Water UK Biosolids Network-Club Project

Review of the evidence base to support the application of low readily available nitrogen organic manures to agricultural land

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Submitted to:

Water UK Biosolids Network
c/o Simon Black
Anglian Water Services Limited:
Lancaster House, Lancaster Way,
Ermine Business Park, Huntingdon,
Cambridgeshire, PE29 6XU

Prepared by:

Fiona Nicholson, Alison Rollett, Anne Bhogal,
James Dowers and John Williams
ADAS Boxworth, Battlegate Road, Boxworth,
Cambridge CB23 4NN
Contact: john.williams@adas.co.uk

Matt Taylor
Grieve Strategic, Garden House,
Back Street, Shipston on Stour,
Warwickshire, CV36 4JL

EXECUTIVE SUMMARY

This study was commissioned by Water UK's Biosolids Network to review the evidence underpinning the management practices enacted in UK legislation to minimise diffuse pollution from applications of organic manures with a low readily available nitrogen (RAN) content, including biosolids. It also aimed to understand the implications of the Environment Agency (EA)'s interpretation of the Farming Rules for Water on the balance of diffuse pollutant losses to air and water following biosolids application to land, and on practical aspects of biosolids management.

The Farming Rules for Water were introduced by Defra in April 2018 (Defra, 2018; SI, 2018) to fulfil obligations on diffuse pollution under the Water Framework Directive. Rule 1 of the Farming Rules for Water relates to the application of organic manures and manufactured fertiliser and states (Defra, 2018):

a) *Application of organic manures and manufactured fertilisers to cultivated agricultural land must be nutrient management planned to meet soil and crop nutrient needs without exceeding these levels and assessed for significant risk of pollution in advance.*

b) *Nutrient Management Planning must take into account the results of testing for Phosphorus, Potassium, Magnesium, pH and Nitrogen levels in the soil, which must be done at least every 5 years*.

A closed period for low RAN manures is not stipulated in the Farming Rules for Water. However, the EA's current interpretation of the Farming Rules for Water does not permit autumn application of organic materials unless crops have a recognised need for autumn nitrogen (N) (e.g. winter oilseeds and grass) to support late season growth in August and September. Under this interpretation applications of biosolids before the establishment of winter cereals would not comply.

Water quality

Evidence from the literature suggests that leaching losses from low RAN livestock manures (cattle and pig farmyard manure -FYM) applied to free draining soils in England are typically <5-10% of total N applied, and on drained clay soils less than 5% of total N applied. From the limited evidence base available, low RAN biosolids products (e.g. dewatered cake) behave similarly to FYM, with losses following autumn (September) applications to free draining soils 7-11% of the N applied, dropping to <3% of the N applied following November and December applications. Indeed this is what MANNER-*NPK* predicts in terms of leaching losses following application of low RAN biosolids products.

Biosolids have low water soluble phosphorus (P) concentrations (typically <10% of total P) and so losses to water are likely to be low. Autumn applications, which are soil incorporated, pose a low risk of P loss via surface runoff. In contrast, applications made to soils with low moisture deficits in the spring and which are left on the soil surface, present a greater surface runoff risk.

Air quality

Biosolids pose a low risk of ammonia and nitrous oxide emission as a result of their low RAN content. Evidence from studies using livestock FYM suggests that ammonia-N losses following applications of these manures are *c.*5% of total N applied. Most ammonia is lost soon after application (80% within 24 hours) and rapid soil incorporation will reduce losses. Management strategies that prevent the rapid soil incorporation of biosolids (e.g. moving applications on winter cereals/autumn stubbles to topdressing on growing crops in spring) will increase ammonia losses. In contrast, nitrous oxide emissions from low RAN manures are largely controlled by soil and weather conditions in the period after application, with no consistent effect of application timing or incorporation method.

Crop available N supply from contrasting biosolids application timings

Outputs from MANNER-*NPK* suggest that the greatest risk of nitrate leaching losses from biosolids is following early autumn (August and September) applications to sandy soils under arable production in high rainfall areas. Nitrate leaching losses from medium/heavy soils were predicted to be lower than from applications to light/sandy soils reflecting the greater water holding capacity of the former. Crop available N supply from biosolids applications was lowest following autumn applications to light sandy soils at 10% of total N applied, with no impact of rapid soil incorporation. This reflects the high risk of nitrate leaching on these soils and suggests that any N saved as a result of reduced ammonia emissions following rapid soil incorporation is subsequently lost by leaching. On medium/heavy soils the crop available N supply from autumn applications is similar to that from spring surface broadcast applications at 15% of total N applied. Thus nitrate leaching losses avoided by applying biosolids in spring are balanced by increased ammonia emissions – an example of pollution swapping.

Practical considerations

Delaying biosolids applications until spring will increase the risks of soil compaction from application machinery. Outputs from the IRRIGUIDE model suggest that soils would not be dry enough to support the weight of application machinery until the end of March in low and moderate rainfall areas, and early/mid-April in high rainfall areas. Delaying applications until late March/early-April is likely to compromise spring crop establishment especially on medium and heavy soils. Applying low RAN biosolids products to growing crops is likely to cause significant crop damage, which will substantially reduce crop yields. It is also likely to increase odour nuisance since material will be left on the soil surface. If applications to spring planted crops could not be made until late March this would lead to planting delays and reduced crop yields.

Key messages

Based on a review of the available literature, and modelling of diffuse pollution losses and soil conditions, this study has found that:

- Biosolids have a low RAN content (typically <20% total N) and consequently pose a low risk of N loss to the environment.
- The risks to the environment from biosolids applications vary according to soil type:
    - On light sandy soils, nitrate leaching is the main loss pathway for autumn applied biosolids. Avoiding spreading in August and September will reduce the environmental impact of biosolids applied to these soil types.
    - On medium/heavy soils in arable production where autumn sown crops predominate, the risk of soil compaction and elevated ammonia emissions from spring applications outweighs any marginal benefits which may arise from reduced nitrate leaching. Consequently, autumn timings on these soil types provide the best overall outcome in terms of minimising the environmental impacts of biosolids applications

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1. Introduction

1.1 Background.

The Farming Rules for Water (formally known as The Reduction and Prevention of Agricultural Diffuse Pollution (England) Regulations 2018) were introduced by Defra in April 2018 (Defra, 2018; SI, 2018) to fulfil obligations on diffuse pollution under the Water Framework Directive (WFD). In England, 17% of 4,950 individual water bodies (917 water bodies) are currently at good or better overall WFD status. The agriculture and rural land management sector is responsible for 30% of those water bodies failing to meet their WFD objectives.

An impact assessment published in 2018 stated that Defra's objective for introducing the Farming Rules for Water was: *"to establish a basic standard of mandatory good practice through the introduction of new basic rules that meet the requirements of the Water Framework Directive without gold-plating. These good practice rules will provide a foundation for water companies, NGOs, voluntary actions and government incentives to build upon to contribute to better farming practices and deliver further reductions in agricultural pollution. Our aim is to reduce diffuse water pollution from agriculture, in a way that minimises costs to the farming sector".*

The rules included in the impact assessment relating to the management of organic materials were:

Organic manures and manufactured fertiliser planning, storage and application, storage

1. A person who has custody or control of agricultural land must ensure that when organic manures and manufactured fertilisers are applied to that land that all reasonable precautions are taken to prevent causing environmental pollution from significant soil erosion or runoff. That person must also ensure that: a) application of organic manures and manufactured fertilisers must be planned in advance to meet and not exceed soil and crop needs, and b) soil testing must be carried out for Phosphorus, Potassium, Magnesium and pH, and Nitrogen levels assessed, at least every 5 years, for cultivated land.

2. Organic manures must not be stored on land: a) within 10 metres of inland freshwaters or coastal waters, b) where there is significant risk of runoff* entering inland freshwaters or coastal waters c) within 50 metres of a spring, well or borehole.

3. A person must not apply organic manures or manufactured fertilisers: a) if the soil is water logged, flooded, or snow covered b) if the soil has been frozen for more than12 hours in the previous 24 hours c) if there is significant risk of causing pollution from soil erosion and run-off.

4. A person must not apply organic manures: a) within 10 metres of inland freshwaters or coastal waters b) within 50 metres of a spring, well or borehole.

5. A person must not apply manufactured fertiliser within 2 metres of inland freshwaters or coastal waters.

Soil management

6. A person who has custody or control of agricultural land must take all reasonable precautions to prevent significant soil erosion and or muddy runoff that could enter inland freshwaters or coastal waters especially from: a) seedbeds, tramlines, rows, beds, stubbles (including harvested land with haulm), polytunnels and irrigation b) poaching by livestock

7. Any land within 5 metres of inland freshwaters or coastal waters must be protected from significant soil erosion (including bankside erosion) or significant runoff by preventing poaching by livestock.

8. Livestock feeders must not be positioned: a) within 10 metres of inland freshwaters or coastal waters, b) where there is significant risk of runoff* from poaching around the feeder entering any inland freshwaters or coastal waters.

Defra consulted on the proposed Farming Rules for Water rules in 2015 and published a summary of the responses (Defra, 2017). These were subsequently used to revise the rules to make them "more practical and easier to follow" for their introduction in April 2018 (Defra, 2018).

The regulations require all land managers to ensure that each application of organic manure or manufactured fertiliser is planned so that it does not exceed the needs of the soil and crop or give rise to a significant risk of agricultural diffuse pollution.

The rules published in 2018 (Defra, 2018) state that:

1a) Application to cultivated land must be planned in advance to meet soil and crop nutrient needs and not exceed these levels

1b) Planning must take into account where there is a significant risk of pollution and the results of testing for Phosphorus (P), Potassium (K), Magnesium (Mg), pH and N levels in the soil, which must be done at least every 5 years. Soil N levels may be determined by assessing the soil N supply instead of testing the soil.

Organic manures must not be applied:

3a) if the soil is waterlogged, flooded, or snow covered

3b) if the soil has been frozen for more than 12 hours in the previous 24 hours

3c) if there is significant risk of causing pollution from soil erosion and run-off

4a) within 10 metres of any inland freshwaters or coastal waters, except, if precision equipment is used, within 6 metres of inland freshwaters or coastal waters

4b) within 50 metres of a spring, well or borehole

Since publication there have been issues regarding interpretation of Rule 1 and their implications. Of particular concern is the recent interpretation by the Environment Agency (EA) of Rule 1a which states that: "A land manager must ensure that, for each application of organic manure or manufactured fertiliser to agricultural land, the application is planned so that it does not (i) exceed the needs of the soil and crop on that land, or (ii) give rise to a significant risk of agricultural diffuse pollution" (SI, 2018).

In 2019, in response to concerns from its members, the NFU sought a clearer explanation of this rule and received a clarification note from the EA which stated (Tried and Tested, 2019):

- Farming rules for water do not impose a ban on the application of any organic manure or manufactured fertiliser
- The nutrient needs of each farm and field can be different in terms of what is required, and when, to meet crop and soil needs. What the farmer needs to know is that anything beyond that, or applications that pose a significant risk of pollution are likely to represent breaches of rules 1-5 of the Farming rules for water
- When assessing compliance Environment Agency officers will consider organic manure and manufactured fertiliser applications, and their planning, on a field by field basis

Recent clarification of the EA's interpretation of the Farming Rules for Water restricts autumn application of organic materials to crops which have a recognised need for autumn nitrogen (e.g. winter oilseeds and grass) to support late season growth in August and September. Under this interpretation applications of biosolids before the establishment of winter cereals would not comply with the Farming Rules for Water.

1.2 Risks of diffuse pollution from organic material applications.

Applications of organic materials pose significant risk of diffuse and point source water and air pollution (Figure 1). Nutrient management planning aims to ensure that applications of organic materials and fertiliser meet but do not exceed crop requirements. As an organic material commonly applied to agricultural land, biosolids are valuable sources of plant nutrients and if used effectively they can reduce the need for applications of manufactured fertilisers to meet optimum crop needs. Fertiliser recommendation systems (e.g. RB209, PLANET, MANNER-*NPK* and other supplementary information) provide guidance on how to make full allowance of the nutrients applied in biosolids and reduce manufactured fertiliser inputs accordingly.

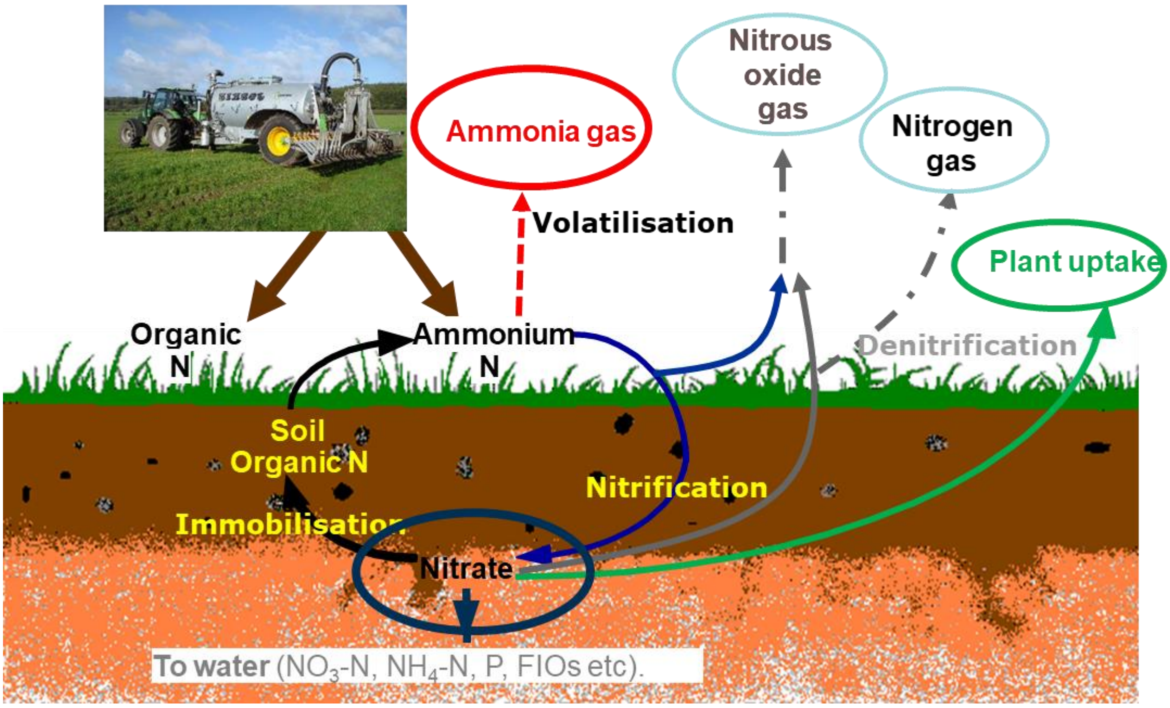

Figure 1. Nitrogen loss pathways following organic material application to land

The risks of nitrate leaching from applications of organic materials vary according to manure type. Nitrogen (N) is present in organic materials such as livestock manures and biosolids in two main forms:

- Readily available N (RAN) i.e. ammonium-N, nitrate-N and uric acid-N (poultry manure only) is N that is potentially available for rapid crop uptake.
- Organic-N is contained in organic forms, which are broken down slowly to become potentially available for crop uptake over a period of months to years.

