June 2021

Research Review No. 91140078

Impact Assessment – Farming Rules for Water

Bhogal, A.[1], Anthony S.[2] and Gooday R.[2]

[1]ADAS Gleadthorpe, Meden Vale, Mansfield, Notts. NG20 9PD
[2]ADAS Wolverhampton, Titan 1 Offices, Coxwell Avenue, Wolverhampton Science Park, Stafford Road, Wolverhampton, WV10 9RT

CONTENTS

1. Abstract

The Farming Rules for Water (FRfW) were introduced in April 2018 to fulfil obligations on diffuse pollution under the Water Framework Directive, particularly in regard to reducing phosphorus (P) losses to water from agriculture. Rule 1 aims to ensure that 'all reasonable precautions' are taken to prevent diffuse pollution following the application of organic manures and manufactured fertilisers. To comply with Rule 1, farmers must demonstrate they have planned nutrient applications to ensure they are applied in quantities that are sufficient to meet, and not exceed, the crop and soil requirements (e.g. by using a recognized nutrient management system, such as RB209).

Compliance with the rules is managed by the Environment Agency (EA) and recent clarification of the interpretation of Rule 1 by the EA has confirmed that farmers must demonstrate that the timing and quantity of organic manure applied is in accordance with crop and soil need at the time of application. This applies to all types of organic manure containing readily available N (RAN), and effectively rules out autumn and winter applications except to a crop that has a nitrogen fertiliser requirement in those seasons (e.g. winter oilseeds and grass to support late season growth in August and September). The lack of an autumn window for applications of all livestock manures, biosolids, digestate and other organic materials which contain RAN is likely to have a significant impact on manure and nutrient management on all farms as in many circumstances it will not be practical to apply manure in spring. Moreover, a change in practice may also increase the risk of losses of pollutants other than nitrate (e.g. ammonia emissions to air, and P loss to water - so called 'pollution swapping'). This impact assessment was commissioned to evaluate the impact of the EA's interpretation of Rule 1 on farm practice and the wider environment.

Currently almost 50 million tonnes (MT) fresh weight of farm manures, 1.9MT compost, 4.3MT digestate (from commercial facilities) and 3.5 MT biosolids are applied to agricultural land in England on an annual basis. A large proportion of these – particularly the solid, low RAN materials (e.g. cattle and pig farmyard manure (FYM), biosolids and compost) are applied and incorporated in the autumn ahead of autumn-sown cereals. This impact assessment focused on livestock manures and a scenario was developed which only permitted their application in the autumn (August and September) to either winter oilseed rape and grass (i.e. crops with a nitrogen fertiliser requirement in the autumn, according to RB209 and NVZ rules). Using British Survey of Fertiliser Practice (BSFP) data, all other livestock manure applications in the months of August - November were moved to spring (February – April, to the same crop types). This resulted in the movement of c. 7 million tonnes of solid manures and 3 million m^3 of slurry from autumn to spring, with spring slurry applications assumed to be applied using band-spreading equipment and solid manures broadcast and incorporated ahead of spring crops or top-dressed onto winter cereals.

Moving applications from autumn to spring poses several practical and logistical challenges to the management of organic materials, particularly associated with storage and spreading. For solid manures this is likely to result in an increase in the number of temporary field heaps and the time that the manure is stored in field. For slurries this will require an increase in the storage capacity of slurry tanks or lagoons. Where there is access to grass, it is likely that a minimum of 6 months storage will be required, however, where slurry is only applied to arable land at least 9 months storage is likely to be necessary. Moreover, ensuring that organic material applications are made to soils when they are strong enough to withstand the weight of spreading equipment, is an important practical consideration. Travelling on 'wet' soils with heavy spreading equipment is likely to cause significant compaction (with associated increased run-off and erosion), especially on clay and medium textured soils. The proportion of days in a month when medium textured soils are at or close to field capacity (i.e. 'wet') was estimated from 30 year average rainfall data (1981-2010) in order to assess the impact of application timing on soil compaction and run-off risk. In the summer and early autumn (May – September), when *c*.50% of solid manures and 35% of slurries are currently applied the risk of soil compaction and run off is low (soils at field capacity for <10% of month). By contrast, in February and March, soils are at field capacity for on average 70-80% of the month, and the number of available spreading days are significantly lower than in summer and early autumn. Moving the majority of autumn applications to spring will increase pressure on the number of days available for safe spreading (i.e. where the risk of soil damage or run-off is low), at a time when there is also a higher risk of soils returning to field capacity shortly after application.

