality, 24(3), 420–425.

García-Albacete, M., Martín, A., & Cartagena, M. C. 2012. Fractionation of phosphorus biowastes: Characterisation and environmental risk. Waste Management, 32(6), 1061–1068.

Geohring, L. D., McHugh, O. V., Walter, M. T., Steenhuis, T. S., Akhtar, M. S., & Walter, M. F. 2001. Phosphorus transport into subsurface drains by macropores after manure applications: Implications for best manure management practices. Soil science, 166(12), 896-909.

Gilley, J. E., Bartelt-Hunt, S., Lamb, S. J., Li, X., Marx, D., Snow, D., Parker, D., & Woodbury, B. 2013. Runoff Nutrient Transport As Affected By Land Application Method, Swine Growth Stage, And Runoff Rate. Transactions of the ASABE (American Society of Agricultural and Biological Engineers), 56, 1295–1303.

Griffiths, R. 2017. Introduction to marketing standards for free-range and organic poultry meat. Available from: https://britishpoultry.org.uk/introduction-to-marketing-standards-for-free-range-and-organic-poultry-meat/#:~:text=Terms%20such%20as%20extensive%20indoor%20%28barn%29%2C%20free%20range%2C,runs%20for%20a%20fixed%20period%20during%20their%20lives. (accessed 26/01/2024)

Guo, M., Tongtavee, N., & Labreveux, M. 2009. Nutrient dynamics of field-weathered Delmarva poultry litter: implications for land application. Biology and Fertility of Soils, 45(8), 829–838.

Guppy, C. N., Menzies, N. W., Blamey, F. P. C., & Moody, P. W. 2005. Do decomposing organic matter residues reduce phosphorus sorption in highly weathered soils?. Soil Science Society of America Journal, 69(5), 1405-1411.

Gupta, G., & Gardner, W. 2005. Use of clay mineral (montmorillonite) for reducing poultry litter leachate toxicity (EC50). Journal of Hazardous Materials, 118(1), 81–83.

Gustafsson, J. P., Mwamila, L. B., & Kergoat, K. 2012. The pH dependence of phosphate sorption and desorption in Swedish agricultural soils. Geoderma, 189–190, 304–311.

Haggard, B., DeLaune, P., Smith, D., & Moore, P. 2005. Nutrient and β17-estradiol loss in runoff water from poultry litters. JAWRA Journal of the American Water Resources Association, 41, 245–256.

Hassanein, A., Lansing, S., & Tikekar, R. 2019. Impact of metal nanoparticles on biogas production from poultry litter. Bioresource Technology, 275, 200–206.

Haygarth, P. M., Condron, L. M., Heathwaite, A. L., Turner, B. L., & Harris, G. P. 2005. The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. Science of The Total Environment, 344(1), 5–14.

He, Z., Zhang, H., Toor, G. S., Dou, Z., Honeycutt, C. W., Haggard, B. E., & Reiter, M. S. 2010. Phosphorus distribution in sequentially extracted fractions of biosolids, poultry litter, and granulated products. Soil Science, 175(4), 154-161.

Heathwaite, A. L., Burke, S. P., & Bolton, L. 2006. Field drains as a route of rapid nutrient export from agricultural land receiving biosolids. Science of the Total Environment, 365(1–3), 33–46.

Heathwaite, A. L., & Dils, R. M. 2000. Characterising phosphorus loss in surface and subsurface hydrological pathways. Science of The Total Environment, 251–252, 523–538.

Heathwaite, A. L., Quinn, P. F., & Hewett, C. J. M. 2005. Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. Journal of Hydrology, 304(1–4), 446–461.

Heinonen-Tanski, H., & Uusi-Kämppä, J. 2001. Runoff of faecal microorganisms and nutrients from perennial grass ley after application of slurry and mineral fertiliser.

Water Science and Technology : A Journal of the International Association on Water Pollution Research, 43, 143–146.

Hill, J. E., & Cade-Menun, B. J. 2009. Phosphorus-31 Nuclear Magnetic Resonance Spectroscopy Transect Study of Poultry Operations on the Delmarva Peninsula. Journal of Environmental Quality, 38(1), 130–138.

Hinedi, Z. R., Chang, A. C., & Lee, R. W. K. 1989. Characterization of Phosphorus in Sludge Extracts Using Phosphorus-31 Nuclear Magnetic Resonance Spectroscopy. Journal of Environmental Quality, 18(3), 323–329.

Hodgkinson, R. A., Chambers, B. J., Withers, P. J. A., & Cross, R. 2002. Phosphorus losses to surface waters following organic manure applications to a drained clay soil. Agricultural Water Management, 57(2), 155–173.

Holford, I. C. R., Hird, C. C., & Lawrie, R. A. 1997. Effects of animal effluents on the phosphorus sorption characteristics of soils. Soil Research, 35, 365–374.

Hooda, P. S., Moynagh, M., Svoboda, I. F., Thurlow, M., Stewart, M., Thomson, M., & Anderson, H. A. 1997. Soil and land use effects on phosphorus in six streams draining small agricultural catchments in Scotland. Soil Use and Management, 13(4), 196–204.

Hooda, P. S., Rendell, A. R., Edwards, A. C., Withers, P. J. A., Aitken, M. N., & Truesdale, V. W. 2000. Relating soil phosphorus indices to potential phosphorus release to water. Wiley Online Library.

Hoover, N. L., Kanwar, R., Soupir, M. L., & Pederson, C. 2015. Effects of Poultry Manure Application on Phosphorus in Soil and Tile Drain Water Under a Corn-Soybean Rotation. Water, Air, & Soil Pollution, 226(5), 138.

Hoover, N. L., Law, J. Y., Long, L. A. M., Kanwar, R. S., & Soupir, M. L. 2019. Long-term impact of poultry manure on crop yield, soil and water quality, and crop revenue. Journal of Environmental Management, 252, 109582.

Hountin, J. A., Couillard, D., & Karam, A. 1997. Soil carbon, nitrogen and phosphorus contents in maize plots after 14 years of pig slurry applications. The Journal of Agricultural Science, 129(2), 187–191.

Hubbard, R. K., Newton, G. L., & Hill, G. M. 2004. Water Quality and the Grazing Animal. Publications from USDA-ARS/ UNL Factulty.

IEA. 2019. Mono-Digestion of Chicken Litter: Tully Biogas Plant, Ballymena, Northern Ireland. Available from: https://www.ieabioenergy.com/blog/publications/mono-digestion-of-chicken-litter-tully-biogas-plant-ballymena-northern-ireland/

Jalali, M., Jalali, M., & McDowell, R. W. 2022. Minimizing phosphorus leaching from a sandy clay loam caused by phosphorus fertilizers. Environmental Monitoring and Assessment, 194(8), 587.