Organic materials with a high RAN content are defined in Nitrate Vulnerable Zone (NVZ) legislation as those where more than 30% of the total N content is present as RAN. They include livestock slurry (from cattle and pigs), poultry manures, liquid anaerobic digestate or liquid digested sewage sludge. In contrast, low RAN organic materials are those where <30% of the total N is in crop available form. These are generally solid materials such as farmyard manure (FYM) from cattle or pigs, compost and biosolids (digested sludge cake, thermally dried sludge, lime stabilised sludge).

1.3 Study aims and scope

The aim of this study was to review the scientific evidence base which underpinned the introduction of management practices to minimise the risks of diffuse pollution from low RAN manure applications, including biosolids. It also sought to understand the implications that the EA's interpretation of the Farming Rules for Water would have on the balance of pollutant losses to air and water and on practical aspects of biosolids management.

This review focused on low RAN organic materials, and on biosolids in particular, but drew on the evidence base for high RAN organic materials if and when appropriate.

2. Current regulations, good practice and guidance

This section presents an overview of current regulations, codes of good practice and guidance relating to the spreading of low RAN organic manures to agricultural land in the UK and elsewhere in Europe.

2.1 NVZ legislation.

The EU Nitrates Directive (Council Directive 91/676/EEC) was adopted in 1991 to reduce water pollution caused by nitrates from agricultural sources. It requires that member states designate as Nitrate Vulnerable Zones (NVZs) areas of land that drain into polluted waters and to set up an Action Programme (AP) in these zones. However it was not until 1996 that regulations were made applying NVZ designation to 8% (approximately 600,000 hectares) of England and in 1998 the first NVZ-AP came into force; following consultation, the AP was reviewed and modified in 2002, 2007 and again in 2011.

The most recent iteration of the NVZ-AP in England is enforced in law under the Nitrate Pollution Prevention Regulations 2015 (SI, 2015). Applications of all organic manures (including biosolids) to agricultural land in designated NVZs must comply with this legislation. The NVZ-AP restricts the amount of N that can be applied with organic manures to individual fields to 250 kg/ha total N in a 12 month period (the 'field limit'). A separate 'whole farm' limit requires that a maximum of 170 kg/ha of N in organic manure (including manure deposited directly by livestock and spreading) can be applied across a holding in each calendar year.

Organic manures with a high RAN content must not be spread to land during designated closed periods, the timing of which depends on soil type and cropping (Table 1).

Table 1. 'Closed periods' for spreading manures with readily available N contents greater than 30% of total N

	Grassland	Tillage land*
Sandy or shallow soils	1 September to 31 December	1 August to 31 December
All other soils	15 October to 31January	1 October to 31 January

* On tillage land with sandy or shallow soils, application is permitted between 1 August and 15 September, provided a crop is sown on or before 15 September

Whilst these closed spreading periods do not apply to manures with a low RAN content, the N supplied by low RAN manures still contributes to the field and whole farm N limits. Other NVZ-AP restrictions relevant to spreading of low RAN organic manures in NVZs are that:

- They cannot be spread:
    - if a field is waterlogged, flooded or covered in snow;
    - if a field is frozen for more than 12 hours in the previous 24 hours;
    - within 50 metres of a spring, well or borehole or 10 metres of surface water.
- They must only be spread on agricultural land being used to grow crops (including grass).
- They must be spread as accurately as possible.
- They must be worked into the soil (unless spread as mulch on sandy soil) as soon as possible, and within 24 hours at the latest if the land is sloping and within 50 metres of surface water that could receive run-off from it.

2.2 NVZ timing restrictions in other countries

Whilst there are currently are no restrictions on the timing of low RAN manure applications in NVZs in England (or Wales and Scotland), restrictions are in place in Northern Ireland and some EU member states as follows:

Northern Ireland. The Nutrients Action Programme 2019-2022 (DAERA/NIEA, 2019) specifies that:

- Organic manures, including slurry, poultry litter, digestate, sewage sludge and abattoir waste, must not be applied from midnight 15 October to midnight 31 January.
- Farmyard manure (FYM) must not be applied from midnight 31 October to midnight 31 January.

Republic of Ireland. The 2017 Nitrates Action Programme (ISB, 2017) specifies closed spreading periods as follows:

- Spreading of FYM is not permitted between 1 November and 12, 15 or 31 January, depending on geographical area.
- Spreading of any other organic manure (e.g. slurry) is not permitted between 15 October and 12, 15 or 31 January, depending on geographical area.

The Netherlands. Solid pig and cattle manure applications are not permitted on grassland between 1 September and 1 February on sandy soils and light loams, and from 16 September to 1 February on clay and peat soils. On arable land, solid pig and cattle manure applications are not permitted on sandy soils and light loams between 1 September and 1 February. There are no restrictions on the timing of solid pig and cattle manure applications to arable land on clay and peat soils.

Denmark. Solid manure applications are not permitted between 15 November and 1 February. (The Livestock Manure Order, 2012)

Although there are restrictions on the timing of low RAN manure applications in these countries, with the exception of applications to grassland and applications to arable crops on sandy soils in the Netherlands, no other country limits applications in the autumn before the 15 October. Closed periods that begin in late autumn/early winter will reduce the risks of low RAN manures being spread on 'wet' soils which will have most benefit in reducing nutrient losses in runoff and have limited impact on reducing nitrate leaching losses.

2.3 Codes of Practice.

The various Codes of Good Agricultural Practice which apply to the devolved nations of the UK describe additional key actions that farmers should follow to protect and enhance water, soil and air quality. The codes complement the NVZ regulations by limiting the amount of N that can be applied with organic manures to any agricultural land to 250kg/ha total N in a 12 month period.

In England the Code of Good Agricultural Practice for Farmers, Grower and Land Managers (Defra, 2009) provides detailed guidance on good practice for the application of livestock manures to reduce fertiliser costs, improve soil structure and reduce the risk of causing pollution. This includes advice on the timing of applications and restrictions on applications in certain areas. The advice and guidance provided in the code for organic material storage and applications is consistent with the requirements of the Farming Rules for Water.

2.4 Nutrient Management Guide (RB209).

The AHDB Nutrient Management Guide (RB209) offers best practice guidance on the application of mineral fertilisers, manures and slurries to crops and grassland. RB209 aims to help farmers and land managers make the most of organic materials, and balance the benefits of fertiliser use against the economic and environmental costs.

The guide recommends that to make best use of their N content, organic materials should be applied at or before times of maximum crop growth, which is generally during the late winter to summer period. Delaying applications until late winter or spring will reduce nitrate leaching and increase manure N use efficiency; this is particularly important for organic materials with a high RAN content and where applications are made to sandy/shallow soils.

RB209 provides detailed guidance on the recommended timings applications of manufactured nitrogen fertiliser for different crops.

- For winter sown wheat and barley there is no requirement for manufactured fertiliser N in the seedbed.
- Autumn applications of manufactured fertiliser N can be applied to the seedbed of autumn sown oilseeds or as a top dressing to encourage autumn growth, although research suggests that crops sown after early September are unlikely to respond.
- For grass, most manufactured fertiliser N should be applied in spring or early summer when sward demand is greatest.

There are intrinsic differences between manufactured fertiliser and organic manures, particularly low RAN organic manures such as biosolids cake. Manufactured fertilisers typically contain nutrients which are in a highly available form (e.g. 100% water soluble). They also have a high nutrient density, in that they contain a high quantity of nutrient per tonne (e.g. ammonium nitrate is 34.5% N), and are manufactured in liquid or granule (prill) forms which can be applied from tramlines. These properties enable them to be top-dressed in spring using comparatively light machinery, from tramlines without driving on and damaging the growing crop. By contrast, low RAN organic materials contain nutrients in a less available form (<30% of total N is readily available) and they have low nutrient density (e.g. approximately 1% of the fresh weight is N). They are also bulky and require different equipment to spread them accurately which often cannot operate from modern tramlines. As a result of these very different properties, there are different issues to consider when managing low RAN organic materials and following advice aimed at manufactured fertilisers is not always appropriate to maximise their value and minimise environmental damage.

Table 2. Crop available N supply from contrasting biosolids applications to different soil types – AHDB's Nutrient Management Guide (RB209).

Biosolids	Autumn[a] (Aug–Oct, 450 mm rainfall to end March)		Winter[a] (Nov–Jan, 250 mm rainfall to end March)		Spring[a] (Feb–Apr)	Summer[a] use on grassland
	Sandy/shallow[b]	Medium/heavy[b]	Sandy/shallow[b]	Medium/heavy[b]	All soils	All soils
Surface-applied, i.e. not soil-incorporated						
Digested cake	10	15	15	15	15	15
Thermally dried	10	15	15	15	15	15
Lime-stabilised	10	15	15	15	15	15
Composted	10	15	15	15	15	15
Soil-incorporated after application – 6 hours for liquids and 24 hours for solids[c]						
Digested cake	10	15	15	15	20	N/A
Thermally dried	10	15	15	15	20	N/A
Lime-stabilised	10	15	15	15	20	N/A
Composted	10	15	15	15	15	N/A

RB209 also provides guidance on the crop available N supply from contrasting manure application timings. For nutrient management planning it is essential that the crop available N (and other nutrients) supplied by organic materials is deducted from the manufactured fertiliser N recommendations. This ensures that the risk of applying excess nutrients is reduced which minimises fertiliser costs and nutrient losses to the environment.

The guidance on crop available N supply from contrasting biosolids application timings to different soil types suggests that crop available N supply from autumn applications on light sandy soils is lowest at 10% of total N applied, with no impact of rapid soil incorporation (Table 2). This reflects the high risk of nitrate leaching on these soil types and suggests that any N saved as a result of reduced ammonia emissions following rapid soil incorporation is subsequently lost by leaching.

On medium/heavy soils the crop available N supply from autumn applications is similar to that from spring surface broadcast applications at 15% of total N applied which suggests that any nitrate leaching losses prevented by applying in the spring are balanced by increased ammonia emissions – an example of pollution swapping.

2.6 Specific controls on biosolids applications to land

There are a number of regulations and codes of practice in place to ensure that biosolids recycling to agricultural land is undertaken safely:

- The Sludge (Use in Agriculture) Regulations (SI, 1989) restrict the quantities of potentially toxic elements (PTEs) that can be applied to land from biosolids. The regulations place legally binding limits on the amounts of zinc, cadmium, lead, copper, chromium, mercury and nickel in biosolids that can be applied. The regulations also provide maximum soil PTE concentrations above which biosolids cannot be applied. The Regulations are complemented by the Code of Practice for the Agriculture Use of Sewage Sludge (DoE, 1996) which set lower soil limits for some PTEs, and in addition provide recommendations on maximum loading rates for molybdenum, arsenic, selenium and fluoride.
- The Safe Sludge Matrix (ADAS, 2001) aims to minimise the risks of microbial pathogen contamination of food from the application of biosolids to agricultural land. The matrix restricts applications of biosolids to those that have been treated to reduce microbial pathogen levels. 'Conventionally' treated biosolids (e.g. digested cake) can only be applied to combinable or animal feed crops, and to grassland if there is a no grazing in the season after application. Enhanced treated sludges (e.g. thermally-hydrolysed cake) can be applied before all crops as long as there is a 10-month harvest interval for fruit, salads, vegetables and horticulture crops and 3 week no-grazing interval on grassland.
- The Biosolids Nutrient Management Matrix (ADAS, 2018) restricts the frequency of biosolids applications based on the soil P index. At soil P index 0, 1 and 2 biosolids can be applied annually, whilst at soil P index 3 and 4 the return periods are restricted according to soil and biosolids product type. At soil P index 5 and above biosolids applications are not permitted.

The Biosolids Assurance Scheme (BAS) brings together the legislative and code of practice controls on biosolids recycling into one independently audited standard which has been adopted by the Water Industry. This standard includes all the restrictions on biosolids use included in the guidance and regulations described above.

3. Evidence based review

In this section, we present a review of the scientific evidence base underpinning the current UK rules and guidance for spreading low RAN organic manures to agricultural land. This includes the evidence relating to the RAN content of different organic manures and pollutant losses via nitrate leaching, surface runoff (phosphorus), and ammonia and greenhouse gas (nitrous oxide and methane) emissions, considering how these pollutant losses routes are affected by different manure management practices as well as soil, environment and climatic conditions.

3.1 Nitrogen content of low RAN manures

Livestock manures

The Nutrient Management Guide (RB209) provides data on the 'typical' nutrient contents of livestock manures based on samples analysed as part of research projects and other national sampling programmes. The data for cattle, sheep, horse and goat FYM were most recently reviewed and updated by Munro et al (2016) as part of Defra Project WT1569, whist the duck and pig FYM data were reviewed and updated in Defra Project SCF0202 (Nicholson and Misselbrook, 2015).

Table 3 shows the typical total N contents of the different types of FYM from RB209. The RAN content of the manures is no longer provided in RB209 except in the form of pie charts, because it can be confused with the crop available N supply which depends on application timing and speed of incorporation. However, we have used the data the previous edition of RB209 (Anon, 2010) and Munro et al (2016) which show that the typical RAN contents of all the FYM types are below 30% of total N (Table 3).

Table 3. Typical total N and RAN contents of different FYM types

FYM type	Dry matter (%)[1]	Total N (kg N/t)[1]	RAN (% total N)[2]
Cattle	25	6.0	10-20
Pig	25	7.0	15-25
Sheep	25	7.0	10-20
Duck	25	6.5	15-25
Horse	25	5.0	10[3]
Goat	40	9.5	5[3]

[1]The Nutrient Management Guide (RB209)
[2]The Fertiliser Manual (RB209) 8th Edition (Anon, 2010)
[3]Munro et al (2016)

Biosolids

Data from RB209 on the typical total N contents of biosolids are shown in Table 4. As with livestock manure, the RAN content of biosolids is no longer given in RB209; however, data from the previous edition of RB209 (Anon, 2010) show that the typical RAN contents of all the biosolids types are between 5 and 15% of total N (Table 4).