A modelling approach was used to estimate changes in nitrate and phosphorus losses to water and ammonia and nitrous oxide losses to air (post storage) resulting from restrictions on the timing of applications required to meet the EA's interpretation of Rule 1. Under current practice (baseline) the application of organic materials in England was estimated to result in annual ammonia volatilisation losses of 31kT NH_3-N, nitrate leaching losses of 9 kT NO_3-N and nitrous oxide emissions of 1kT NO_2-N, respectively. Baseline annual phosphorus losses from farm manure applications were estimated at 0.7kt P loss. Nutrient losses from manure applications were estimated to contribute c. 20% of the total ammonia emission, 2.5% of the total nitrate-N loss and 15% of the total P loss from all agricultural sources. The EA's interpretation of Rule 1 was predicted to:

- reduce nitrate leaching losses by c. 60% (1.5% decrease in the total loss from agriculture)
- increase in ammonia emission by c. 10% (2% increase in total emissions from agriculture)
- increase P loss by c. 30% (5% increase in the total loss from agriculture)

The increase in ammonia emissions and P losses largely reflected the reduction in soil incorporation resulting from a change from applications to autumn stubbles to top-dressing to growing crops in spring. Increases in P losses are also exacerbated by applications to wet soils in the spring. Soil incorporation is an important mitigation method for controlling ammonia volatilisation to air and P losses to water via surface runoff/bypass flow. For livestock manures, these impacts are likely to be greatest in the East of England where most pig and poultry manures are applied ahead of autumn cropping.

The FRfW aims to prevent diffuse pollution following the application of organic manures and manufactured fertilisers, stating that materials should not be applied ‘if there is a significant risk of agricultural diffuse pollution’. This impact assessment has shown that the effective management of organic materials requires consideration of the ‘balance of risks’ to water, air and soil, as well as practical considerations, taking into account not only the type of organic material and when it is applied, but how and where it is applied. Autumn applications to light textured soils present the greatest risk of nitrate leaching. The risks of soil damage from spring applications is also lowest on light soils. By contrast ammonia losses to air and phosphorus losses to water pose the greatest pollution risk on clay and medium soils, and spring applications pose a significant risk of compaction on these soil types. Clay and medium soils also have more limited opportunities for spring cropping (and hence the potential for soil incorporation). Based on these findings, a matrix has been drafted, which aims to guide the industry on when and where organic materials can be used most effectively to reduce the risks that applications pose to diffuse water and air pollution and soil compaction.

2. Introduction

The Farming Rules for Water were introduced in April 2018 (Defra, 2018; SI 2018) to fulfil obligations on diffuse pollution under the Water Framework Directive (WFD, 2000/60/EC; SI 2017) particularly in regard to reducing diffuse phosphorus (P) pollution. Eight rules were published to reduce diffuse water pollution from agriculture in a way that minimised costs to the farming sector by focusing on nutrient use efficiency and soil management. Rule 1 (Regulation 4 of the Reduction and Prevention for Agricultural Diffuse Pollution (England) Regulations 2018) aims to ensure that 'all reasonable precautions' are taken to prevent diffuse water pollution following the application of organic manures and manufactured fertilisers. More specifically the rule states:

1a) Application to cultivated land must be planned in advance to meet soil and crop 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.