James, K. L., Randall, N. P., & Haddaway, N. R. 2016. A methodology for systematic mapping in environmental sciences. Environmental Evidence, 5(1), 7.

Jędrczak, A., Królik, D., Sądecka, Z., Myszograj, S., Suchowska-Kisielewicz, M., & Bojarski, J. 2014. Testing of Co-Fermentation of Poultry Manure and Corn Silage. Civil and Environmental Engineering Reports, 13(2), 31–47.

Kacprzak, M., Malińska, K., Grosser, A., Sobik-Szołtysek, J., Wystalska, K., Dróżdż, D., Jasińska, A., & Meers, E. 2023. Cycles of carbon, nitrogen and phosphorus in poultry manure management technologies – environmental aspects. Critical Reviews in Environmental Science and Technology, 53(8), 914–938.

Kelley, A., Wilkie, A. C., & Maltais-Landry, G. 2020. Food-based composts provide more soil fertility benefits than cow manure-based composts in sandy soils. Agriculture, 10(3), 69.

Kibet, L. C., Allen, A. L., Kleinman, P. J. A., Feyereisen, G. W., Church, C., Saporito, L. S., & Way, T. R. 2011. Phosphorus Runoff Losses from Subsurface-Applied Poultry Litter on Coastal Plain Soils. Journal of Environmental Quality, 40(2), 412–420.

Kirkham, M. B. 1982. Agricultural use of phosphorus in sewage sludge. In Advances in Agronomy (Vol. 35, Issue C).

Kladivko, E. J., & Nelson, D. W. 1979. Surface Runoff from Sludge-Amended Soils. Journal (Water Pollution Control Federation), 51(1), 100–110.

Kleinman, P. J. A., Needelman, B. A., Sharpley, A. N., & McDowell, R. W. 2003. Using soil phosphorus profile data to assess phosphorus leaching potential in manured soils. Soil Science Society of America Journal, 67(1), 215–224.

Kleinman, P. J. A., Sharpley, A. N., Moyer, B. G., & Elwinger, G. F. 2002. Effect of mineral and manure phosphorus sources on runoff phosphorus. Journal of Environmental Quality, 31(6), 2026–2033.

Kleinman, P. J. A., Sharpley, A. N., Wolf, A. M., Beegle, D. B., Elliott, H. A., Weld, J. L., & Brandt, R. 2006. Developing an Environmental Manure Test for the Phosphorus Index. Communications in Soil Science and Plant Analysis, 37(15–20), 2137–2155.

Kleinman, P. J. A., Wolf, A. M., Sharpley, A. N., Beegle, D. B., & Saporito, L. S. 2005. Survey of Water-Extractable Phosphorus in Livestock Manures. Soil Science Society of America Journal, 69(3), 701–708.

Larson, B. G. 1991. Saskatchewan Pork Industry Manure Management Recommendations. Saskatoon, SK, Canada: Saskatchewan Pork Producers' Marketing Board.

Leytem, A. B., Sims, J. T., & Coale, F. J. 2004. Determination of phosphorus source coefficients for organic phosphorus sources: laboratory studies. Journal of Environmental Quality, 33(1), 380—388.

Lichtenberg, E., Doug, P., & Lynch, L. 2002. Economic Value of Poultry Litter Supplies In Alternative Uses. Center for Agricultural and Natural Resource Policy. https://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.675.1762&rep=rep1&type=pdf

Liedl, B. E., Bombardiere, J., & Chatfield, J. M. 2006. Fertilizer potential of liquid and solid effluent from thermophilic anaerobic digestion of poultry waste. Water Science and Technology, 53(8), 69–79.

Magette, W. L., Ronan Hallissey, Kristen Hughes, and Emer Cosgrove. Eutrophication from Agricultural Sources : Field- and Catchment-Scale Risk Assessment. Environmental Protection Agency, 2007.

Maguire, R. O., Dou, Z., Sims, J. T., Brake, J., & Joern, B. C. 2005. Dietary Strategies for Reduced Phosphorus Excretion and Improved Water Quality. Journal of Environmental Quality, 34(6), 2093–2103.

Maguire, R. O., Hesterberg, D., Gernat, A., Anderson, K., Wineland, M., & Grimes, J. 2006. Liming Poultry Manures to Decrease Soluble Phosphorus and Suppress the Bacteria Population (b). Journal of Environmental Quality, 35(3), 849–857.

Maguire, R. O., Plumstead, P. W., & Brake, J. 2006. Impact of Diet, Moisture, Location, and Storage on Soluble Phosphorus in Broiler Breeder Manure (a). Journal of Environmental Quality, 35(3), 858–865.

Maguire, R. O., & Sims, J. T. 2002. Soil Testing to Predict Phosphorus Leaching. Journal of Environmental Quality, 31(5), 1601–1609.

Mahmood, F., Khan, I., Ashraf, U., Shahzad, T., Hussain, S., Shahid, M., Abid, M., & Ullah, S. 2017. Effects of organic and inorganic manures on maize and their residual impact on soil physico-chemical properties. Journal of Soil Science and Plant Nutrition, 17, 22–32.

Malik, M. A., Marschner, P., & Khan, K. S. 2012. Addition of organic and inorganic P sources to soil – Effects on P pools and microorganisms. Soil Biology and Biochemistry, 49, 106–113.

Mallin, M. A., & Cahoon, L. B. 2020. The Hidden Impacts of Phosphorus Pollution to Streams and Rivers. BioScience, 70(4), 315–329.

Maltais-Landry, G., Neufeld, K., Poon, D., Grant, N., Nesic, Z., & Smukler, S. 2018. Protection from wintertime rainfall reduces nutrient losses and greenhouse gas emissions during the decomposition of poultry and horse manure-based amendments. Journal of the Air & Waste Management Association, 68(4), 377-388.

Marshall, S. K., & Laboski, C. A. M. 2006. Sorption of Inorganic and Total Phosphorus from Dairy and Swine Slurries to Soil. Journal of Environmental Quality, 35(5), 1836–1843.

Marszałek, M., Kowalski, Z., & Makara, A. 2019. The Possibility of Contamination of Water-Soil Environment as a Result of the Use of Pig Slurry. Ecological Chemistry and Engineering S, 26, 313–330.