These data were most recently reviewed in 2016 by Williams et al. (2016). A database of the total N contents of biosolids products applied to agricultural land in England and Wales was obtained from the UK Water Industry Biosolids Network based on the analysis of a large number of samples from water companies across England and Wales. The data indicated that the total N contents of all biosolids products in the 8th Edition of RB209 (Anon, 2010) were similar to the more recent data supplied by the Water Industry and therefore did not need to be revised. However, the dry matter content of composted and lime stabilised biosolids were changed to 40% and 25%, respectively (Table 4). Information on the RAN content of biosolids is not routinely collected by the Water Industry, so Williams et al (2016a) used additional data from LINK project 0988 (ref), and the Water Industry/AHDB funded OPTI-S project (Sagoo et al., 2018). The RAN content of the biosolids products was 23% of total N for digested cake, 6% for thermally dried, 3% for lime stabilised and 7% for composted biosolids; these were very similar to those reported in Anon (2010) therefore no changes to the values were recommended. As a comparison, Rigby et al (2016) collated data on TN concentrations for all biosolids types reported in the global literature. These ranged from 0.7–15% on a dry solids (DS) basis, with overall mean and median values of 4.1% DS and 4.4% DS, respectively, compared to the RB209 values in Table 4 which equate to TN concentration of 2.8 – 4.4% DS.

As part of this study, data was supplied by the 9 English water companies (Anglian Water, Northumberland Water, Severn Trent Water, South West Water, Southern Water, Thames Water, United Utilities, Wessex Water and Yorkshire Water) For digested cake, lime stabilised and composted biosolids the Water Company data on dry matter and total N contents are in good agreement with those in RB209 (Table 4). The RAN content digested cake (13% TN) is very similar to the RB209 value (15%), whilst for lime stabilised biosolids the RAN content (5% TN) is about half the value in RB209 (10% TN). All the biosolids types (including the advance treated material) had RAN contents of <25% TN which confirms their classification as low RAN organic manures.

Table 4. Typical N content of biosolids (RB209) compared with means calculated from data supplied by 8 Water Companies for this project)

Biosolids type	Source	Dry matter (%)	Total N (kg N/t)	RAN (% total N)
Digested cake	RB209[1]	25	11	15[2]
	Water Company	25	11	13
Thermally dried	RB209[1]	95	40	5[2]
	Water Company[3]	-	-	-
Lime stabilised	RB209[1]	25	8.5	10[2]
	Water Company	31	9.1	5
Composted	RB209[1]	40	11	5[2]
	Water Company	-	-	-
Advanced AD	RB209[1]	-	-	-
	Water Company	27	12	20

[1]Data from The Nutrient Management Guide (RB209)
[2]Data from The Fertiliser Manual (RB209) 8th Edition (June 2010)
[3]Thermally dried biosolids are no longer produced in England

3.2 Nitrate leaching

Factors affecting nitrate leaching

The factors affecting nitrate (NO_3-N) leaching losses following autumn/winter organic material applications to soils are well understood. They are the same for all types of organic manures and include the rate, method and timing of application, soil type, cropping/ground cover and rainfall/drainage following application. Nitrate leaching losses are determined primarily by the amount of 'available' N (i.e. NO_3-N and NH_4-N) remaining in the soil at the start of over-winter drainage and the movement of water through the soil over-winter.

Following an autumn/winter application of organic manure, the amount of available N remaining in the soil at the start of drainage and therefore at risk of leaching depends on:

- **Quantity of available N applied**. Slurry and poultry manures are 'high' in RAN and typically contain c.35-70% of total N as RAN. Cattle and pig FYM are 'low' in RAN and typically contain c.10-25% of total N as RAN (Table 3); similarly, biosolids are low in RAN containing <25% of total N as RAN (Table 4). A proportion of the organic N applied with organic manures can also become available following application by the process of mineralisation (i.e. the 'mineralisable N' content). Rigby et al (2016) conducted a comprehensive review of the mineralisable N content of different biosolids products and concluded that the amount was directly proportional to the total organic N content and degree of organic matter stability, and decreased in the order aerobically digested (not applicable to UK products) > thermally dried > lime stabilised > anaerobically digested > composted. However, as the rate of mineralisation is temperature dependent, and drops to low levels at temperatures < 5 ^{o}C, the amount released prior to the start of drainage and therefore at risk of leaching in the autumn/winter is likely to be negligible (Bhogal et al., 2016).
- **Ammonia volatilisation losses**. Ammonia emissions following land spreading will reduce the amount of available N remaining in the soil and at risk of leaching. Therefore strategies that reduce ammonia losses (i.e. soil incorporation or bandspreading/ shallow injection of slurry) may increase NO_3-N leaching losses ('pollution swapping').
- **Crop N uptake in the autumn**. Where manures are applied in the autumn/winter either to a growing crop or prior to autumn establishment of a crop, N uptake in the autumn period prior to the start of over-winter drainage will reduce the amount of available N remaining in the soil and at risk of leaching. Autumn crop uptake is typically greatest by grass or well established oilseeds (c.20 kg/ha) and lower for cereals (c.5-10 kg/ha). A well-established cover crop can also take up between 30 and 100 kg/ha N (White et al., 2016) and can be very effective at reducing nitrate leaching losses (Bhogal et al., 2020).

The movement of water through the soil over-winter will depend on:

- **Rainfall**. The quantity of rainfall between the date of manure application and the end of soil drainage is known as the 'effective' rainfall. Leaching losses will be greater in high rainfall areas than in low rainfall areas; it will also be greater from earlier autumn application timings than later timing as the quantity of rainfall between the date of application and end of drainage is increased.
- **Soil texture and the way water moves through the soil**. On free draining sandy soils drainage occurs via matrix flow, with NO_3-N moving down with infiltrating water as it displaces soil water. In contrast, on poorly drained medium and heavy textured soils, surface runoff is likely to occur in rapid response to rainfall events, because of the impermeable nature of the soil

matrix. Where an effective drainage system is present, much of the water that would otherwise be lost as surface runoff, will move rapidly from the soil surface through macropores that have developed naturally or have been created through the installation of pipe drains, mole drains or sub-soiling fissures, with transit times influenced by rainfall volume and intensity.

The current NVZ manure and fertiliser application rules are based upon these principles, with closed spreading periods in place for organic materials with a high RAN content (>30% of total N content – i.e. slurries, digestates and poultry manures) which vary with soil type (earlier start and end date for shallow/sandy soils) and cropping (grass vs arable). There are currently no restrictions on the timing of cattle, pig or sheep FYM, biosolids or compost applications within NVZs as the RAN content of these materials is typically relatively low. The following sections summarise the evidence base that underpins these rules.

Nitrate leaching following livestock manure applications

A large body of research was undertaken in the UK pre-2000 on NO_3-N leaching from free draining soils which pose the greatest risk of leaching loss. This research provided the evidence base for the current NVZ closed-spreading periods for high RAN manures (Defra projects NT1402, NT1410 and OC896; Beckwith et al., 1998, Chambers et al, 2000). Nitrate leaching losses were highest following applications of slurry and poultry manure which typically have RAN contents greater than 30% of total N, whereas losses from FYM applications were lower reflecting their lower RAN content (typically 20-25% of total N for fresh FYM that has not been stored before application and c.10% of total N for FYM that has been stored for more than 3 months). For the slurry/poultry manure applications, NO_3-N leaching losses following September, October and November applications were typically in the range 10-20% of total N applied, whilst N losses following applications in December or January were not significantly elevated above those from untreated controls. Nitrate leaching losses from September, October and November FYM applications were lower than from the slurry/poultry manures at between 3 and 7% of total N applied (Figure 2). More recent results from Defra project AC0116 (Defra, 2015) have confirmed these findings. Here, the highest nitrate leaching losses were measured on a sandy loam soil following a September application of layer manure, broiler litter or pig slurry (where NO_3-N losses amounted to 14-25% of the total N applied) ahead of a winter wheat crop, compared to a loss equivalent to 3% of the total N applied following the application of pig FYM.

On heavier textured soils (e.g. medium/heavy soils), the research evidence in England quantifying nutrient leaching losses following solid manure applications is limited to data from Defra project WQ0118 'Cracking clays'. Here, experiments were carried out over four drainage seasons (2007/08 to 2010/11) on hydrologically isolated cracking clay experimental sites at ADAS Faringdon (Oxon.), ADAS Boxworth (Cambs.) and Rowden North Wyke (Devon). The Faringdon and Rowden sites only included slurry applications, however the Boxworth site included both slurry (cattle and pig) and solid manure applications (cattle FYM, pig FYM, broiler litter and layer manure). At ADAS Boxworth, the solid manures and slurries were applied to arable stubbles in the autumn and incorporated into the soil 1-2 days after application. In 2007/08 and 2009/10 solid manures and slurries were applied before the establishment of winter wheat and in 2008/09 and 2009/10 solid manure applications were made before the establishment of winter oilseeds.

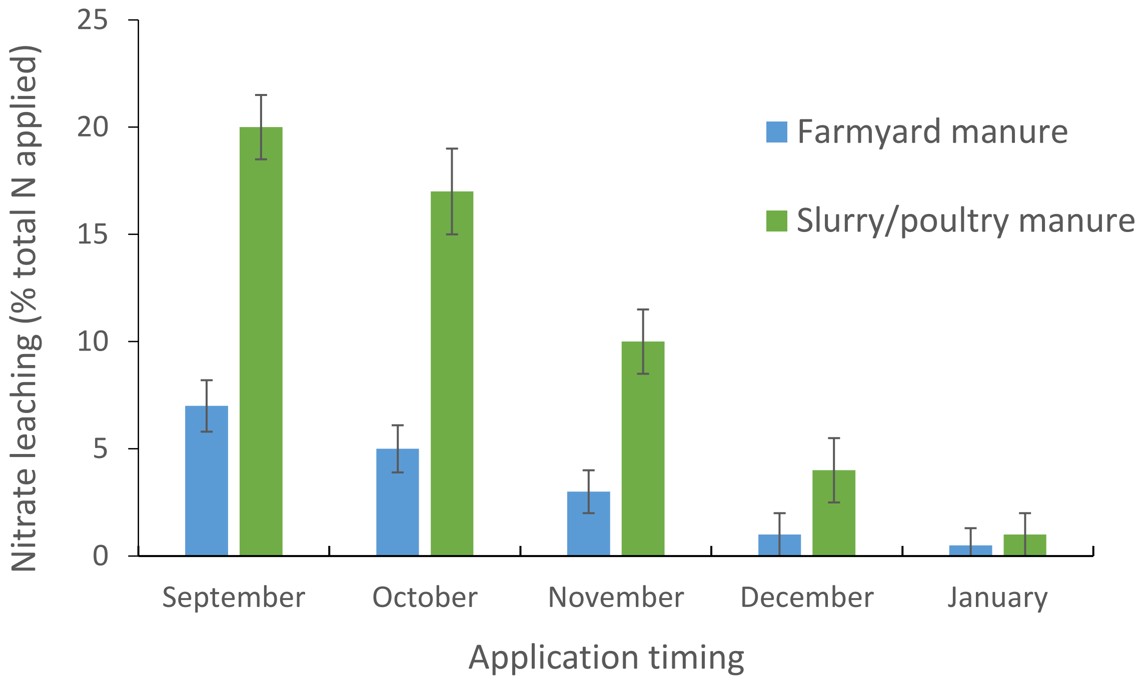

Figure 2. Nitrate leaching losses following different application timings of FYM and high RAN manures to arable free draining sandy and shallow soils over chalk (1990/1 to 1993/4) (Chambers *et al*., 2000)

Nitrate leaching losses from the autumn applied pig and cattle FYM were very low (<1% of total N applied for cattle FYM applied in 2007/08 and 2008/09, and c.1% of total N applied for pig FYM applied in 2009/10 and 2010/11), reflecting the low proportion of manure total N present as RAN (Table 5). In contrast, leaching losses were greatest from pig slurry (c.13-16% of total N applied) and poultry manure (c.8-12% of total N applied) applied to winter wheat in 2007/08 and 2009/10, reflecting the high RAN content of these manures (Table 5) and low uptake of manure N by the winter wheat crop between application and the start of drainage. Nitrate leaching losses following the autumn slurry and poultry manure applications before the drilling of winter oilseeds were lower (<5% of total N applied) than from winter cereal cropped land; reflecting the uptake of manure N by the actively growing oilseed crop.

There is a large body of international research measuring NO_3-N leaching from high RAN pig slurry, cattle slurry and poultry manures, but not from solid pig/cattle FYM. Those studies that have investigated land application of solid pig/cattle FYM have focussed on ammonia volatilisation losses or crop N recovery of manure N. For example, Hansen et al (2004) measured greater crop manure N recovery from spring compared to autumn cattle FYM applications, presumably due to NO_3-N leaching losses from the autumn applications, however NO_3-N leaching losses were not measured as part of the study.

Table 5. Nitrate leaching losses following autumn applications of contrasting manure types to a heavy clay soil (Defra project WQ0118)

Manure type	Manure RAN content (%)	Leaching loss (% total N applied)
Autumn applications to winter wheat (2007/08)		
Cattle slurry	39	1
Pig slurry	84	13
Cattle FYM	15	0
Broiler litter	30	8
Autumn applications to winter oilseeds (2008/09)		
Cattle slurry	44	0
Pig slurry	88	1
Cattle FYM	5	0
Broiler litter	28	0
Autumn applications to winter wheat (2009/10)		
Cattle slurry	53	4
Pig slurry	81	16
Pig FYM	9	1
Layer manure	31	12
Autumn applications to winter oilseeds (2010/11)		
Cattle slurry	60	4
Pig slurry	83	6
Pig FYM	6	1
Layer manure	46	2

Nitrate leaching following biosolids applications

There has been little recent UK work investigating leaching losses from biosolids. However, Rigby et al (2016) reference two major research programmes completed in the UK in the 1990s which studied N losses from biosolids-amended soil (Smith et al 1994; Misselbrook et al 1996, Shepherd 1996).