An Impact Assessment carried out by Defra prior to the introduction of these rules, demonstrated both environmental and economic benefits of adopting the measures proposed (Defra, 2018). The assessment used the FARMSCOPER model (Gooday et al. 2013) to estimate net impacts on farm income and the environment; for Rule 1, the scenario tested included using a fertiliser recommendation system, integrating fertiliser and manure nutrient supply and not applying P fertiliser to high P soils (FARMSCOPER measures 22, 23 and 32 respectively). Defra subsequently published guidance on the rules in April 2018 which stated:

'For all farming and horticultural land you must plan:

- How much fertiliser or manure to uses, so you do not use more than your crop or soil needs
- By assessing the pollution risks
- By taking into account the weather conditions and forecasts at the time you want to apply manure or fertiliser on your land

You can use the Nutrient Management Guide RB209 to work out the nutrients your soil or crop needs.'

Compliance with the rules is managed by the Environment Agency (EA). In 2019, in response to concerns from its members regarding the EA's interpretation of Rule 1, the NFU sought a clearer explanation of this rule and received a clarification note from the EA (Tried and Tested, 2019) which stated:

- 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.

Further clarification of the interpretation of Rule 1 by the EA in autumn 2020 confirmed that farmers must demonstrate that the timing and amount of organic manure applied is in accordance with crop and soil need at the time of application. This applies to all types of organic manure containing readily available N (RAN), and effectively rules out autumn and winter applications except to a crop that has a nitrogen fertiliser requirement in those seasons (e.g. winter oilseeds and grass to support late season growth in August and September). For example, farmyard manure (FYM) contains RAN, but as winter wheat does not have a requirement for manufactured nitrogen fertiliser in the autumn or winter, the EA's current interpretation is that its application should not be permitted.

Under the Nitrate Vulnerable Zone (NVZ) Action Plan (SI, 2015) which aims to control nitrate leaching, closed spreading periods were introduced for the application of manures with a high RAN content (such as slurries, digestate and poultry manures) in areas where nitrate concentrations in ground and surface waters exceed 11.3 mg/l NO_3-N (c. 60% of England). The FRfW apply to the whole of England so any changes in manure management required to meet the EA's interpretation of Rule 1 will affect all farmers using organic materials both inside and outside the current NVZ areas. Autumn and winter spreading of low RAN organic materials is common practice, with approximately 50-60% of the c. 45 million tonnes of FYM produced in the UK spread at this time.

Limiting the amounts of organic materials which contain readily available N that can be spread in the autumn (i.e. all livestock manures, biosolids, digestate and other organic 'wastes' except green compost) is likely to have a significant impact on manure and nutrient management on all farms as soil and crop types may not be suited to spring manure application timings. The change in practice may also result in greater losses to the environment via other routes e.g. ammonia emissions to air, P loss to water and increased risk of soil damage as a result of applications being made to wet soils that cannot bear the weight of manure spreading equipment.

2.1. Objectives

The overall aim of this work was to assess and evaluate the impact of the EA's interpretation of Rule 1 of the Farming Rules for Water (FRfW) on the application of organic materials during the autumn and winter on farmers and growers, as well as the wider environment.

The detailed objectives of the study were to:

- Assess the impact of restrictions on farm practice (including changes to where, when and how manures are stored and applied, and associated financial and environmental risk associated with these changes)
- Assess the impact of restrictions on the wider environment, including impacts on soil health and organic matter/carbon storage, water (N & P) and air quality (NH_3, GHG and odour)
- Draft a matrix on the responsible use of organic materials
- Identify gaps in knowledge and further research required.

3. Methodology

A modelling approach was used to estimate changes in nitrate and phosphorus losses to water and ammonia and nitrous oxide emissions to air as a result of the restrictions on the timing of applications required to meet the EA's interpretation of FRfW Rule1. Detailed national survey records were used to determine quantities and timing of manure application for baseline (present day) and future scenarios based on the EA's interpretation of Rule 1. This section of the study report provides an over-view of the datasets and models used.