Massé, D. I., Talbot, G., & Gilbert, Y. G. 2011a. On farm biogas production: a method to reduce GHG emissions and develop more sustainable livestock operations. Animal Feed Science and Technology, 166, 436–445.

Massé, D. I., Talbot, G., & Gilbert, Y. 2011b. A scientific review of the agronomic, environmental and social benefits of anaerobic digestion. Anaerobic Digestion: Processes, Products and Applications, 109–131.

Mcconnell, D., Doody, D., Elliott, C., Matthews, D., & Ferris, C. 2016. Impact of slurry application method on phosphorus loss in runoff from grassland soils during periods of high soil moisture content. Irish Journal of Agricultural and Food Research, 55.

McDowell, R. W., & Sharpley, A. N. 2001. Soil phosphorus fractions in solution: influence of fertiliser and manure, filtration and method of determination. Chemosphere, 45(6), 737–748.

McGrath, J. M., Sims, J. T., Maguire, R. O., Saylor, W. W., Angel, C. R., & Turner, B. L. 2005. Broiler diet modification and litter storage: impacts on phosphorus in litters, soils, and runoff. Journal of Environmental Quality, 34(5), 1896–1909.

Mishra, A., Benham, B. L., & Mostaghimi, S. 2006. Sediment and Nutrient Losses from Field-Scale Cropland Plots Treated With Animal Manure and Inorganic Fertilizer. Water, Air, and Soil Pollution, 175(1), 61–76.

Montgomery, M. B., Ohno, T., Griffin, T. S., Honeycutt, C. W., & Fernanadez, I. J. 2005. Phosphorus Mineralization and Availability in Soil Amended with Biosolids and Animal Manures. Biological Agriculture & Horticulture, 22(4), 321–334.

Moore Jr., P. A., Daniel, T. C., & Edwards, D. R. 2000. Reducing Phosphorus Runoff and Inhibiting Ammonia Loss from Poultry Manure with Aluminum Sulfate. Journal of Environmental Quality, 29(1), 37–49.

Mostaghimi, S., Younos, T. M., & Tim, U. S. 1992. Effects Of Sludge And Chemical Fertilizer Application on Runoff Water Quality 1. JAWRA Journal of the American Water Resources Association, 28(3), 545–552.

Murnane, J. G., Fenton, O., & Healy, M. G. 2018. Impacts of zeolite, alum and polyaluminum chloride amendments mixed with agricultural wastes on soil column leachate, and CO2 and CH4 emissions. Journal of Environmental Management, 206, 398–408.

Nanzer, S., Oberson, A., Berger, L., Berset, E., Hermann, L., & Frossard, E. 2014. The plant availability of phosphorus from thermo-chemically treated sewage sludge ashes as studied by 33P labeling techniques. Plant and Soil, 377(1–2), 439–456.

Nguyen, M. L., & Goh, K. M. 1992. Nutrient cycling and losses based on a mass-balance model in grazed pastures receiving long-term superphosphate applications in New Zealand: 1. Phosphorus. The Journal of Agricultural Science, 119(1), 89–109.

Nicholson, F. A., Chambers, B. J., & Smith, K. A. 1996. Nutrient composition of poultry manures in England and Wales. Bioresource Technology, 58(3), 279–284.

Nicholson, F., Bhogal, A., Taylor, M., McGrath, S., & Withers, P. 2018. Long-term Effects of Biosolids on Soil Quality and Fertility. Soil Science, 183(3), 89–98.

Noack, S. R., McLaughlin, M. J., Smernik, R. J., McBeath, T. M., & Armstrong, R. D. 2012. Crop residue phosphorus: speciation and potential bio-availability. Plant and Soil, 359, 375-385.

Nolan, T., Troy, S. M., Gilkinson, S., Frost, P., Xie, S., Zhan, X., Harrington, C., Healy, M. G., & Lawlor, P. G. 2012. Economic analyses of pig manure treatment options in Ireland. Bioresource Technology, 105, 15–23.

NRW. 2022. Compliance Assessment of Welsh River SACs Against Phosphorus Targets. Available from: https://naturalresources.wales/evidence-and-data/research-and-reports/water-reports/compliance-assessment-of-welsh-river-sacs-against-phosphorus-targets/?lang=en

O'Bryan, C. A., Crandall, P., Jaroni, D., Ricke, S. C., & Gibson, K. E. 2017. Assessment of nitrogen and phosphorus loads present in environments impacted by alternative poultry processing operations utilized in pasture-raised poultry production. Renewable Agriculture and Food Systems, 32(1), 33-42.

O'Donnell, C., Egan, A., Harrington, J., Barnett, D., Forrestal, P., & Power, N. 2021. An overview on deficit and requirements of the Irish national soil phosphorus balance. Science of The Total Environment, 785, 147251.

O'Flynn, C. J., Fenton, O., Wilson, P., & Healy, M. G. 2012. Impact of pig slurry amendments on phosphorus, suspended sediment and metal losses in laboratory runoff boxes under simulated rainfall. Journal of Environmental Management, 113, 78–84.

Olsen, S. R., & Watanabe, F. S. 1970. Diffusive supply of phosphorus in relation to soil textural variations. Soil Science, 110, 318–327.

O'Rourke, S. M., Foy, R. H., Watson, C. J., Ferris, C. P., & Gordon, A. 2010. Effect of Varying the Phosphorus Content of Dairy Cow Diets on Losses of Phosphorus in Overland Flow Following Surface Applications of Manure. Journal of Environmental Quality, 39(6), 2138–2146.

Omeira, N., Barbour, E. K., Nehme, P. A., Hamadeh, S. K., Zurayk, R., & Bashour, I. 2006. Microbiological and chemical properties of litter from different chicken types and production systems. Science of The Total Environment, 367(1), 156–162.

Orr, R. J., Murray, P. J., Eyles, C. J., Blackwell, M. S. A., Cardenas, L. M., Collins, A. L., Dungait, J. A. J., Goulding, K. W. T., Griffith, B. A., Gurr, S. J., Harris, P., Hawkins, J. M. B., Misselbrook, T. H., Rawlings, C., Shepherd, A., Sint, H., Takahashi, T., Tozer, K. N., Whitmore, A. P., Lee, M. R. F. 2016. The North Wyke Farm Platform: effect of temperate grassland farming systems on soil moisture contents, runoff and associated water quality dynamics. European Journal of Soil Science, 67(4), 374–385.

Owens, P. N., Deeks, L. K., Wood, G. A., Betson, M. J., Lord, E. I., & Davison, P. S. 2008. Variations in the depth distribution of phosphorus in soil profiles and implications for model-based catchment-scale predictions of phosphorus delivery to surface waters. Journal of Hydrology, 350(3), 317–328.