Smith et al. (1994) found that N recovery by a winter wheat crop following liquid mesophilic anaerobic digested biosolids applied and incorporated into the soil increased from 25 to 55% of total N applied following October and December applications, respectively. Although it was not directly measured, they attributed this to greater losses via nitrate leaching following the October application. Shepherd (1996) measured nitrate leaching losses following applications of raw liquid, digested liquid and dewatered digested cake, applied in September, November or January ahead of either winter barley or spring barley, with the liquid sludges either injected or surface applied. September applications were applied prior to ploughing and drilling winter barley, or to fallow plots ahead of spring barley; later applications were only to the fallow plots ahead of spring barley. The earlier the application the greater the leaching risk, with losses highest following the injected liquid digested sludge (which had the highest readily available N content) and the lowest losses following broadcast dewatered cake applications. Figure 3 is a re-working of these results, showing the effect of date of application. Similar to the findings with livestock manures, losses were highest following September applications and liquid digested biosolids (with an average RAN content of 58%) where losses were 35-40% of the N applied, compared to dewatered cake (with an average RAN content of 20%) where losses were 7-11% of the N applied. Nitrate leaching following November and December applications (to fallow land) were considerably lower, ranging from 2-3% of the N applied for both liquid and solid biosolids. These losses equated to a total N loss of 4 – 17 kg ha^{-1} from dewatered cake compared with 3-98 kg ha^{-1} from digested liquid. Shepherd (1996) concluded that "liquid digested sludge had a high leaching risk (similar to animal slurry), and dewatered cake was less susceptible to leaching (akin to FYM)".

Misselbrook (1996) conducted a similar study to Shepherd (1996), but biosolids applications were made to two freely draining grassland sites. Applications were made in autumn, winter and spring (following first silage cut), with raw sludge applied by injection and digested sludge both as a surface application and injection. Dewatered cake was not included in this study. Up to 24, 11 and 6% of the applied total N was leached from injected digested, surface-applied digested and injected raw sludge respectively, with autumn applications giving rise to greater losses than winter application. Injection of digested liquid sludge as compared to surface application exacerbated nitrate leaching losses, because preventing N loss via ammonia volatilisation meant that more N was available to be lost through leaching.

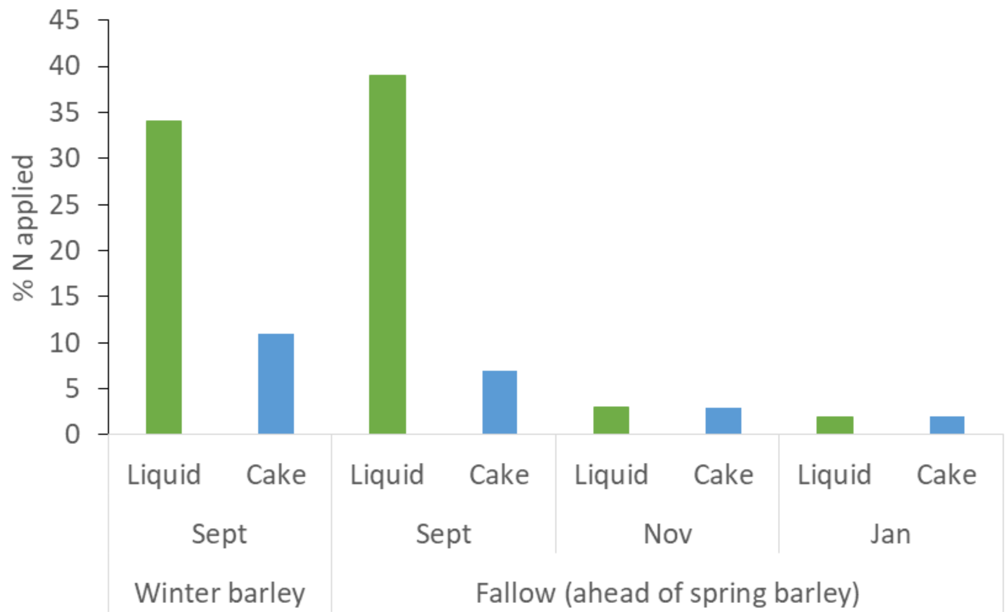

Figure 3. Nitrate leaching following digested liquid and dewatered biosolids cake applications to a sandy soil in 1991 (re-drawn using data from Shepherd, 1996).

Since this work in the early 1990's we could find no UK field-based studies on nitrate leaching following biosolids additions. A targeted literature review of more recent work specifically relating to biosolids was undertaken using web of science, restricting the search to papers published since 2010, using the search terms 'biosolids and nitrate leaching'. This search gave 44 papers which were subsequently reduced to 11 papers on the basis of their titles. None of these papers were UK studies and most were either laboratory incubation studies using 'leaching columns' or lysimeters.

Summary

There have been very few studies measuring nitrate leaching losses from biosolids additions. Of those which have been conducted, the basic principles established from studies with livestock manures have been confirmed, i.e. that the amount of N lost via leaching is dependent on biosolids type (particularly its readily available N content), time and method of application.

Studies which have compared NO_3-N leaching losses from autumn applied pig/cattle FYM with leaching losses from pig/cattle slurries and poultry manures have consistently shown lower leaching losses from FYM (typically 10-15% of total N as RAN), than slurries and poultry manures (typically >30% of total N as RAN). For low RAN livestock manures (cattle and pig FYM) applied to free draining soils in England nitrate leaching losses are typically <5-10% of total N applied, and on drained clay soils less than 5% of total N applied. From the limited evidence base available, low RAN biosolids products (e.g. dewatered cake) behave similarly to a FYM, with losses following autumn (September) applications to free draining soils 7-11% of the N applied, dropping to <3% of the N applied following November and December applications.

3.3 Phosphorus

Introduction

Sustainable phosphorus (P) use is increasingly important both agronomically (i.e. ensuring sufficient supply so as not to limit crop yields) and environmentally (i.e. not causing excess losses to watercourses). It is therefore very important to understand the effects of biosolids (and other low RAN manure) applications on soil P supply and eutrophication risk. Modern agriculture is dependent on P additions to sustain crop yields, maintain soil fertility and replenish nutrients removed during crop harvest. Both livestock manures and biosolids contain valuable amounts of P and organic matter, and are increasing viewed as more sustainable alternatives to resource depleting inorganic P fertilisers (Withers et al., 2015).

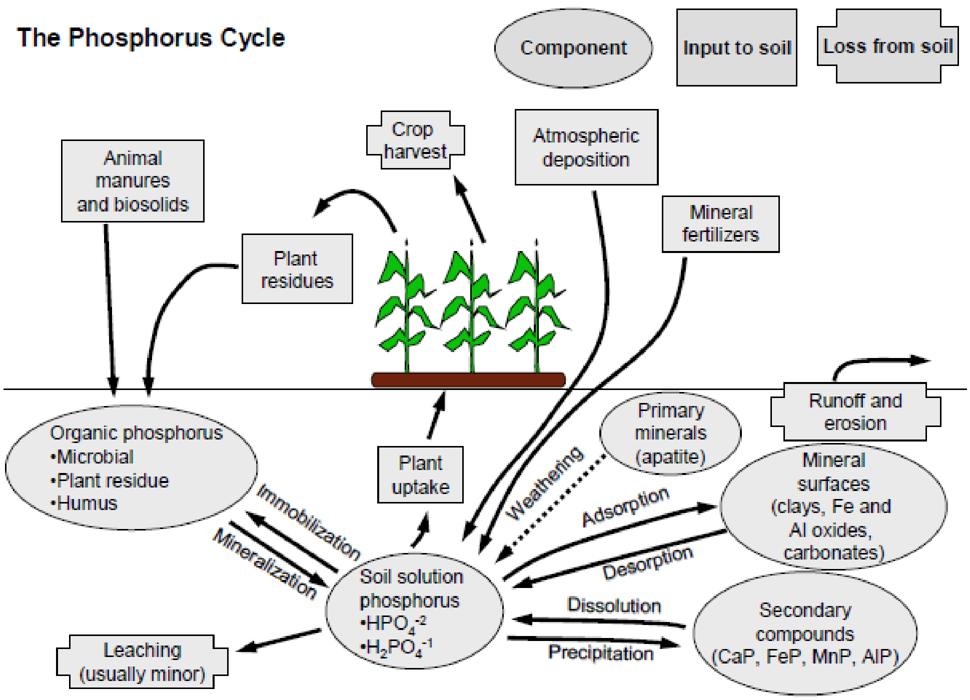

Figure 4. The phosphorus cycle

Only a relatively small fraction of soil P is available to plants and large amounts of P are continually needed to maintain a pool of readily available soil P to supply crop needs. On fertile soils most of the P (>90%) taken up by crops comes from the soil, and biosolids applications play an important role in maintaining the long-term soil fertility (Withers, 2011). However, a build-up of P in soils (e.g. as a result of regular applications of organic materials that supply greater amounts of P than are taken off by crops) may result in an increased risk of P losses to water via runoff, leaching and erosion (Figure 4).

Losses of P and sediment from land to surface water systems can conceptually be considered as a process comprising source, mobilisation, transport and delivery phases (Silgram et al., 2008). Losses are driven by the amount of flow through the landscape, and in particular where there is a risk of rapid runoff and good connectivity between the field and the watercourse. Farming practices influence the amount of P that is mobilised in flow, either by providing an increased P source (e.g. application of P) or by altering the mobilisation of P (timing of P input or provision of land cover) or by altering the rate of flow (e.g. tramlines running downslope).

P is lost from agricultural soils in both dissolved (DP) and particulate (PP, associated with soil particles) forms mainly through surface runoff and leaching. While subsurface pathways can be significant in P transfer to water, especially in soils with low P-retention properties and/or significant preferential flow pathways (e.g. cracking clay soils), it is reasonably well established that in most watersheds, P export occurs mainly in overland flow (e.g. Catt et al., 1998; Hart et al., 2004). In general terms, P concentrations in land runoff are dependent on both the quantities of P present in, or on, the soil, and the extent to which water moving through, or over, the soil captures these sources (Withers and Hodgkinson, 2009). Sources of P at the surface which may be mobilised in runoff include soil, crop residues, fertilisers and organic manure creating the potential for a wide range in the composition and concentration of any P present.

Effect of manure type

It is important to understand the forms of P present in manure/biosolids to predict potential P solubility, availability to plants and likelihood of loss to water. Total P content is a poor indicator of susceptibility to runoff and leaching. Recent studies have shown that the amounts of water soluble P (WSP) in different types of amendments are directly related to their release characteristics and potential for runoff after application (Kleinman et al., 2002; Withers et al., 2001). WSP has, therefore, been proposed as a valuable indicator of potential P loss from organic material amended soils.

Overall manufactured P fertilisers, such as triple super phosphate-TSP, typically have the greatest proportion of total P that is in a water soluble form, 80-90%. In comparison, both livestock manures (c.15-50% WSP depending on manure type) and biosolids (c.10% WSP depending on treatment strategy) have lower WSP contents than manufactured fertilisers. As a result, P losses tend to be greatest from manufactured fertiliser, less from manure and least from biosolids applied at similar P rates (e.g. Withers et al. 2001).

Biosolids differ markedly from other organic manures due to the treatment processes that they have undergone (e.g. enhanced anaerobic digestion, dosing with Ca, Fe or Al, thermal drying or composting) prior to application to agricultural land. Phosphate forms stable complexes with an extensive range of cations, and chemical precipitation with Fe, Al and Ca salts removes P from solution into the solid fraction; this can more than double the concentration of P in the biosolids. As a result, biosolids that have undergone enhanced P removal, by chemical precipitation with Ca, Fe or Al salts have higher contents of total P, Fe, Al and Ca.

P availability in biosolids is strongly influenced by the wastewater treatment processes (O'Connor et al., 2004; Elliott et al., 2005; White et al., 2010). Sludge treatment with high Al and/or Fe doses results in biosolids having low WSP concentrations, with Fe and Al phosphates as the dominant P forms (Shober and Sims, 2007). In addition, heat drying has been found to reduce P extractability by an average of 75% compared to dewatering processes (Smith et al., 2002).

The trend for lower WSP in enhanced treated products (limed and thermally dried) has been confirmed in many studies (Frossard et al., 1996; McCoy et al., 1986; Maguire et al., 2000a, b). For example, Penn and Sims (2002) demonstrated that biosolids produced using lime and/or metal (Al/Fe) salts had a lower WSP content, and caused lower increases in soil P and runoff when added to soils, compared with biosolids without added lime and/or metal salts. Overall, biosolid treatments that produce relatively dry biosolids, like heat drying, tend to reduce WSP (Brandt et al., 2004). Other work has found that thermal drying reduced P solubility in soil by as much as 70-80% compared with conventionally treated products (e.g. Frossard et al, 1996).

Overall P applied to soils in biosolids is less prone to runoff and subsurface drainage loss than the same amount of P applied in livestock manures and chemical fertilizers.

Effect of application method and timing

Timing of manure application to agricultural soils remains a contentious topic in nutrient management planning, particularly with regard to impacts on nutrient loss in runoff and downstream water quality (Liu et al., 2017). P loss via surface runoff is primarily controlled by the timing, rate, form and method of manure application as well as antecedent and post-application rainfall. However, the method/timing of manure and biosolids applications has traditionally been based on avoiding N losses to water or the atmosphere rather than controlling P. For almost all crops, manure/biosolids applications based on N requirements will supply more total P than is taken off in the crop.

Ideally, manure or biosolids are applied at times when nutrients can be best used by crops, in places where soils are not P saturated, and under conditions where offsite nutrient losses are minimized. Large P applications left on the surface of wet, frozen, compacted and intensively under drained soils in high rainfall areas are particularly vulnerable to P loss. In addition, short time intervals between manure application and rainfall in spring can lead to significant nutrient runoff losses (Smith et al., 2007; Vadas et al., 2007; Komiskey et al., 2011). Concentrations of P in runoff are often greatest during the first significant rainfall event following P application, but can remain high for several weeks, or even months after application (Smith et al., 2001; Withers and Bailey, 2003; Withers et al., 2003).

Effect of incorporation method and timing

The risk of P loss peaks immediately post application and subsequently decreases over time as the applied P increasingly interacts with the soil and is converted to more recalcitrant forms (Eghball et al., 2002). In areas where DP loss is a cause for concern, best practice suggests immediate incorporation of manure P (Kleinman et al., 2002). On arable fields, the risk of soluble P loss from biosolids is minimised by ploughing down before drilling; on grassland the risk of incidental P loss may need to be reduced by careful timing of application. Research has shown that incorporation of manure into the soil profile either by tillage or by subsurface placement decreases the potential for P loss in runoff (Figure 5).