3.1. Physical Geography

All model calculations were carried out on a disaggregated spatial grid of 10 by 10 km^2 cells (*n* 1,500) covering England, and the results summarised to Environment Agency and Natural England 'Public Face Area' administrative boundaries (*n* 14; Figure 3-1), This approach ensured disclosure of survey statistics relating to individual farms was prevented and represent the spatial uncertainties in the distribution of spread manures around originating farms. The calculation and reporting scales were sufficient to maintain a representation of the distinct agricultural regions of England and of the variation in soil type and climate that are important factors controlling environmental emissions.

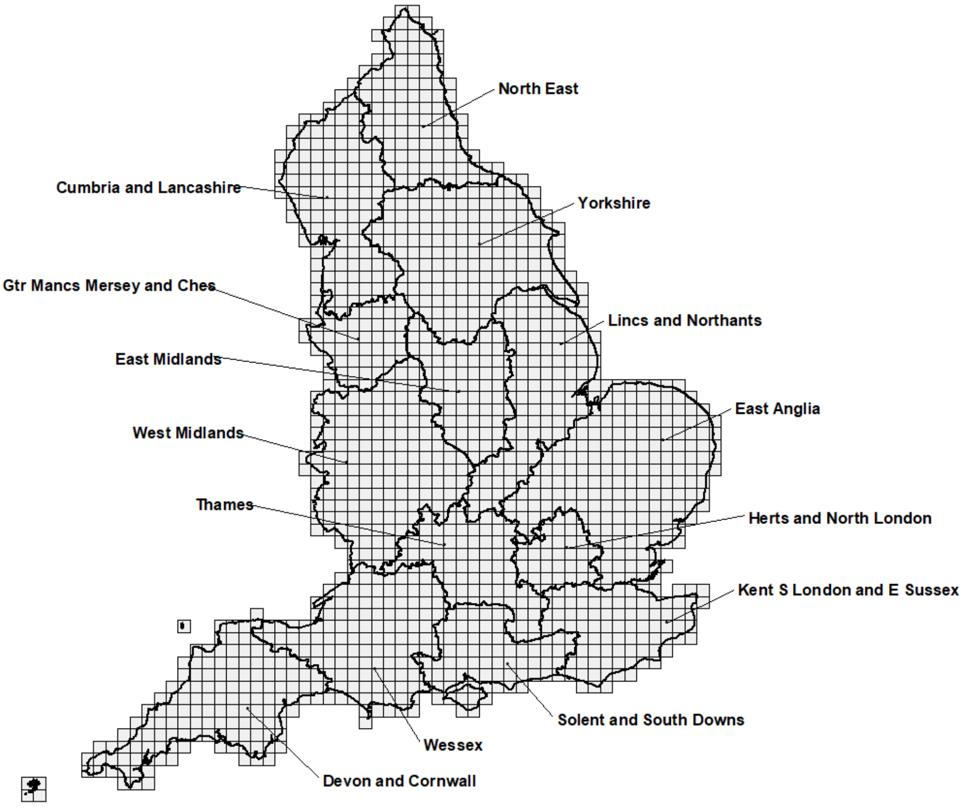


Figure 3-1 Environment Agency and Natural England 'Public Face Area' administrative boundaries (Environment Agency, 2017).