Pagliari, P. H., & Laboski, C. A. M. 2012. Investigation of the Inorganic and Organic Phosphorus Forms in Animal Manure. Journal of Environmental Quality, 41(3), 901–910.

Pautler, M. C., & Sims, J. T. 2000. Relationships Between Soil Test Phosphorus, Soluble Phosphorus, and Phosphorus Saturation in Delaware Soils. Soil Science Society of America Journal, 64(2), 765–773.

Penn, C. J., & Sims, J. T. 2002. Phosphorus Forms in Biosolids-Amended Soils and Losses in Runoff. Journal of Environmental Quality, 31(4), 1349–1361.

Peyton, D. P., Healy, M. G., Fleming, G. T. A., Grant, J., Wall, D., Morrison, L., & Fenton, O. 2016. Nutrient, metal and microbial loss in surface runoff following treated sludge and dairy cattle slurry application to an Irish grassland soil. Science of the Total Environment, 541, 218-229.

Pierson, S. T., Cabrera, M. L., Evanylo, G. K., Kuykendall, H. A., Hoveland, C. S., McCann, M. A., & West, L. T. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. Journal of Environmental Quality, 30(5), 1784–1789.

Pierzynski, G. M., McDowell, R. W., & Thomas Sims, J. 2005. Chemistry, Cycling, and Potential Movement of Inorganic Phosphorus in Soils. In Phosphorus: Agriculture and the Environment (pp. 51–86). John Wiley & Sons, Ltd.

Pizzeghello, D., Berti, A., Nardi, S., & Morari, F. 2011. Phosphorus forms and P-sorption properties in three alkaline soils after long-term mineral and manure applications in north-eastern Italy. Agriculture, ecosystems & environment, 141(1-2), 58-66.

Potash Development Association. 2005. Soil analysis: Key to nutrient management planning. Potash Development Association.

Pote, D. H., Daniel, T. C., Nichols, D. J., Sharpley, A. N., Moore Jr., P. A., Miller, D. M., & Edwards, D. R. 1999. Relationship between Phosphorus Levels in Three Ultisols and Phosphorus Concentrations in Runoff. Journal of Environmental Quality, 28(1), 170–175.

Rashid, M., Price, N. T., Gracia Pinilla, M. Á., & O'Shea, K. E. 2017. Effective removal of phosphate from aqueous solution using humic acid coated magnetite nanoparticles. Water Research, 123, 353–360.

Preedy, N., McTiernan, K., Matthews, R., Heathwaite, L., & Haygarth, P. 2001. Rapid Incidental Phosphorus Transfers from Grassland. Journal of Environmental Quality, 30(6), 2105–2112.

Psenner, R., Boström, B., & Pettersson, K. 1988. Fractionation of phosphorus in suspended matter and sediment. Verh. Internat. Verein. Limnol., 22(January), 219–228.

Quilbé, R., Quilbé, Q., Serreau, C., Wicherek, S., Bernard, C., Yol`, Y., Thomas, Y., & Oudinet, J.-P. 2005. Nutrient Transfer By Runoff From Sewage Sludge Amended Soil Under Simulated Rainfall. Environmental Monitoring and Assessment, 100, 177–190.

Requejo, M. I., & Eichler-Löbermann, B. 2014. Organic and inorganic phosphorus forms in soil as affected by long-term application of organic amendments. Nutrient Cycling in Agroecosystems, 100, 245-255.

Rigby, H., Clarke, B. O., Pritchard, D. L., Meehan, B., Beshah, F., Smith, S. R., & Porter, N. A. 2016. A critical review of nitrogen mineralization in biosolids-amended soil, the associated fertilizer value for crop production and potential for emissions to the environment. Science of the Total Environment, 541, 1310–1338.

Rothrock, M., Gibson, K., Micciche, A., & Ricke, S. 2019. Pastured Poultry Production in the United States: Strategies to Balance System Sustainability and Environmental Impact. Frontiers in Sustainable Food Systems, 3.

Rothwell, S.A., Forber, K.J., Dawson, C.J., Salter, J.L., Dils, R.M., Webber, H., Maguire, J., Doody, D.G. and Withers, P.J.A., 2022. A new direction for tackling phosphorus inefficiency in the UK food system. Journal of Environmental Management, 314, p.115021.

Romeis, J. J., Jackson, C. R., Risse, L. M., Sharpley, A. N., & Radcliffe, D. E. 2011. Hydrologic and Phosphorus Export Behavior of Small Streams in Commercial Poultry-Pasture Watersheds1. JAWRA Journal of the American Water Resources Association, 47(2), 367–385.

Rowarth, J. S., Gillingham, A. G., Tillman, R. W., & Syers, J. K. 1985. Release of phosphorus from sheep faeces on grazed, hill country pastures. New Zealand Journal of Agricultural Research, 28(4), 497–504.

Rowe, M. (2017). Free-range poultry farming contributing to increase in river pollution, warn authorities. Available from: https://www.countryfile.com/wildlife/wildlife-stories/free-range-poultry-farming-contributing-to-increase-in-river-pollution-warn-authorities (accessed 26/01/2024)

Russell, E. 1962. Soil conditions and plant growth (G. and C. Longmans (ed.); 9th ed.).

Sauer, T. J., Daniel, T. C., Moore Jr., P. A., Coffey, K. P., Nichols, D. J., & West, C. P. 1999. Poultry Litter and Grazing Animal Waste Effects on Runoff Water Quality. Journal of Environmental Quality, 28(3), 860–865.

Sauer, T. J., Daniel, T. C., Nichols, D. J., West, C. P., Moore Jr., P. A., & Wheeler, G. L. 2000. Runoff Water Quality from Poultry Litter-Treated Pasture and Forest Sites. Journal of Environmental Quality, 29(2), 515–521.

Scarlat, N., Dallemand, J.-F., & Fahl, F. 2018. Biogas: Developments and perspectives in Europe. Renewable Energy, 129, 457–472.

Schroeder, P. D., Radcliffe, D. E., & Cabrera, M. L. 2004. Rainfall Timing and Poultry Litter Application Rate Effects on Phosphorus Loss in Surface Runoff. Journal of Environmental Quality, 33(6), 2201–2209.

Schuster, N., Bartelt-Hunt, S., Durso, L., Gilley, J. E., Li, X., Marx, D., Schmidt, A., & Snow, D. 2017. Runoff Water Quality Characteristics Following Swine Slurry Application under Broadcast and Injected Conditions. Transactions of the ASABE, 60, 53–66.