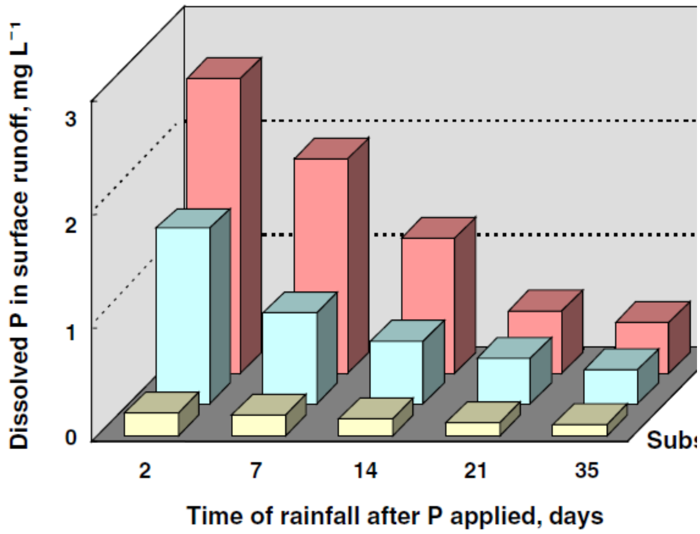

Figure 5. The effect of applying dairy manure (100 kg P/ha) by different methods on surface runoff from a grassy silt loam soil decreases with time (red: surface broadcast, blue: ploughed and yellow; sub-surface placement) (Source: Sharpley et al., 2006).

Effect of crop type

In general P runoff from permanently vegetated areas (e.g. grassland) is low compared to areas where annual crops are grown using conventional tillage. The P loss increases as the P is removed in particulate form with the eroding soil. Rainfall is often heaviest over autumn/winter and where this is combined with large areas of bare soil large amounts of P may be lost (e.g. maize).

Direct drilling may reduce soil erosion and losses of particulate P from erodible soil. However, the effect on dissolved P is more variable. Direct drilling will often lead to soil compaction, which can lead to P accumulation on the soil surface, and as a consequence, an increase in P loss in runoff (Xia et al., 2020). The overall impact of a given production system on P runoff to local surface waters will, therefore, be primarily dependent upon relative rates of sediment loss and the P levels in these eroding soil surfaces.

Soil factors

Soils have a finite capacity to bind P. When a soil becomes saturated with P, desorption of soluble P can be accelerated, with a consequent increase in dissolved inorganic P in runoff. Flynn and Withers (2007) found that soil P sorption capacity and binding energy have a large influence on the release of P into the soil solution (leachate) and to runoff water. Soils release P much more easily when they become P saturated (e.g. from overuse of P fertilisers) and the soil P buffering capacity is reduced. The P buffering capacity of soils (the ability of the soil to replenish dissolved P in solution as it removed) depends on the quantities and forms of Fe, Al and Ca present in the soil; P is more strongly bound in the order Fe>Al>Ca (Withers, 2011). The immobilization of P in soil by Ca, Fe or Al cations increases the P buffering capacity and reduces the availability of P to crops and to land runoff.

Normally, as soils receiving fertilizer and manure P become more P saturated, the binding energy falls. However, Maguire (2000) found that the application of higher quantities of biosolids amended with Fe and Al did not increase P in runoff because the degree of P saturation in soil was not increased. This was because the biosolids increased the P sorption capacity and hence the soil's buffering capacity (Lu and O'Connor, 2001). Field studies by White et al. (2010) have shown that runoff P for soils amended with Fe-treated biosolids was not significantly different from that for the untreated control soil.

Withers et al. (2001) measured runoff P from field plots that had previously received P from different sources, and concluded that there was a lower risk of P runoff following application of biosolids compared with other agricultural P amendments at similar P application rates. However, the risk of P loss from biosolid-amended soil was dependant not only on the type of biosolid applied but also on the nature and degree of P saturation of the soil.

Runoff P concentrations will greatly increase once P saturation exceeds a threshold of 20-30% (e.g. Kleinman et al., 2000; Nair et al., 2004). P saturation threshold broadly equates to Olsen soil P indices of 3, 4 and 5 for sand, loam and clay soils, respectively. Where soils have little capacity to accept any more P because they are already close to soil P saturation then biosolids with a high Fe content may still be applied without further increasing SRP release (Withers, 2011). Although such a strategy can only be adopted when the risk of erosion is low and not in areas with high rainfall and on land that leaves bare soil at critical times (e.g. late sown winter cereals, potatoes, maize etc.).

Climatic conditions

Applying biosolids to dry soils and incorporating soon after application will limit the impact of heavy rainfall events after application on P losses to water. In arable production systems autumn applications are likely to pose the lowest risk of P losses to water. Also, increasing the length of time between biosolids application and a rainfall or runoff event (i.e. using weather forecasting to minimise the risk of rainfall occurring application) will reduce the risk of P transport in runoff.

Summary

Biosolids applications pose a low risk of P loss to water as they have a low water soluble P concentrations (typically less than 10% of total P)

Autumn biosolids applications which are incorporated into the soil present a low risk of P loss via surface runoff. In contrast applications that are made to soils with low moisture deficits in the spring which are left on the soil surface pose a greater risk of loss via surface runoff.

The highest risk of P loss occurs on soils which are saturated with P. Managing biosolids applications on soils at P index <5 by regular soil sampling and avoiding annual applications of biosolids where soil P is not limiting crop growth will reduce the risks of excessive soil P levels.

3.3 Ammonia emissions

Introduction

Ammonia (NH_3) emissions to air contribute to acid deposition and can cause eutrophication of sensitive ecosystems, with re-deposition of emitted NH_3 also contributing to indirect nitrous oxide emissions from soils. In addition, NH_3 reacts with acids in the atmosphere forming particulate matter which may pose a threat to human health (Webb et al. 2004).

There is an extensive body of research in the UK (and elsewhere) on NH_3 emissions following land application of livestock manures and slurries [see for example Sommer et al. (1997), Huijsmans et al. (2001), Misselbrook et al. (2002) and Webb et al. (2004)]. In the UK, this information has been used to populate the National Ammonia Emissions Inventory (Misselbrook et al. 2015) and provide guidance for farmers on minimising NH_3 emissions from manures (Defra 2009).

The research has shown that the amount and rate of NH_3 release following land spreading depends on the organic material properties, as well as a range of spreading, soil and environmental factors (Nicholson et al. 2013).

Effect of manure type

In developing the MANNER-*NPK* decision support tool, Chambers et al, 1999 and Nicholson et al., 2013 collated data from UK experiments where NH_3 emissions had been measured following livestock manure applications to agricultural land. Using this data, four 'standard' NH_3 loss curves were derived (by fitting a MIchaelis-Menton equation) and showed that the pattern of NH_3 losses over time was different for different manure types (cattle, slurry, pig slurry, cattle/pig/duck FYM and poultry manure). Losses from slurry and FYM were found to be more rapid than from poultry manures; for FYM half the NH_3 is lost within *c.*15 hours of spreading (Table 6).

Because the RAN content of FYM is lower than slurry and poultry manure (Table 3) the total amount of N lost via NH_3 from an equivalent total N application rate is also less. The concept of NH_3 Emission Factors (EFs) can be used to illustrate this, where the total NH_3-N lost is expressed as a percentage of the total N (TN) applied. For example, Thorman et al. (2020) measured NH_3 emissions from a range of manure types applied to 3 grassland and 3 arable sites, at 2 application timings (autumn and spring). Lower NH_3 losses were measured from FYM applied in both autumn and spring (EF 1.1 -2.8 % TN applied) than from slurry (EF 20.7 – 24.9 % TN applied) and poultry manure (EF 5.7 -10.4 %TN applied), reflecting the much lower readily available N content of the FYM. Nicholson et al (2017) also reported much lower NH_3 EFs for FYM and compost (low RAN manures) than for food-based digestate and slurry (high RAN manures), across 3 sites and 2 application timings (Table 7).

Table 6. Michaelis-Menton equation parameters for the 4 'standard' NH_3 loss curves (Nicholson et al., 2013)

Manure type	N_{max} (% RAN applied)	K_m (hours)
Cattle slurry	32.4	7.5
Pig slurry	25.5	11.6
FYM (cattle, pig and duck)	68.3	14.9
Poultry manure	52.3	40.4

Note: N_{max} is the maximum potential NH_3 loss as the time after application approaches infinity; K_m is the time (in hours) when the N lost is half of N_{max}.

Table 7. Mean cross-site NH_3 emission factors (Nicholson et al., 2017)

Manure type/treatment	Emission factor (% total N applied)
Food based digestate – surface broadcast	42
Food based digestate – bandspread	38
Livestock slurry – surface broadcast	31
Livestock slurry – bandspread	24
Livestock FYM	4.5
Compost	3.3

Note: mean EFs for the 3 sites and 2 application timings.

At the time that MANNER-*NPK* was developed there was no UK field experimental data on NH_3 losses from biosolids applications to land, so these were assumed to follow the same pattern as poultry manure (because neither biosolids nor poultry manures are straw based solid manures). We are not aware of any subsequently published UK data, and very little information is available from elsewhere in the world. The little information we were able to locate is summarised below.

Early laboratory-based work by Donovan and Logan (1983) in the USA found large differences in the amount of NH_3 lost from different sludge types, with the greatest losses recorded from a lime-stabilised sludge with a pH of 12 (Figure 6) than from the anaerobic and aerobic sludges and the compost. Figure 6 also shows that, for the sludge types tested, the majority of NH_3 losses occurred within 12 hours after application, similar to FYM (Table 6). A later laboratory incubation study, also from the USA, found that more NH_3 was volatilized from biosolids than from compost, which was probably due to the higher total N concentration and lower C/N ratio of the biosolids (He et al, 2003). In contrast, Pu et al. (2010) examined the effects of biosolids type, soil type and polymer addition on NH_3 volatilization at a soil incubation temperature of 30 °C, but found that NH_3 losses over 72 days were minimal, accounting for <4% of the applied RAN.

A field study by Beauchamp et al. (1978) found that 56-60% of the RAN applied in anaerobically digested sewage sludge was volatilized after 5-7-days following application at rates of 116 and 134 t/ha in May and October, respectively.

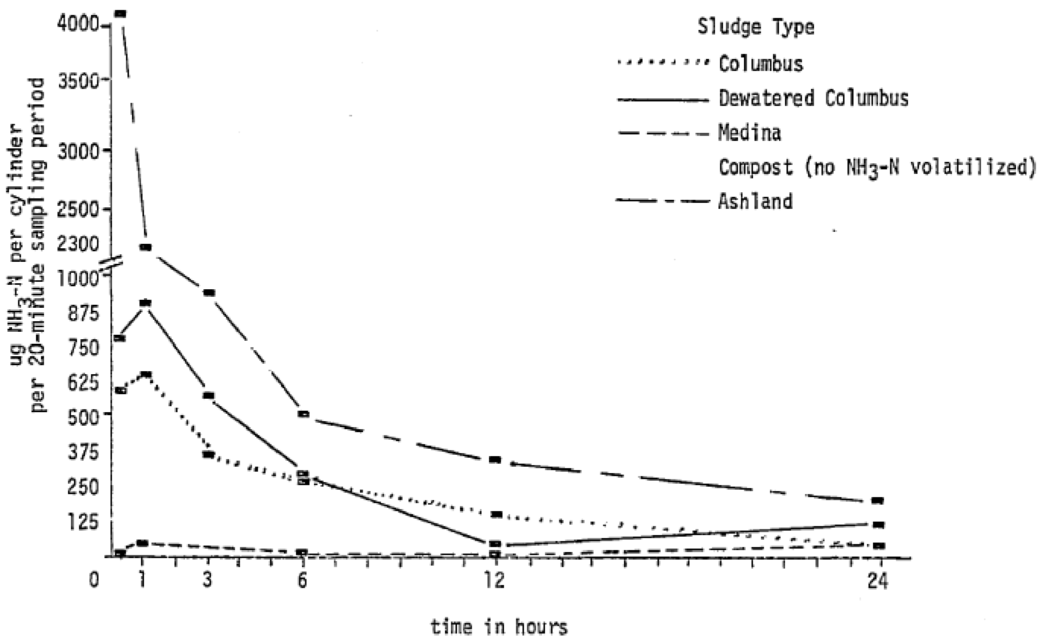

Figure 6. NH_3 volatilisation from different sludge types over a 24 hour sampling period (Donovan, 1982).

One of the few field studies we found was from Australia, where Robinson and Polglase (2000) applied dewatered biosolids from three sewage treatment plants and calculated volatilisation using a mass balance approach. They found that most N loss occurred within 1 week of application, with 71-81% of the RAN applied lost after 3 weeks.

Effect of soil incorporation

Soil incorporation method and timing have been demonstrated to strongly influence NH_3 losses from solid manures. The UK data used to derive the MANNER-*NPK* algorithms (Nicholson et al. 2013) indicated that the technique used for manure incorporation (plough, rotavator, disc or tine) will affect NH_3 losses from manures as shown in Table 8. Ploughing is the most effective technique, reducing the N_{max} (i.e. the maximum potential amount of RAN lost via NH_3 volatilisation) to 10% of that of surface broadcast manure for FYM and 5% for poultry manure.

Table 8. Effect of soil incorporation technique on N_{max}

	Adjustment to N_{max}		
Incorporation technique	**FYM**	**Poultry**	**Slurry**
Plough	0.1	0.05	0.1
Rotavator	0.2	0.1	0.115
Disc	0.3	0.2	0.2
Tine	0.7	0.3	0.3

Note: N_{max} is the maximum potential NH_3 loss as the time after application approaches infinity

For all manure types, the more rapidly the manure is incorporated into the soil following application, the lower will be the amount of NH_3 lost. This was nicely illustrated by Thorman et al (2020) who reported that poultry manure applied in autumn to arable soils and incorporated within 24 hours had similar NH_3 losses to those from FYM which remained on the soil surface. In spring, the poultry manure was not incorporated after application and in this case there were significantly higher NH_3 losses from the poultry manure than from the FYM.