3.1.1. Soil Type

For each 10 by 10 km^2 cell, the percentage distribution of the 7 'RB209' soil types included in AHDB's 'Nutrient Management Guide (RB209)', were sourced from a summary previously prepared under Defra project AC0114. This summary used the Land Information System – National Soil Map (NATMAP) of soil association boundaries mapped at a scale of 1:250,000 with supporting information on the relative area of individual soil series within each association, and the 'RB209' soil type assigned to each series. Model calculations were made for each soil type present within a cell, and the model results area weighted as required. The percentage distribution of soil types to which manures are applied differed from the distribution in the NATMAP in response to the pattern of managed manure production across the cells. Appendix Table 11.1 summarises the percentage of the managed manure spread to each 'RB209' soil type, by 'Public Face Area', in the baseline (present day) scenario. The national area-weighted average properties of the group of all soil series belonging to each 'RB209' soil type in England, including their total and plant available water content at field capacity, were sourced from a summary also previously prepared under Defra project AC0114 (Appendix; Table 11.2) with supporting information from the MAFF Agricultural Land Classification of England and Wales (MAFF, 1988) and Tetagan *et al*. (2011) for the water content of peat and shallow stony soils.

3.1.2. Climate

For each 10 by 10 km^2 cell, the long-term (1981 to 2010) observed monthly average minimum and maximum daily air temperature, rainfall, sun hours, wind speed and number of rain days (> 1 mm) were derived from the UKCP09 Met Office 'Gridded and Regional land Surface Climate Observation Dataset' at a native spatial resolution of 5 by 5 km^2 (Hollis and McCarthy, 2017).

3.1.3. Crop and Grass Area

For each 10 by 10 km^2 cell, the annual average areas of farmed crop and improved grass land in the period 2015 to 2019 were taken from the spatial database of the Agricultural Ammonia and Greenhouse Gas Inventory (AAGHGI). The areas were disaggregated into seven crop groups (Appendix Table 11.3), and the cereal group further disaggregated into spring and winter sown types.

The total areas belonging to all farms within each cell were originally sourced from the annual June Agricultural Survey (Defra, 2015 to 2019) and spatially referenced in a process involving government records of the centroids of fields belonging to a farm, of cattle recorded by the Cattle Tracing System (CTS) and belonging to a farm, the parish within which the farm is located and/or the full post-code of the farm office.

3.2. Manure Quantity

Estimates of the annual average quantity of total nitrogen and readily available nitrogen (ammonium and nitrate) in managed manure produced by pig, poultry and cattle within each 10 by 10 km^2 cell were sourced from the spatial database of the Agricultural Ammonia and Greenhouse Gas Inventory (AAGHGI, 2015 to 2019).

The AAGHGI modelling framework implements the 'Feed into Milk' (Thomas, 2004) and the 'Energy and Protein Requirements of Ruminants' (AFRC, 1993) energy requirements models to calculate the annual dry matter and protein intake of cattle, and thereby nitrogen excretion in dung and urine, taking into account animal productivity (rate of growth, lactation and gestation) and seasonal changes in available feed types. Nitrogen excretion by pig and poultry is estimated by feed and product balances (see, for example, Nicholson *et al*., 2016; Cottrill and Smith, 2006). The AAGHGI then implements an explicit representation of the nitrogen mass flow (Webb and Misselbrook, 2004) for all animal types in the farm manure management chain, to determine the quantity of excreta that is managed as manure and stored by various means prior to spreading to crop and grassland, using national survey data on type and time spent by animals in housing, and the types of manure storage in use.

The quantities of nitrogen were extracted from the AAGHGI database at the point of spreading, net of any prior effects of organic nitrogen mineralisation and ammonia volatilisation (and any emission mitigation methods in place) in housing and storage. These were then converted to estimates of the tonnage or volume of manure at point of spreading by division by a reference measured average nitrogen content of manures leaving storage, derived from the ADAS Manure Analysis Database (MANDE; Defra projects NT26005 and SCF0202; 2003 and 2015). The phosphorus content of manure at point of spreading was similarly estimated by division of the nitrogen quantity by a reference measured average nitrogen to phosphorus content ratio in manures leaving storage. Appendix Table 11.4 summarises the reference nutrients content of managed manures used in this study.