Serna, M. D., & Pomares, F. 1992. Nitrogen mineralization of sludge-amended soil. Bioresource Technology, 39(3), 285–290.

Shappell, N. W., Billey, L. O., & Shipitalo, M. J. 2016. Estrogenic activity and nutrient losses in surface runoff after winter manure application to small watersheds. Science of The Total Environment, 543, 570–580.

Sharpley, A., & Moyer, B. 2000. Phosphorus Forms in Manure and Compost and Their Release during Simulated Rainfall. Journal of Environmental Quality, 29(5), 1462–1469.

Sharpley, A. N. 1995. Dependence of runoff phosphorus on extractable soil phosphorus. Wiley Online Library.

Sharpley, A. N. 1997. Rainfall Frequency and Nitrogen and Phosphorus Runoff from Soil Amended with Poultry Litter. Journal of Environmental Quality, 26(4), 1127–1132.

Sharpley, A. N., & Menzel, R. G. 1987. The Impact of Soil and Fertilizer Phosphorus on the Environment (N. C. B. T.-A. in A. Brady (ed.); Vol. 41, pp. 297–324). Academic Press.

Sharpley, A.N., Daniel, T., Sims, T., Lemunyon, J., Stevens, R., & Parry, R. 2003. Agricultural Phosphorous and Eutrophication, second ed. US Department of Agriculture, Washington, D.C., p. 38.

Shepherd, M. A., & Withers, P. J. 2001. Phosphorus leaching from liquid digested sewage sludge applied to sandy soils. Journal of Agricultural Science, 136(4), 433–441.

Shepherd, M., & Bhogal, A. 1998. Regular applications of poultry litter to a sandy arable soil: effects on nitrate leaching and nitrogen balance. J. Sci. Food Agric, 78, 19–29.

Shober, A. L., & Sims, J. T. 2007. Integrating phosphorus source and soil properties into risk assessments for phosphorus loss. Soil Science Society of America Journal, 71(2), 551-560.

Sibbesen, E., & Sharpley, A. N. 1997. Setting and justifying upper critical limits for phosphorus in soils. https://eurekamag.com/research/002/956/002956049.php

Siddique, M. T., & Robinson, J. S. 2003. Phosphorus Sorption and Availability in Soils Amended with Animal Manures and Sewage Sludge. Journal of Environmental Quality, 32(3), 1114–1121.

Siddique, M. T., & Robinson, J. S. 2004. Differences in phosphorus retention and release in soils amended with animal manures and sewage sludge. Soil Science Society of America Journal, 68(4), 1421-1428.

Sims, J. T. & Wolf, D. C. 1994. Poultry waste management: agricultural and environmental issues. Adv.
Agronomy, 52, 2-72.

Sims, J. T., Edwards, A. C., Schoumans, O. F., & Simard, R. R. 2000. Integrating Soil Phosphorus Testing into Environmentally Based Agricultural Management Practices. Journal of Environmental Quality, 29(1), 60–71.

Sims, J. T., Simard, R. R., & Joern, B. C. 1998. Phosphorus Loss in Agricultural Drainage: Historical Perspective and Current Research. Journal of Environmental Quality, 27(2), 277–293.

Smith, B. C., Andersen, D. S., Harmon, J. D., & Stinn, J. P. 2017. Case study of swine finishing manure nutrient characteristics for land application. In 2017 ASABE Annual International Meeting (p. 1). American Society of Agricultural and Biological Engineers.

Smith, B. C., Andersen, D. S., Harmon, J. D., & Stinn, J. P. 2017. Case Study of Swine Finishing Manure Nutrient Characteristics for Land Application. In 2017 ASABE Annual International Meeting (p. 1). ASABE.

Smith, D. R., Owens, P. R., Leytem, A. B., & Warnemuende, E. A. 2007. Nutrient losses from manure and fertilizer applications as impacted by time to first runoff event. Environmental Pollution, 147(1), 131–137.

Smith, D. R., Moore Jr., P. A., Griffis, C. L., Daniel, T. C., Edwards, D. R., & Boothe, D. L. 2001. Effects of Alum and Aluminum Chloride on Phosphorus Runoff from Swine Manure. Journal of Environmental Quality, 30(3), 992–998.

Smith, D. R., Jarvie, H. P., Harmel, R. D., & Haney, R. L. 2019. The Role of Field-Scale Management on Soil and Surface Runoff C/N/P Stoichiometry. Journal of Environmental Quality, 48(5), 1543–1548.

Smith, K. A., Jackson, D. R., & Withers, P. J. A. 2001. Nutrient losses by surface run-off following the application of organic manures to arable land. 2. Phosphorus. Environmental Pollution, 112(1), 53–60.

Soupir, M. L., Mostaghimi, S., & Yagow, E. R. 2006. Nutrient Transport from Livestock Manure Applied to Pastureland Using Phosphorus-Based Management Strategies. Journal of Environmental Quality, 35(4), 1269–1278.

Spargo, J. T., Evanylo, G. K., & Alley, M. M. 2006. Repeated Compost Application Effects on Phosphorus Runoff in the Virginia Piedmont. Journal of Environmental Quality, 35(6), 2342–2351.

Stevenson, F. J. 1982. Humus Chemistry. Wiley-Interscience:

Stiles, W. 2017. Poultry manure management.

Sustainable Energy Ireland. 2003. An Assessment of the Renewable Energy Resource Potential of Dry Agricultural Residues in Ireland. Phosphorus Forms in Manure and Compost and Their Release%0Dduring Simulated Rainfall

Sutton, A. L., Nelson, D. W., Hoff, J. D., & Mayrose, V. B. 1982. Effects of Injection and Surface Applications of Liquid Swine Manure on Corn Yield and Soil Composition. Journal of Environmental Quality, 11(3), 468–472.

Sutton, A. L., Nelson, D. W., Mayrose, V. B., & Nye, J. C. 1978. Effects of Liquid Swine Waste Applications on Corn Yield and Soil Chemical Composition. Journal of Environmental Quality, 7(3), 325–333.

Sutton, A. L., Nelson, D. W., Mayrose, V. B., Nye, J. C., & Kelly, D. T. 1984. Effects of Varying Salt Levels in Liquid Swine Manure on Soil Composition and Corn Yield. Journal of Environmental Quality, 13(1), 49–59.

Szogi, A. A., & Vanotti, M. B. 2009. Prospects for phosphorus recovery from poultry litter. Bioresource Technology, 100(22), 5461–5465.