The efficacy of rapid soil incorporation has also been reported in studies with biosolids. Early laboratory studies (Donovan and Logan, 1983) showed that NH_3 volatilisation from different periods of incorporation increased linearly with time, and concluded that sewage sludge should be incorporated as soon as possible to minimize NH_3 loss. Similar laboratory incubation studies undertaken by He et al (2003) in the USA found that soil incorporation of biosolids reduced NH_3 volatilization losses by 5-fold compared with surface application. The only evidence from field-based studies that we could locate was from Sweden, where Willen et al (2016) applied mesophilically digested and dewatered sewage sludge to arable land in spring, and measured NH_3 emission for 24 hours following application. There was no statistically significant difference between NH_3 emissions from treatments where the sludge was immediately incorporated compared to those where incorporation was delayed by 4 hours. However, there was a tendency for lower NH_3 emissions from the immediate incorporation treatment. It was shown that 55-65% of NH_3 losses occurred in the first 3 hours after application for both treatments i.e. before incorporation in the delayed incorporation treatment (Willen, 2016), emphasising the need for very rapid incorporation to minimise NH_3 losses.

Other factors affecting ammonia losses

As well as soil incorporation technique and timing, various other factors have been shown to influence NH_3 volatilization from field applied manures (i.e. soil moisture content, land use, wind speed, rainfall after spreading, slurry dry-matter content, slurry application technique). These factors have been shown to mainly effect NH_3 losses from slurry (and other liquid manures), and Nicholson et al (2013) did not make adjustments to the MANNER-*NPK* algorithms for solid manures (FYM, poultry) in the same way that was done for slurries.

There is some early research which investigated various soil and environmental factors that might effect NH_3 losses following biosolids spreading to land. In laboratory experiments Ryan and Keeney

(1975) found that the amount of NH_3 volatilised was significantly affected by soil clay and organic matter contents and the interaction between these variables, with 11 to 60% of the RAN applied lost. Terry et al (1978) used synthetic wastewater sludges in laboratory incubation experiments and found that NH_3 volatilization was 61% greater from soils of pH 7.5 than from soils of pH 6.0 or 5.3. Ammonia volatilization was enhanced by rapid drying of the soil which inhibited nitrification. There was no effect of sludge application rate (11.2 to 44.8 t/ha), but volatilization was reduced by a single large application, rather than smaller multiple applications. In a series of laboratory experiments, Donovan and Logan (1983) studied the effect of a single variable on NH_3 emissions from biosolids. They concluded that:

- Initial soil moisture contents ≤1.5 MPa tension increased NH_3 volatilization compared with air-dry soil.
- NH_3 volatilisation was significantly greater from the soil at pH 7 5 compared with pH 6.7 and pH 5.1.
- NH_3 volatilisation increased with increased temperature (12.8, 18.3 and 26.7°C).
- When the sludge contained large sludge particles, NH_3 loss increased with vegetative cover. However, cover had no effect when the sludge was well homogenized.

There is little evidence from field studies to support these findings. However, Beauchamp et al. (1978) applied anaerobically digested sewage sludge in the field at rates of 116 and 134 t/ha in May and October, respectively, and measured NH_3 losses using the horizontal flux method. Fluxes generally followed a diurnal pattern with maximums occurring at about midday. Flux generally decreased with time in an exponential manner, so the 'half life' of the RAN applied in the sludge was 3.6 and 5.0 days for May and October, respectively. Air temperature appeared to be most closely related to flux rate especially in the 2-3 days following application.

Summary

- Ammonia emissions are highest following surface broadcast applications which are not incorporated.
- Evidence from livestock FYM is that NH_3 loss is quick (80% within 24 hours) and that rapid incorporation will reduce losses. Few measurements from field applied biosolids, therefore still best to use FYM data.
- Management strategies that prevent the rapid soil incorporation of biosolids (e.g. moving applications on winter cereals from autumn stubbles to topdressing on growing crops in spring) will increase ammonia losses.

3.4 Nitrous oxide emissions

Introduction

Nitrous oxide (N_2O) is a greenhouse gas with a global warming potential *c.*300-fold greater than carbon dioxide (IPCC, 2007). The current UK greenhouse gas (GHG) inventory estimates that for 2017, 70% of N_2O was produced from agricultural sources (Brown et al., 2019). The majority of this (65%) was directly emitted from agricultural soils and includes emissions following the application of livestock manure, biosolids, compost and manufactured N fertiliser (Brown et al., 2019), although these losses are generally small in agronomic terms.

There are many factors which may affect N_2O emission from organic manures following application to land including application timing and method of incorporation, manure composition, soil type, temperature and rainfall (Chadwick et al., 2011). Carbon (C) added to soil through manure application can stimulate denitrification and also speed up soil respiration, depleting oxygen in soil pores, and triggering denitrification and release of N_2O (e.g. Lazcano et al., 2016). Pre-treating manure (or biosolids) by solid separation or anaerobic digestion reduces the amount of degradable C applied in a single application to the soil and hence tends to decrease N_2O emissions relative to untreated materials (Montes et al., 2013). High N_2O emissions (via denitrification) are favoured in wet (anaerobic) conditions (Firestone and Davidson, 1989), so organic manure application to very wet soils or before heavy rainfall should be avoided. Maintaining soil at pH 6.5 and above has also been shown to help reduce N_2O emissions (Mkhabele et al., 2006).

Effect of manure type

The Intergovernmental Panel on Climate Change (IPCC) Tier 1 methodology sets a single default N_2O EF for organic manures (including biosolids) of 1% of total N applied (IPCC, 2006), although following a refinement to the IPCC method this has recently been disaggregated to 0.6% and 0.5% of total N applied in wet and dry climates respectively (IPCC, 2019). It is notable that a recent global meta-analysis by Zhou et al. (2017) reported a much higher overall mean N_2O EF for manures of 1.8% (n = 146).

There is little information to verify if the Tier 1 IPCC default values are valid for UK manures and climatic conditions, or whether it would be possible to assign different EFs for different manure types. Recently, Thorman et al (2020) published results from a comprehensive set of field experiments designed to address this issue. Direct N_2O EFs were calculated from measurements of emissions from livestock manures (pig slurry, cattle slurry, cattle FYM, pig FYM, poultry layer manure and broiler litter) applied in autumn and spring at 3 arable and 3 grassland sites in the UK. EFs ranged from -0.05 to 2.30% of total N applied, with the variability driven by a range of factors including differences in manure composition, application method, incorporation and climatic conditions. When data from the autumn applications were pooled, the mean N_2O EF for poultry manure (1.5%) was found to be greater than for FYM (0.4%) and slurry (0.7%). For the spring applications, there were no significant differences in the mean N_2O EFs for poultry manure (0.5%), slurry (0.4%) and FYM (0.2%). The low emissions from FYM following both the autumn and spring application timings were probably because a much lower proportion of the total N applied was in a RAN form and hence available for nitrification. Cayuela et al. (2017) also attributed the higher N_2O EF from organic liquid fertilisers (0.8%) compared with organic solid fertilisers (0.2%) to differences in the RAN content of the manures. Likewise, in a review of N_2O emissions from agricultural soils in Eastern Canada the mean N_2O EF from liquid manure (1.7%) was found to be considerably higher than from solid manure (0.3%) (Gregorich et al., 2005).

Other studies have also explored differences in EFs between different manure types. Loro et al. (1997) ascribed greater N_2O fluxes from cattle FYM than from slurry to the greater quantity and longer duration of availability of the C applied with FYM, whereas Rochette et al. (2008) found no clear treatment differences following application of liquid and solid manures. A recent global meta-analysis (Charles et al., 2017) identified three groups of organic materials with similar N_2O EFs: the high-risk group included animal slurries, waste waters and biosolids (mean EF 1.2%); the medium-risk group included solid manure, composts + fertilisers, and crop residues + fertilisers (mean EF 0.35%); and the low-risk group included composts, crop residues, paper mill sludge and pellets (mean EF 0.02%). The EF depended on the composition of the material (C/N ratio), soil properties (texture, drainage, organic C and N) and climatic (precipitation) factors.

Some laboratory work has been undertaken to measure N_2O emissions following biosolids application to soil. For example, Pu et al. (2010) examined the effects of biosolids type, soil type and polymer addition on the mechanisms and extent of denitrification at a soil incubation temperature of 30 °C. They reported that 24% of TN applied in MAD biosolids and 29% for AeD biosolids was lost through denitrification over 105 days, indicating that denitrification is potentially a major pathway of gaseous N losses, particularly under warm and moist soil conditions. A UK laboratory incubation study (Rigby and Smith, 2013) examined the effects of different digestate types (including dewatered, anaerobically digested biosolids) and soil types on N availability in soils. They found that N release in digestate-amended soil depended on the digestate type, and that overall N release is modulated by digestate mineral and mineralisable N contents. However, Thorman et al (2009) pointed out that whilst this type of laboratory experiment can help with understanding the processes influencing N_2O emissions, the results obtained are not directly transferable to field situations.

In a review of the literature, Thorman et al (2009) identified a limited number of published field studies measuring N_2O emissions following biosolids application to agricultural land in the UK:

- At a grassland site in the east of Scotland, Ball et al. (2004) reported N_2O losses following the application of injected liquid digested sludge (25-215 kg N ha^{-1} $application^{-1}$), surface spread composted sludge (460-615 kg N ha^{-1} $application^{-1}$), surface spread thermally dried pellets (510 kg N ha^{-1} $application^{-1}$) and injected cattle slurry (200-430 kg N ha^{-1} $application^{-1}$), with 2 applications each year (in April and June). Nitrous oxide EFs ranged from 0.2% (pellets) to 4.3% (cattle slurry) after the April application timings, and from 0.3% (pellets) to 5.5% (cattle slurry) after the June application timings. Losses of N_2O from all 3 biosolids products were similar. Residual effects from the composted sludge (measured 2 years after application) increased grass yields, but did not increase N_2O emissions.
- At the same Scottish site, Jones et al. (2007) measured emissions for 2 years, with 2 applications per year in April and June. Surface applications of thermally dried pellets (1535 kg N ha^{-1} $application^{-1}$), broiler litter (1240 kg N ha^{-1} $application^{-1}$) and cattle slurry (150-380 kg N ha^{-1} $application^{-1}$) led to EFs in the range of 1.3-4.3%, 0.5-2.6% and 0.2-0.5%, respectively. The relatively high EFs from the thermally dried pellets were probably due to the high N application rates, which increased soil mineral N and associated N_2O production.
- In the west of Scotland, Scott et al. (2000) measured EFs of up to 1% following high rates (1000-1500 kg N ha^{-1}) of application of digested sludge cake to grassland, which was followed by soil incorporation and reseeding.
- Defra project CC0256 (Defra, 2001) measured N_2O losses at 2 arable sites (in Hampshire and the Midlands) following spring applications (March-May) of two liquid digested sludge products, cattle slurry and pig slurry. All had similar mean EFs (0.35%, 0.40%, 0.43% and 0.93%, respectively). The authors concluded that there was insufficient evidence to support separate EFs for liquid digested sludge compared with cattle/pig slurry.

We are not aware of any subsequent UK work where N_2O emissions from biosolids have been measured in the field. However, the wider international literature contains several examples where greater N_2O emissions have been measured from biosolids-amended soil in agricultural systems (e.g. Sharma et al. 2017, Liu et al. 2014, Thangarajan et al. 2013). In terms of differences between biosolids types, thermal drying has been found to reduce the mineralization of organic N in anaerobic digested biosolids and thus decrease N_2O emissions (Case et al. 2016). A Canadian field study (Kamal, 2019) also compared N_2O emissions from different biosolids types (alkaline treated, mesophilic anaerobic digested - MAD, and composted). There was a significant ($p<0.05$) effect of biosolids type on N_2O emissions during the growing season, with cumulative emissions from the mesophilic anaerobic digested biosolids higher than those from the alkaline treated and composted biosolids and the control. Another Canadian study (Singh, 2020) found that the C/N ratio of the biosolids was the most important driver of differences in N_2O emissions as a result of differences in microbial respiration and

N mineralization. As a result mesophilic anaerobically digested biosolids with the lowest C/N ratio had higher cumulative N_2O emissions than alkaline treated and composted biosolids.

Thorman et al (2009) cautioned that although the results of non-UK studies are useful for comparison, they should not be used to draw specific conclusions about the UK situation because of differences in biosolids treatment processes and composition, environmental conditions etc.

Effect of soil incorporation

Incorporation would be expected to reduce N losses from NH_3 volatilisation (see Section 3.2), hence conserving N in the soil for subsequent loss as N_2O or nitrate leaching (an example of so-called 'pollution swapping'). Following incorporation, reduced soil oxygen concentrations from buried manure decomposition may result in the formation of anaerobic micro-sites within the soil matrix suitable for denitrification and subsequent N_2O generation (Webb et al., 2014). Thus, greater N_2O emissions may be expected following incorporation in autumn in comparison with manure left on the soil surface in spring.

The effects of solid livestock manure incorporation on N_2O losses have been explored in UK field studies (Thorman et al., 2006; Thorman et al., 2007a) which showed that whilst incorporation sometimes increased N_2O emissions (particularly on light textured soils) this was not always the case, and that the effect may be related to an interaction between soil texture and weather conditions.

In Canada, Kamal (2019) evaluated the effects of different biosolids types (alkaline treated, mesophilic anaerobic digested - MAD, and composted) on N_2O emissions, and reported a tendency for higher N_2O emissions where biosolids were incorporated compared with surface application. Similarly, a Swedish study (Willen et al., 2016) found that delayed incorporation (after 4 hours) tended to reduce N_2O emissions compared with immediate incorporation. In contrast another Canadian study (Singh, 2020) found no difference in cumulative N_2O emissions between surface applied and incorporated biosolids.

Effect of application timing

Numerous studies have shown that N_2O production increases with temperature and can be stimulated with a rise in soil moisture (e.g. Lazcano et al., 2016). In the UK, Thorman et al (2020) reported a tendency for higher manure N_2O EFs in the autumn than in the spring which was in close agreement with a previous UK study where Thorman et al. (2007b) showed that direct N_2O losses were greater ($P<0.05$) from slurry applications in autumn/winter (EF 1.1%) than from those in spring (EF 0.5%). This probably reflects differences in soil moisture/temperature conditions, but also the lower levels of crop N uptake in the autumn/winter compared with spring.