3.3. Manure Management

Estimates of the percentage of each managed manure type spread to each crop group and the methods of application were derived from the British Survey of Fertiliser Practice (BSFP), using aggregate data for the years 2015 to 2019 to increase the sample size and improve the robustness of the derived statistics.

The annual British Survey of Fertiliser Practice (BSFP) commenced in 1983 (Chalmers, 2001), extending a survey for England and Wales that has existed in some form since 1942 (Chalmers *et al*., 1990). In recent years it has been based on an annual sample of around 1300 farms, randomly selected from the June Agricultural Census, and stratified by robust farm type and size. The survey is restricted to farms having a total area of improved grassland and crops of at least 20 ha.

The BSFP records individual manure applications to field blocks on surveyed farms. The information collected includes the manure type (slurry; solid; *and* farm produced digestate), animal type (pig; poultry; and cattle), crop type (re-classified into the crop groups used in this study), rate of application and area manured, month of application, method of application (broadcast; band spread; *and* injection), and the time delay to incorporation if incorporated (within 6 hours; within 24 hours; within 1 week; *and* within more than 1 week). Statistical information on the method of incorporation (mouldboard plough; *and* disc or tine) were separately sourced from the Defra Farm Practices Survey for England (2014).

The BSFP survey records were subject to quality control, with around 15% of records discarded because of missing data or implausible practices such as manure incorporation on permanent grassland (not re-sown). Comparison of the estimated percentage of manure spread by month, with and without record discards, showed only small differences (Appendix, Table 11.5). It is therefore unlikely that this analysis has been significantly affected by the record discards. The number of individual farms and applications of manure contributing to the quality controlled dataset ranged from <20 farms (for pig & poultry manures) to over 400 farms (cattle FYM; Appendix Table 11.6). The small number of records for pig and poultry manures was the justification for aggregating data across a five year (2015 to 2019) period to represent the baseline.

The BSFP survey records of manure applications were individually input to the computer models of emissions (see below), allowing easy implementation of any scenario change in management, for example a change of month of application. For this study, the survey findings from farms located only in England were analysed, and the emission model outputs were scaled to the national level using the inverse of the achieved sampling fraction in each survey strata in each year. The count of the total number of farms in England in each strata were taken from the spatial database of the AAGHGI, and the count of the number of surveyed farms (after record discards) directly from the BSFP database.

3.4. Emission Models

3.4.1. Drainage

For each 10 by 10 km^2 cell, for each 'RB209' soil type and crop group, estimates of long-term (1981 to 2010) monthly average surface runoff, canopy and crop evapotranspiration, and soil drainage (effective rainfall) were calculated using the Mean Climate Drainage Model (MCDM; Anthony, 2003). This model combines Penman-Monteith calculations of potential evapotranspiration (Allen *et al*., 1998) with the monthly water balance book-keeping approach of Thomas (1981) and Alley (1984) to estimate monthly values of the soil moisture deficit and drainage at a location. The MCDM calculations used an appropriate seasonal profile of crop canopy height and leaf area index, and of root depth, based on the crop parameterisation used by the MORECS model (Hough and Jones, 1997). Seasonal surface run-off was estimated using the empirical Standard Percentage Runoff (SPR; Boorman *et al.,* 1995) index assigned to each 'RB209' soil type, modified by current soil moisture deficit, to scale output from a meta-model of the Green-Ampt (1911) infiltration excess calculation applied to a modelled distribution of rainfall intensities and maximum soil infiltration rates (Anthony, 2003). All emission models used the same MCDM monthly runoff and drainage calculations as input to their loss calculations.