Teagasc. 2017. Soil Index System. https://www.teagasc.ie/crops/soil--soil-fertility/soil-analysis/soil-index-system/

Tisdale, S. L., Nelson, W. L., & Beaton, J. D. 1985. Soil fertility and fertilizers. Collier Macmillan Publishers.

Toor, G. S., & Sims, J. T. 2016. Phosphorus Leaching in Soils Amended with Animal Manures Generated from Modified Diets. Journal of Environmental Quality, 45(4), 1385–1391.

Torbert, H. A., Potter, K. N., Hoffman, D. W., Gerik, T. J., & Richardson, C. W. 1999. Surface Residue and Soil Moisture Affect Fertilizer Loss in Simulated Runoff on a Heavy Clay Soil. Agronomy Journal, 91(4), 606–612.

Toth, J. D., Dou, Z., Ferguson, J. D., Galligan, D. T., & Ramberg Jr., C. F. 2006. Nitrogen- vs. Phosphorus-based Dairy Manure Applications to Field Crops. Journal of Environmental Quality, 35(6), 2302–2312.

Turner, B. L. 2004. Optimizing Phosphorus Characterization in Animal Manures by Solution Phosphorus-31 Nuclear Magnetic Resonance Spectroscopy. Journal of Environmental Quality, 33(2), 757–766.

Tuszynska, A., Czerwionka, K., & Obarska-Pempkowiak, H. 2021. Phosphorus concentration and availability in raw organic waste and post fermentation products. Journal of Environmental Management, 278(2), 111468.

Vadas, P. A., Gburek, W. J., Sharpley, A. N., Kleinman, P. J. A., Moore Jr., P. A., Cabrera, M. L., & Harmel, R. D. 2007. A Model for Phosphorus Transformation and Runoff Loss for Surface-Applied Manures. Journal of Environmental Quality, 36(1), 324–332.

Vadas, P. A., Meisinger, J. J., Sikora, L. J., McMurtry, J. P., & Sefton, A. E. 2004. Effect of poultry diet on phosphorus in runoff from soils amended with poultry manure and compost. Journal of Environmental Quality, 33(5), 1845–1854.

Vadas, P. A. 2006. Distribution of Phosphorus in Manure Slurry and Its Infiltration after Application to Soils. Journal of Environmental Quality, 35(2), 542–547.

Van den Broeck, M., Rhazi, L., Waterkeyn, A., Madihi, M., Grillas, P., Kneitel, J., & Brendonck, L. 2019. Livestock disturbances in Mediterranean temporary ponds: A mesocosm experiment with sheep manure and simulated trampling. Freshwater Biology, 64.

Vanden Bossche, H., Audic, J. M., Huyard, A., Gascuel-Odoux, C., Trolard, F., & Bourrié, G. 2000. Phosphorus losses from sewage sludge disposed on a field: Evidence from storm event simulations. Water Science and Technology, 42(9), 179–186.

Vogel, T., Kruse, J., Siebers, N., Nelles, M., & Eichler-Löbermann, B. 2017. Recycled Products from Municipal Wastewater: Composition and Effects on Phosphorus Mobility in a Sandy Soil. Journal of Environmental Quality, 46(2), 443–451.

Vučić, V., & Müller, S. 2021. New developments in biological phosphorus accessibility and recovery approaches from soil and waste streams. Engineering in Life Sciences, 21(3–4), 77–86.

Walker, P., & Kelley, T. 2003. Solids, organic load and nutrient concentration reductions in swine waste slurry using a polyacrylamide (PAM)-aided solids flocculation treatment. Bioresource Technology, 90(2), 151–158.

Welsh Government. 2020. June 2020 Survey of Agriculture and Horticulture: Results for Wales.

Welsh Government. 2022. Review of Welsh soil evidence.Avialable from: https://www.gov.wales/review-welsh-soil-evidence

Welsh Government. 2023. Survey of agriculture and horticulture: June 2023. https://www.gov.wales/survey-agriculture-and-horticulture-june-2023-html

Welsh Government. 2021. The Control of Agricultural Pollution Regulations: Summary of measures and dates from when they will apply. https://www.gov.wales/sites/default/files/publications/2021-01/the-water-resources-control-of-agricultural-pollution-wales-regulations-2021.pdf

Wheeler, E., Patterson, P., Xin, H., Gates, R., Gregory, C., Pekel, A., Burley, H. K., Adrizal, A., Topper, P., & Adviento-Borbe, M. 2010. Ammonia Emission, Manure Nutrients And Egg Production Of Laying Hens Fed Distiller Dried Grain Diets. ASABE - International Symposium on Air Quality and Waste Management for Agriculture 2010.

White, E. 1960. The Distribution and Subsequent Disappearance of Sheep Dung on Pennine Moorland. Journal of Animal Ecology, 29(2), 243–250.

Williams, C. 2020. Part 2: Greenhouse gases produced in pig rearing. Available from: https://businesswales.gov.wales/farmingconnect/news-and-events/technical-articles/part-2-greenhouse-gases-produced-pig-rearing (accessed 27th February 2024)

Withers, P. J. A., Flynn, N. J., Warren, G. P., Taylor, M., & Chambers, B. J. 2016. Sustainable management of biosolid phosphorus: a field study. Soil Use and Management, 32(1), 54–63.

Withers, P. J. A., Hodgkinson, R. H., Adamson, H., & Green, G. 2007. The impact of pasture improvement on phosphorus concentrations in soils and streams in an upland catchment in Northern England. Agriculture, Ecosystems & Environment, 122(2), 220–232.

Withers, P. J. A., Rothwell, S. A., Forber, K. J., & Lyon, C. 2022. Re-focusing Phosphorus use in the Wye Catchment. RePhOKUs. Available from: https://councillors.herefordshire.gov.uk/documents/s50101856/RePhoKUs_Wye_Report_310522.pdf (accessed 18th March 2024)

Withers, P. J. A., Clay, S. D., & Breeze, V. G. 2001. Phosphorus Transfer in Runoff Following Application of Fertilizer, Manure, and Sewage Sludge. Journal of Environmental Quality, 30(1), 180–188.

Withers, P. J. A., Davidson, I. A., & Foy, R. H. 2000. Prospects for Controlling Nonpoint Phosphorus Loss to Water: A UK Perspective. Journal of Environmental Quality, 29(1), 167–175.

Wu, L., Harris, P., Misselbrook, T. H., & Lee, M. R. F. 2022. Simulating grazing beef and sheep systems. Agricultural Systems, 195, 103307.