Grassland studies in Ireland (Bourdin et al., 2014; Cahalan et al., 2015) have also shown that the influence which the season and timing of cattle slurry applications have on N_2O emissions is driven by soil and climatic conditions (moisture/rainfall and temperature). A Canadian field study (Kamal, 2019) reported a seasonal effect where N_2O flux was highest from 6 July to 16 August when the soil moisture content was low but the air temperature was high. In Sweden, Willen et al (2016) found that N_2O were generally lower after spring than after autumn application, because of drier soil and greater crop N uptake in spring

Summary

The magnitude of N_2O losses from organic manures has been shown to depend on the proportion of the total N applied in a RAN form; the lower the RAN content the less N is available for nitrification and subsequent loss as N_2O. Studies with biosolids have found that their C/N ratio was the most important factor driving N_2O emissions, so that biosolids products with lower C/N ratios have higher cumulative N_2O emissions. Thermal drying has also been found to reduce the mineralization of organic N in anaerobic digested biosolids and thus decrease N_2O emissions.

For both FYM and biosolids, there is evidence that soil incorporation following land application can increase N_2O losses relative to surface broadcast, although this is not always the case. Where seasonal differences in N_2O emissions have been reported, these are thought to be a result of complex interactions between temperature, rainfall (soil moisture), manure composition or manure management and other soil and environmental factors, as well as crop N uptake.

Overall, research suggests that there are no consistent effects of application timing or soil incorporation on N_2O emissions from manure applications. As biosolids cake has a low RAN N content land application poses a low risk of N_2O emission compared with manufactured fertiliser and high RAN manures (i.e. slurries and digestate)

3.5 Methane emissions

Currently, the standard IPCC methodology (IPCC, 2006) does not include a specific EF for direct methane (CH_4) emissions from soils, recognising that in most circumstances (i.e. under aerobic soil conditions) emissions are likely to be low, although within the manure management section of the IPCC methodology it is acknowledged that there is a small emission. Indeed, well-drained aerated soils can act as a sink for CH_4 (Yamulki et al., 1999).

Various studies have shown that CH_4 emissions increase following the application of organic materials to agricultural soils. Laboratory experiments carried out by Chadwick & Pain (1997) showed that CH_4 can be emitted immediately following the surface application of dairy or pig slurry to grassland, but emissions decreased to background levels after 48 hours. Such additions may lead to anaerobic soil conditions that result in CH_4 production by increasing soil moisture and through the addition of an instant supply of utilisable carbon. However, Chadwick & Pain (1997) reported that the majority of emitted CH_4 was derived from the slurry itself and not from the soil. The brevity of CH_4 emissions following the application of farm slurries to grassland was further illustrated in a field experiment, where 90% of total emissions occurred during the first 24 hours (Chadwick et al., 2000). Small emissions of CH_4 were also measured following solid manure applications (i.e. layer manure and beef FYM), with the authors again suggesting that these were derived from the solid manures rather than from the soil.

In a study investigating pollutant losses following spreading of livestock manures, compost and food-based digestate at 3 UK agricultural sites, WRAP (2016) also reported that the majority of the CH_4 emissions occurred immediately after spreading of the organic materials. Cumulative CH_4 emissions from the solid materials (FYM and compost) were lower than from the liquids (digestate and slurry). Methane emissions from slurry were higher than from the food-based digestate, probably because most of the 'available' carbon in the digestates had already been lost during the anaerobic digestion process. For both the liquid organic materials, CH_4 emissions from the bandspread material were consistently greater than from the broadcast applications. This may be because bandspreading a liquid organic material creates anaerobic conditions in the band which are more conducive to CH_4 production.

Because biosolids products applied to land have been anaerobically digested, there will usually be little carbon remaining in a form which is readily lost as CH_4. Nevertheless, there have been a small number of UK research studies where CH_4 losses following biosolids application to land have been measured. Ball et al. (1994) reported that CH_4 emissions generally only lasted for 2-3 days after the injection of liquid digested sludge or cattle slurry into grassland. Emissions from composted sewage sludge and thermally dried sludge pellets were negligible. In later years at the same site, Jones et al. (2005) reported that CH_4 emissions from thermally dried sludge pellets and poultry manure treatments did not differ from the untreated control.

In a review for UKWIR, Andrews et al. (2009) concluded that the available UK field evidence supported the IPCC view that CH_4 emissions following the application of organic materials to land are minimal, a view which is reinforced by a recent study in Sweden where Willen et al (2016) found that CH_4 emissions from digested and dewatered sewage sludge applications were negative or negligible.

4. Impact of application timings on the balance of N losses to air and water

4.1 MANNER-*NPK* assessment of contrasting biosolids application timings

MANNER-*NPK* (Nicholson et al, 2013) was used to estimate the impact of preventing biosolids applications in the autumn to most arable crops. All the MANNER-*NPK* scenarios were based on the following assumptions:

- The applications were all of digested sludge cake applied at a rate of 240 kg TN/ha (35 kg RAN/ha)
- Autumn (pre-10th October) applications were soil incorporated by plough within 12-24 hours of spreading. Applications after this date were not soil incorporated.
- Rainfall between application and end of drainage was 550 mm (low rainfall), 690 mm (medium rainfall) and 950 mm (high rainfall).
- The light soil was loamy sand over loamy sand, and the heavy soil was clay loam over clay loam.
- The crop type was a late sown winter cereal.

When interpreting the MANNER-*NPK* outputs, it is important to be aware that because of the lack of UK field data on NH_3 emissions from biosolids applied to land (see Section 3.3), the losses calculated by MANNER-*NPK* are based on NH_3 volatilisation patterns derived using data for poultry manures. Similarly, UK experimental data on organic N mineralization from biosolids were not available when MANNER-*NPK* was developed; it was therefore assumed that N mineralization from biosolids follows the 'high rate' mineralization release curve described in MANNER-*NPK* (see also Bhogal et al., 2015).

Nitrate leaching

Soil type and rainfall have a strong influence on diffuse water pollution and nitrate leaching losses are likely to be greater from sandy soils than from medium/heavy soils because of differences in water holding capacity. Nitrate leaching losses will be greater in high rainfall areas than low rainfall areas, reflecting the higher drainage volumes that wash nitrate out of the soil and beyond plant rooting depth.

The MANNER-*NPK* outputs illustrate how changing the timing of biosolids applications can affect the amount of N lost by nitrate leaching on light and heavy soils and are summarised in Figure 7. For example, biosolids applied on the 12th September to a light sandy soil in arable production in a low rainfall area was predicted to lose c.14% of total N by nitrate leaching losses. However, if the same application was from mid-November onwards, nitrate leaching losses were predicted to be less than <1% of total N applied (Figure 7).

The effect of soil type on nitrate leaching losses can also be clearly seen by comparing Figures 7a and 7b. For example, a biosolids application on 12th September to the heavy soil in a low rainfall area was predicted to result in nitrate leaching losses of 4% of total N applied (Figure 7b), compared with 14% from the same application timing to the light soil (Figure 7a). The lower losses from the heavy soil reflect the greater water holding capacity, compared to light soils which requires greater drainage volume to leach nitrate beyond rooting depth.

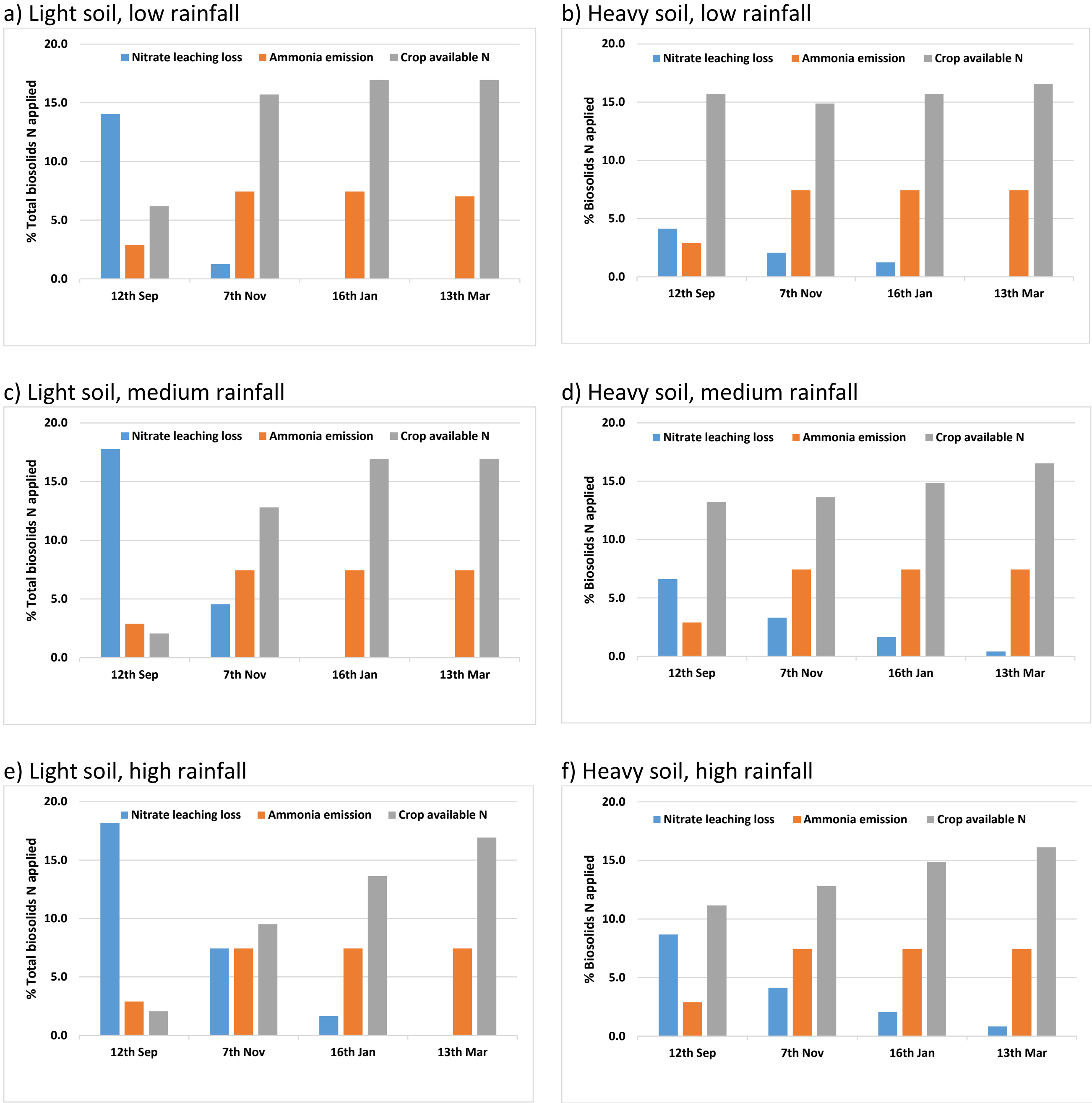

Figure 7. Nitrogen losses and crop available N from biosolids applied at different times of year to light or heavy soil in a low, medium and high rainfall areas.

Although not specifically included in the scenarios tested, nitrate leaching losses following applications to grassland will be lower than those from arable soils. This is because grass N uptake in the period after biosolids application is greater than that by a winter cereal, and this will reduce the amount of biosolids N in the soil which is at risk of subsequent nitrate leaching (see Section 3.2).

Overall, the MANNER-*NPK* outputs confirm findings from the literature (and outlined in Section 3.2) that the greatest risk of nitrate leaching losses from biosolids spreading is following early autumn applications to sandy soils in arable production.

Ammonia emissions and crop N recovery

The MANNER-*NPK* estimates indicated that NH_3 volatilisation losses from biosolids applied in September and incorporated (by ploughing) into the soil within 12-24 hours of spreading were lower than where the biosolids were left on the soil surface. However, this meant that more N was conserved in the soil and subsequently lost over winter via nitrate leaching (Figure 7). On the light soils in a low rainfall area this resulted in greater overall N losses from the September application timing (17% of total N applied) and less N available for crop uptake (6% of total N applied), Figure 7.

On the heavy soil in all rainfall areas, even though more N was leached from the September application, the overall N loss was similar to the other application timings (7-10% of total N applied) and the N available for crop offtake was around 15% of total N applied for all application timings. This suggests that on these soil types, which comprise the majority of agricultural soils in the UK, management strategies to reduce ammonia emissions are likely to have the biggest impact on reducing overall N losses to the environment and help maximise crop available N supply from biosolids applications.

It is interesting to compare these findings with a study in Sweden where Willen et al (2017) undertook a comprehensive environmental assessment of biosolids storage and land application. These authors found that systems with autumn application are preferable to systems with spring application for all impact categories if nitrate leaching is excluded.

5. Practical and other implications

Moving applications from autumn to spring poses several practical and logistical challenges to the management of biosolids and other solid organic materials.

5.1 Soil conditions

An important practical consideration in the management of biosolids, as with all low RAN organic manures, is ensuring that applications are made to soils when they are strong enough to withstand the weight of the spreading equipment. Travelling on 'wet' soils with spreading equipment which weighs well in excess of 20 tonnes is likely to cause significant compaction, especially on clay and medium textured soils. Soil compaction is recognised as a threat to soil quality and increases the risk of runoff and erosion even if it is confined to tramlines (Silgram et al., 2010).

Observations from the Defra funded Cracking Clays project (WQ0118) suggested that when slurry was applied to soils with a soil moisture deficit of less than 10 mm, significant soil compaction occurred. Soil moisture content will vary according to texture as well as rainfall and evapotranspiration. Generally, soils are close to field capacity (i.e. all the soil pores capable of holding water under gravity are full) during the winter months and dry out during spring and summer as evapotranspiration usually exceeds rainfall, leading to a soil moisture deficit. In arable rotations, soils are typically driest in late summer and start to wet up in the autumn following harvest. On an annual basis the return to field capacity and the date when soil moisture deficits form will depend on weather conditions, with soils in higher rainfall areas staying wetter for longer in spring

An assessment was carried using the IRRIGUIDE water balance model (Bailey and Spackman, 1996) to identify when soil moisture deficits would typically be sufficient (i.e. soils would be dry enough) to allow biosolids applications to pose a low risk of soil compaction. The model was used to estimate daily soil moisture deficits for two soil types (sandy loam and clay loam), under winter cereal cropping for 9 locations chosen to be representative of the area covered by each Water Company in England and Wales. The model was run using 10 year (2008-2017) average climate data for each site. The model uses information on volumetric moisture content, crop cover, rooting depth and weather data to estimate evapotranspiration and soil drainage.