3.4.2. Nitrate

Over-winter nitrate leaching losses from spread manures were calculated as a function of 'RB209' soil type water content at field capacity (a measure of retention) and the effective rainfall (rainfall net of canopy and crop evapotranspiration) between the date of manure application and end of soil drainage, as calculated by the MCDM (above). The nitrate elution functions were those used by the MANNER-*NPK* model (Nicholson *et al*., 2013). Separate elution functions were used for free draining soils and for poorly drained soils with the potential for by-pass flow, based on simulations with the SACFARM (Addiscott, 1977; Addiscott and Whitmore, 1991) and EDEN (Gooday *et al*., 2008) models. The calculations took account of organic nitrogen mineralisation, ammonia volatilisation and rate of nitrification of the manure ammonium nitrogen content following application, which together affected the quantity of manure inorganic nitrate at risk of leaching.

3.4.3. Ammonia and Nitrous Oxide

Ammonia emissions from spread manures, as a percentage of the readily available nitrogen content, were calculated as a function of the manure type and readily available nitrogen content, and the method and time delay to incorporation, using the empirical functions in MANNER-*NPK* model (Nicholson *et al*., 2013). Rapid (within 6 hours) and deep (mouldboard plough) incorporation act to minimise ammonia volatilisation, at the risk of increasing the quantity of manure inorganic nitrogen remaining in the soil that is subsequently leached.

Nitrous oxide emissions from spread manures were calculated as a fixed percentage of the manure total nitrogen applied, limited to the manure readily available nitrogen content, using the meta-analysis reported by Thorman *et al.* (2020) for manure spreading experiments in the United Kingdom.

3.4.4. Phosphorus

The soluble phosphorus content of each manure type, at risk of loss in surface runoff and drain flow, was a fixed 31% of the total phosphorus content for solid manure and 47% for liquid manure (Davison *et al*., 2008). Incidental phosphorus loss in runoff and drain flow from spread manures was calculated as a function of the manure type and soluble phosphorus content, and the rainfall frequency and time delay to incorporation, using the empirical functions in the PSYCHIC model (Davison *et al*., 2008).

4. Current (baseline) practice

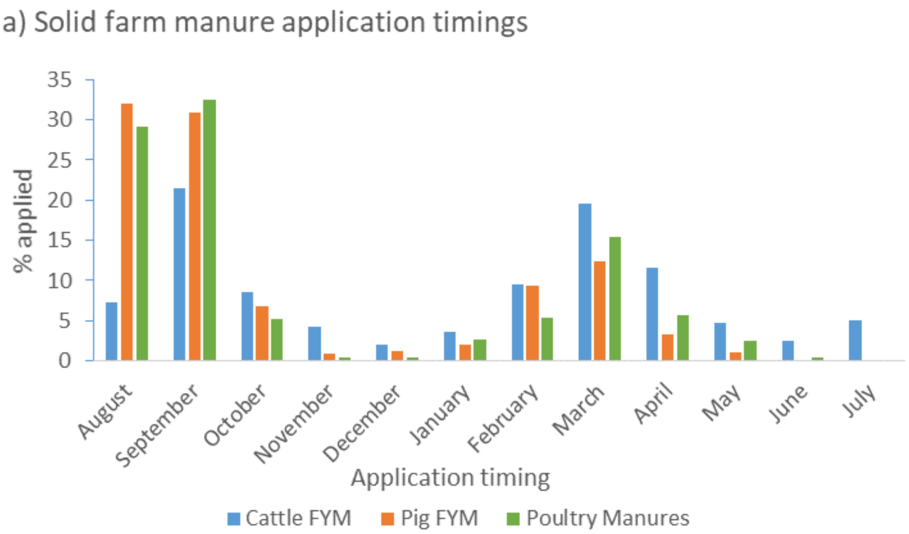
Currently almost 50 million tonnes-MT (fresh weight) of farm manures are applied to agricultural land in England on an annual basis (BSFP database). In addition to this, approximately 1.9MT compost, 4.25MT digestate (from commercial facilities, excluding water companies; WRAP, 2020) and 3.5 MT biosolids (assuredbiosolids.co.uk) are also applied (Table 4.1).