Xie, R., & MacKenzie, A. F. 1986. Urea and Manure Effects on Soil Nitrogen and Corn Dry Matter Yields. Soil Science Society of America Journal, 50(6), 1504–1509.

Xu, X., Wang, Y., Zhang, H., Yin, D., Dari, B., & Xu, J. 2020. Soil phosphorus storage capacity as affected by repeated phosphorus addition in an Ultisol. Communications in Soil Science and Plant Analysis, 51(14), 1960–1968.

Yang, X., Post, W. M., Thornton, P. E., & Jain, A. 2013. The distribution of soil phosphorus for global biogeochemical modelling. Biogeosciences, 10(4), 2525–2537.

Zahan, Z., & Othman, M. Z. 2019. Effect of pre-treatment on sequential anaerobic co-digestion of chicken litter with agricultural and food wastes under semi-solid conditions and comparison with wet anaerobic digestion. Bioresource Technology, 281, 286–295.

Zhang, Y. S., Werner, W., Scherer, H. W., & Sun, X. 1994. Effect of organic manure on organic phosphorus fractions in two paddy soils. Biology and Fertility of Soils, 17(1), 64–68.

Appendices

Appendix A. Literature search results and filtering

A.1 Poultry

See accompanying spreadsheet.

A.2 Pig

See accompanying spreadsheet.

A.3 Sheep

See accompanying spreadsheet.

A.4 Cow

See accompanying spreadsheet.

A.5 Food waste

See accompanying spreadsheet.

A.6 Biosolids

See accompanying spreadsheet.

A.7 Farmyard/livestock

See accompanying spreadsheet.

Appendix B. Comparison table

See accompanying spreadsheet.

Appendix C. Key Articles

See accompanying spreadsheet.

Appendix D. Interview meeting minutes

D.1 Prof. Philip Haygarth

Separate document.

D.2 Dr Shane Rothwell

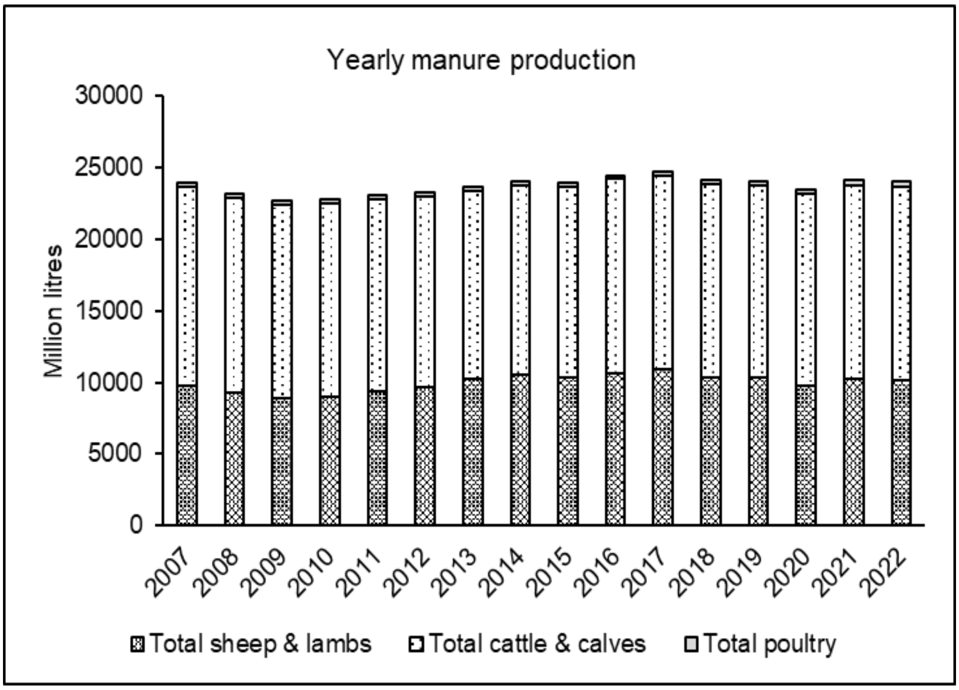
Separate document.

D.3 Prof. Rishi Prasad

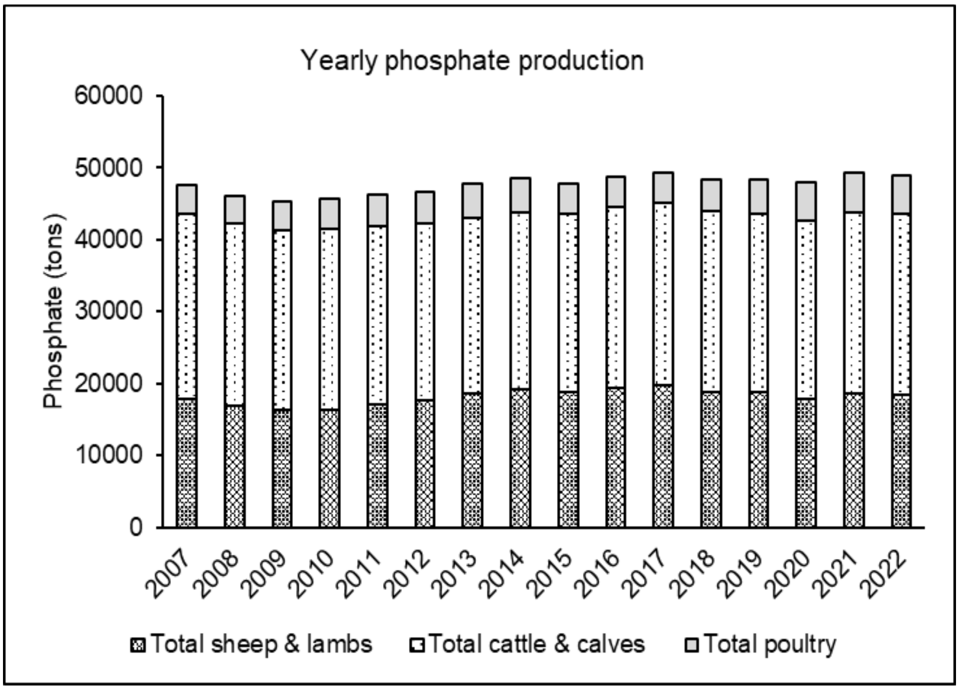
Separate document.