The model runs indicate that soils in high and moderate rainfall areas typically return to field capacity between late September and early November (Figures 8 and 9), with soils in low rainfall areas not returning to field capacity until late December and January (Figure 10). Soil moisture deficits of greater than 10 mm were predicted in low and moderate rainfall areas between the middle of and end of March, and early April in high rainfall areas on both soil types.

This suggests that in most seasons it would not be possible to spread biosolids without causing a risk of soil compaction until late March. This is likely to delay biosolids applications to winter cereal crops until stem extension which typically begins in early April. The potential physical damage to the plants caused by the biosolids application along with potential crop contamination issues is likely to make topdressing to cereal crops in spring impractical (i.e. farmers will not allow it due to crop damage and the effect this will have on yields, as well as the negative effects on soil structure). For spring planted crops, if applications could not be made until late March this would require farmers to delay planting, which would reduce crop yields, which would also be impractical.

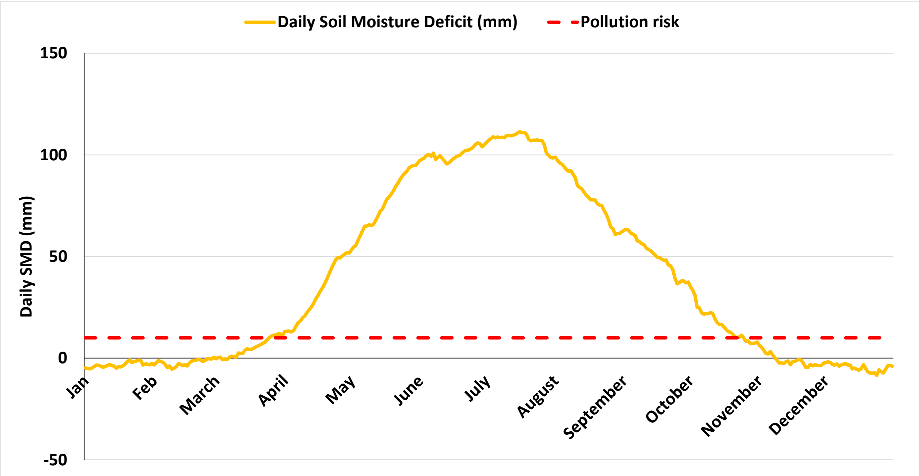

Figure 8 Predicted soil moisture deficits on clay soil under winter wheat in high rainfall areas (>700mm)

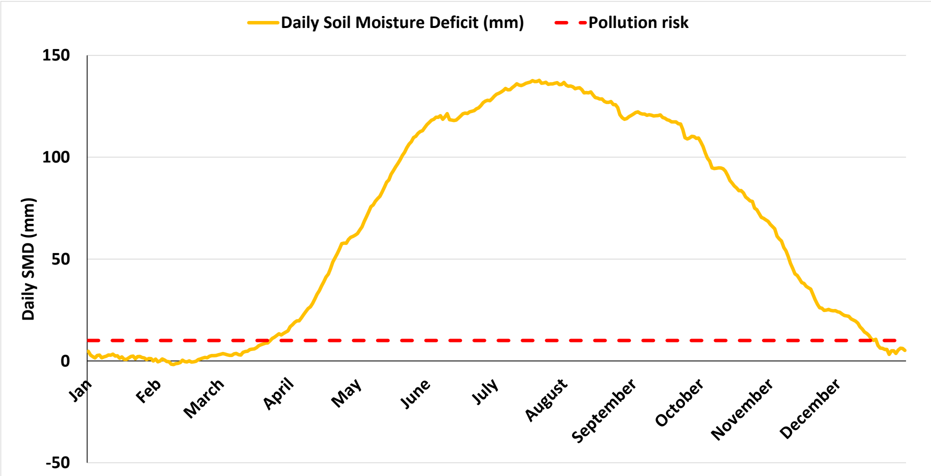

Figure 9. Predicted soil moisture deficits on clay soil under winter wheat in moderate rainfall areas (600-700mm)

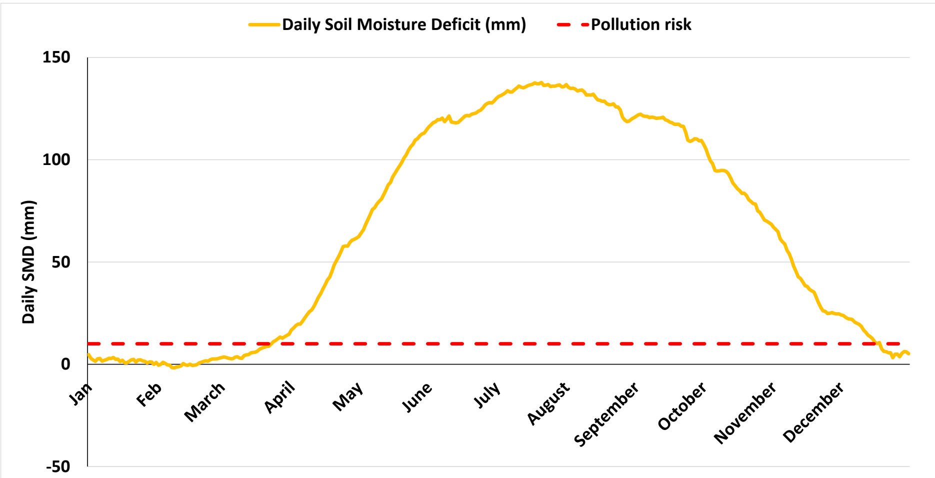

Figure 10. Predicted soil moisture deficits on clay soil under winter wheat in low rainfall areas (<600 mm)

5.2. Crop type

On arable land, biosolids applications to stubble can be incorporated into the soil as part of normal cultivation operations required for crop establishment, which brings with it the various benefits discussed earlier (e.g. reduced ammonia emissions, reduced risk of P losses). Topdressing to growing cereal crops is not a practical option for most farmers and is likely to result in reduced nutrient use efficiency; soil and crop damage; and reduced crop yield and quality. Additionally, the bulk nature of biosolids (like all bulky organic materials) makes it difficult to spread evenly across tramline spacings which typically extend to 36m. The applications are also likely to reduce crop productivity through smothering. Uneven application can result in sub-optimal nutrient supply to some plants, which can limit growth, and excessive nutrient supply to others, which may cause lodging in cereal and oilseed crops and lead to excessive N uptake which can also affect crop quality.

Additional cultivations to remediate soil compaction caused by application machinery may also make it impossible to apply biosolids before spring crop establishment.

5.3 Odour and associated public perception issues

Top-dressing biosolids to growing crops is also likely to increase odour nuisance as odour emitting surfaces will remain on the soil or contaminate the growing crop. More rigorous odour mitigation measures might be required including:

- Selecting application sites which are remote from residential settlements and housing,
- Restricting applications to small areas of land at any one time,

Incorporating biosolids into soils after application also breaks the source, pathway, receptor route for potential contamination of food products. Whilst there is no risk of any contamination of food resulting from topdressing biosolids to cereal crops, public perception and stakeholder reaction may prevent the practice.

5.4 Storage requirements

Bretell et al (2013) estimated that recycling biosolids to UK agricultural land involves the use of over 5,000 individual temporary field heaps each year. A Defra literature review of pollutant losses from solid manures stored in temporary field heaps (Williams et al., 2015a) found that free drainage of leachate from solid manure heaps is likely to contain high concentrations of nutrients (N and P), faecal indicator organisms (FIOs) and biochemical oxygen demand (BOD). Similar findings were reported by Brettell et al. (2013), who also found that leachates from biosolids field heaps can contain elevated concentrations of multiple pollutants. Consequently, undiluted leachate from solid manure or biosolids field heaps entering surface or ground water could pose a significant threat to water quality. Increasing the number of temporary field heaps and storing biosolids for longer is likely to increase the risk of point source pollution.

In practice, pollutants in leachate infiltrating soil underneath a field heap (and in run-off from the heaps) are likely to be either retained in the soil or diluted with 'uncontaminated' water from the rest of the field. Thus, pollutant concentrations will be reduced provided that there are sufficient 'barriers' between the field heap and the receiving water, with distance and slope and the presence of field drains being important influencing factors. The NVZ-AP rules (and the BAS) do not allow manure to be stacked in the field if it will give rise to free drainage and state that 'the field heap site must occupy as small a surface area as is practically required to support the mass of the heap and prevent it from collapsing' which will minimise these risks in NVZs. Indeed, Williams et al (2016b) concluded that results from field experiments to quantify the risk of ground and surface water pollution from solid manures stored in field heaps did not support changing the current guidance.

5.5 Soil health and carbon storage

Soil organic matter (SOM) is a key indicator of soil health; loss of SOM (due to changes in management, land-use and climate) is one of the most important threats facing our soils and can contribute to global warming (e.g. Lal 2016). Biosolids are typically made up of 40–70% organic matter, with the organic carbon (C) content ranging from 20–50%; the organic matter has usually undergone some degree of stabilization through anaerobic or aerobic digestion before being land applied (Torri et al 2014).

Biosolids applications to land have been extensively studied and shown to increase SOM (and hence soil organic C) across a range of soil types, climates, and cropping systems (e.g. Nicholson et al. 2018; Toffey & Brown, 2020). In their UK study, Powlson et al. (2012) reported an average increase in SOC of 180 kg C ha^{-1} yr^{-1} t^{-1} dry solids (ds) from digested biosolids applications (applied at *c*.8 t ds ha^{-1}), three times higher than the rate for farm manures (60 kg C ha^{-1} yr^{-1} t^{-1} ds). These results indicate that biosolids are a good source of stable OM for building up SOM levels, which has profound implications for soil physical properties and overall soil health. Whether or not this increase in SOM constitutes genuine C sequestration is still a matter for debate, depending as it does on the alternative fate of the biosolids if not applied to land (e.g. incineration, landfill) and whether or not the increase in SOM is finite and reversible.

Changing the timing of biosolids applications (from autumn to spring) would probably have a minimal effect on C storage in soils, although it is more difficult to assess the overall impact of the complete loss of the OM inputs from biosolids on soil health. However, if the change in interpretation reduces the quantities of organic manures applied to agricultural soils, this will negatively affect soil health and C storage in soils. Whilst biosolids application to land will always present some risk of diffuse pollution, it completes natural nutrient and OM cycles, contributes to UN Sustainable Development Goals and is recognised as the best practicable environmental option in most circumstances.

6. Summary and conclusion

Water quality

Evidence from the literature suggests that leaching losses from low RAN livestock manures (cattle and pig FYM) applied to free draining soils in England nitrate leaching losses are typically <5-10% of total N applied, and on drained clay soils less than 5% of total N applied. From the limited evidence base available, low RAN biosolids products (e.g. dewatered cake) behave similarly to FYM, with losses following autumn (September) applications to free draining soils at 7-11% of the N applied, dropping to <3% of the N applied following November and December applications. Indeed this is what MANNER-*NPK* predicts in terms of leaching losses following application of low RAN biosolids products.

Biosolids applications pose a low risk of P loss to water as they have a low water soluble P concentrations (typically <10% of total P). Autumn biosolids applications which are incorporated into the soil present a low risk of P loss via surface runoff. In contrast applications that are made to soils with low moisture deficits in the spring and which are left on the soil surface pose a greater risk of loss via surface runoff.

Air quality

Biosolids pose a low risk of ammonia and nitrous oxide emission reflecting their low RAN content. Evidence from livestock FYM suggests that ammonia emissions following applications of low RAN manures are c.5% of total N applied. Most of the ammonia emission occurs soon after application (80% within 24 hours) and soil incorporation within a few hours of application will reduce losses. Management strategies that prevent the rapid soil incorporation of biosolids (e.g. moving from applications on winter cereals/autumn stubbles to topdressing growing crops in spring) will increase ammonia losses. Nitrous oxide emissions from low RAN manures are largely controlled by soil and weather conditions in the period after application, with no consistent effect of application timing or method.

Crop available N supply from contrasting biosolids application timings

Outputs from MANNER-*NPK* suggest that the greatest risk of nitrate leaching losses are following early autumn (August and September) biosolids applications to sandy soils in arable production in high rainfall areas. Nitrate leaching losses from biosolids applications to medium/heavy soils were predicted to be lower than from applications to light/sandy soils reflecting the greater water holding capacity of medium/heavy soils. Crop available N supply from biosolids applications was lowest following autumn applications to light sandy soils at 10% of total N applied with no impact of rapid soil incorporation. This reflects the high risk of nitrate leaching on these soil types and suggests that any N saved as a result of reduced ammonia emissions following rapid soil incorporation are subsequently lost by leaching. On medium/heavy soils the crop available N supply from autumn applications is similar to that from spring surface broadcast applications at 15% of total N applied which suggests that any nitrate leaching losses saved by applying in the spring are balanced by ammonia emissions – an example of pollution swapping.

Practical considerations

Delaying biosolids applications until spring will increase the risks of soil compaction from application machinery. Outputs from the IRRIGUIDE model suggest that soils would not be dry enough to support the weight of application machinery until the end of March in low and moderate rainfall areas and early/mid-April in high rainfall areas. Delaying applications until late March/early April is likely to compromise spring crop establishment especially on medium and heavy soils. Applying low RAN biosolids products to growing crops in spring is likely to cause significant crop damage which will substantially reduce crop yields. It may also increase odour nuisance, given that the material will be left on the soil surface, and increase the risk of P runoff to water courses.

Conclusions

Based on recent data supplied by 9 Water Companies, the study was able to confirm that biosolids have a low RAN content (<30% total N) and pose a low risk of N loss to the environment. This also provides confirmation that they should be subject to the same diffuse pollution controls as other low RAN organic materials, including FYM and compost.

Information from the evidence review and from MANNER-*NPK* outputs suggest that preventing biosolids applications on 'high risk' sandy soils in arable production in the early autumn period would significantly reduce nitrate leaching losses. Sandy soils are suited to spring cropping which provide opportunities for late winter/early spring biosolids applications. Soil incorporation of spring applied biosolids (if soil conditions allow) before crop establishment will minimise ammonia losses and maximise crop N recovery, whilst also reducing odour nuisance and the risk of P runoff.

On medium/heavy soils nitrate leaching losses following autumn biosolids applications are much lower than on light sandy soils and supply the same quantity of crop available N as spring applications. Topdressing growing crops in spring with bulky low RAN organic manures (such as biosolids) is impractical on these soils due to the risk of soil compaction and/or crop damage. It would also result in increased ammonia volatilisation, odour nuisance and P loss (through run-off), as incorporation is not possible where a growing crop is present.

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