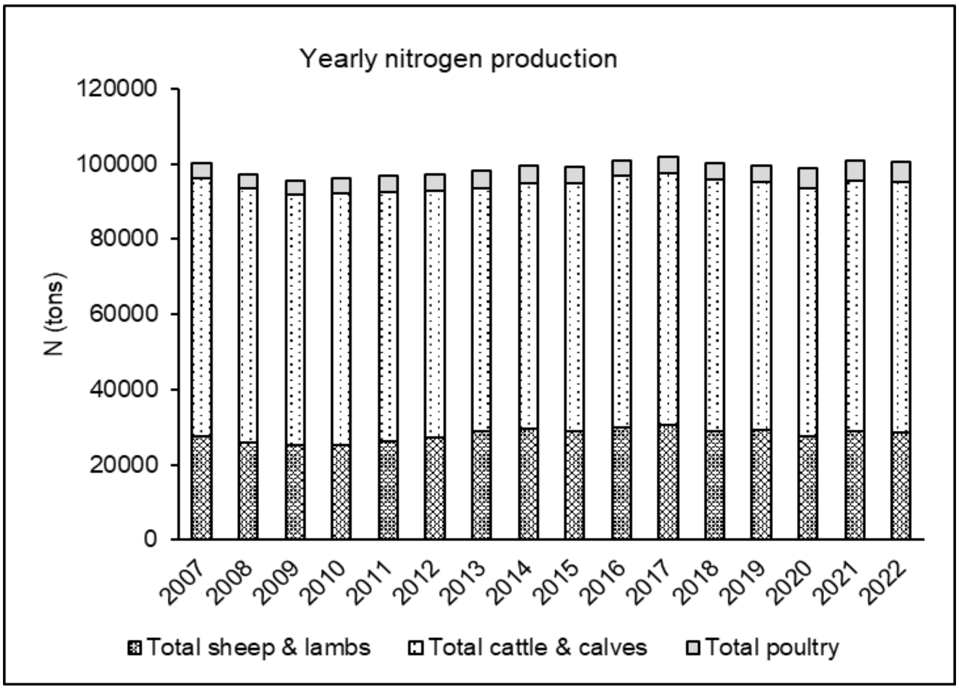
Appendix E. Manure production in Wales and manure phosphate and N production (2007 – 2022)

The figures in this appendix present Wales yearly manure production, yearly phosphate production and yearly nitrogen production across the period 2007 – 2022. Data on livestock numbers (cattle, sheep and poultry only) was provided by Natural Resources Wales (NRW)[39] and manure, nitrogen and phosphate production figures were obtained from Schedule 1 of the The Control of Agricultural Pollution (Wales) Regulation (CoAPR) (Welsh Government, 2021). Total livestock numbers and average production figures were used to calculate the values presented in the figures below.



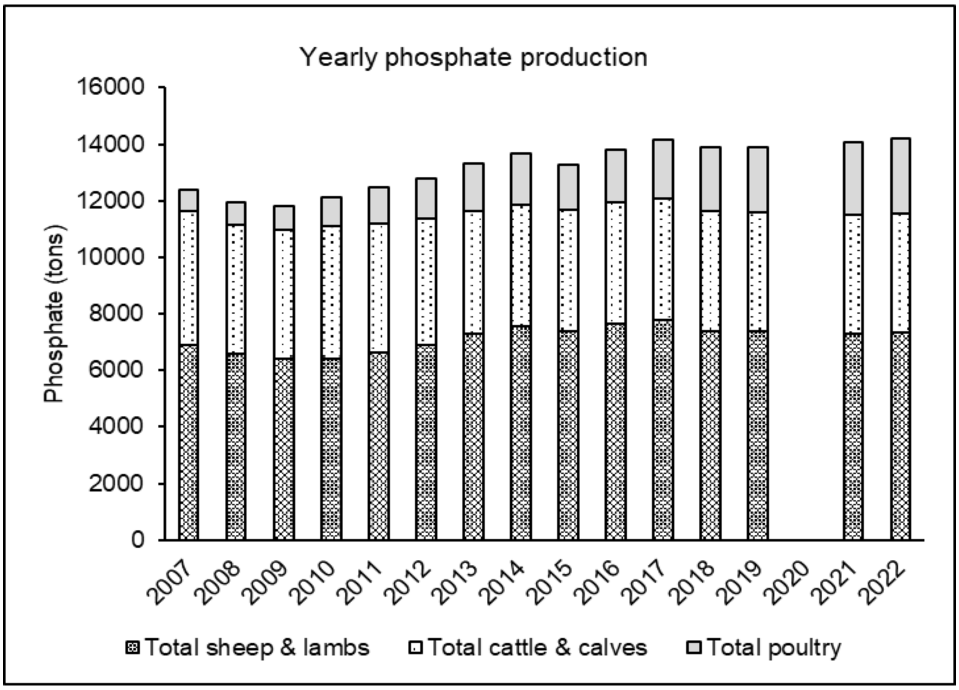
[39] Email received from Dave Johnston (NRW, 12/02/2024)


Yearly phosphate production
Phosphate (tons)
60000
50000
40000
30000
20000
10000
0
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
Total sheep & lambs
Total cattle & calves
Total poultry


Yearly nitrogen production
N (tons)
120000
100000
80000
60000
40000
20000
0
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
Total sheep & lambs
Total cattle & calves
Total poultry

Appendix F. Manure phosphate production in Powys (2007 – 2022)

The figure in this appendix presents Powys yearly phosphate production across the period 2007 – 2022. Data on livestock numbers (cattle, sheep and poultry only) was provided by Natural Resources Wales (NRW)[40] and phosphate production figures were obtained from Schedule 1 of the The Control of Agricultural Pollution (Wales) Regulation (CoAPR) (Welsh Government, 2021). Total livestock numbers and average production figures were used to calculate the values presented in the figure. Note: regional breakdown was not available in 2020 due to smaller scale survey carried out during COVID-19 pandemic.



[40] Email received from Dave Johnston (NRW, 12/02/2024)

Data Archive Appendix

Data outputs associated with this project are archived on server–based storage at Natural Resources Wales.

The data archive contains:

[A] The final report in Microsoft Word and Adobe PDF formats.

[B] Spreadsheets containing the full article list downloaded from Scopus and the outcomes of the 2-stage filtering process.

[C] The meeting notes from the interviews with academics in Adobe PDF format.

Metadata for this project is publicly accessible through Natural Resources Wales' Library Catalogue https://libcat.naturalresources.wales (English Version) and https://catllyfr.cyfoethnaturiol.cymru (Welsh Version) by searching 'Dataset Titles'. The metadata is held as record no 161354.

Further copies of this report are available from library@cyfoethnaturiolcymru.gov.uk