Table 4.1 Estimated quantities of organic materials applied to agricultural soils in England

Organic Material	Total applied (MT, fresh weight)
Livestock manures:	**48.2**
Cattle FYM	14.2
Cattle Slurry	24.9
Pig FYM	2.1
Pig Slurry	2.9
Poultry Manures	1.9
Farm-based Digestates	2.2
Compost	**1.9**
Digestate*	**4.3**
Biosolids	**3.5**

*From commercial plants, not farm-based; 2018 data

Of the solid farm manures, approximately 45% of cattle FYM, 70% of pig FYM and 68% of poultry manures are applied in the autumn/early winter period (August – December), particularly in the months of August and September (Figure 4-1a), with over 60% of the pig and poultry manure and 25% of cattle FYM applications applied to autumn sown cereals and oilseeds (Figure 4-2a,c,e). The opposite pattern is seen with slurries (particularly cattle slurry), with the majority applied in spring, and only c.10% of cattle slurry and 35% of pig slurry applied in the autumn (Figure 4-1b). The majority of cattle slurry is applied to grassland, but almost half of the pig slurry is currently applied to autumn sown cereals and oilseeds (Figure 4-2b,d). Manure-based digestate applications are split evenly between autumn sown crops, spring sown crops and grassland, although this distribution varies with feedstock type, with pig and poultry digestates applied to cropland and cattle digestates applied to grassland.



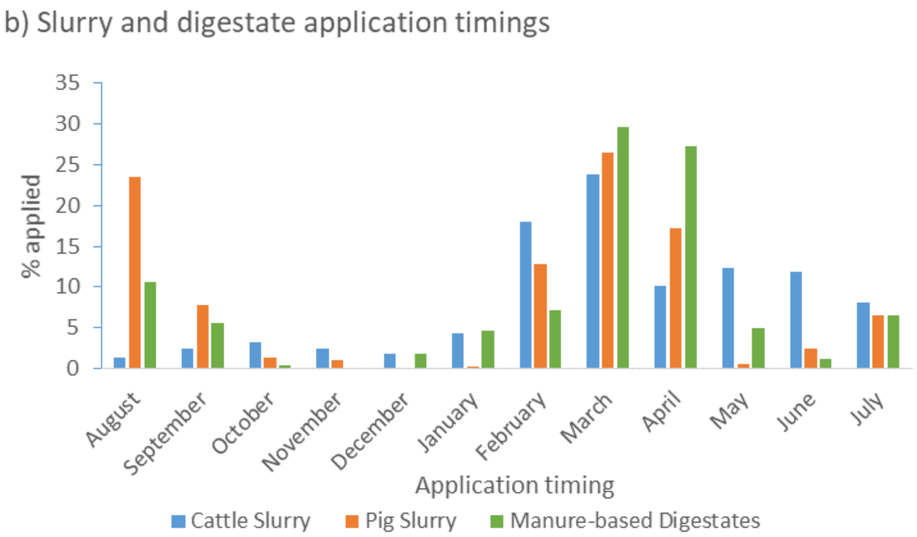


Figure 4-1 Timing of farm manure applications in England

Biosolids are typically applied at similar timings to pig FYM and poultry manures, with approximately 50% applied in August and September, largely ahead of winter cereals and oilseed rape (Figure 4-3; data supplied by 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; M. Taylor, pers.comm.).

A brief survey of Renewal Energy Association (REA) members was conducted by REA as part of this study (J. Grant, pers.comm). Due to the small sample size (18 respondents; 4 compost producers and 14 digestate producers), the following information is only indicative of the timing and destination of applications, and not necessarily representative of all producers. For compost (4 responses, 60,000t or c.3% of total compost applied to land), approximately 50% was applied in the autumn and winter (September – February) ahead of cereals and oilseeds, with spring application ahead of spring cereals and maize (Figure 4-4a). For digestate (15 respondents, including both food-based and farm-based digestate producers; 720,000t or c.15% of digestate application to land), over 70% was applied in the spring and summer ahead of spring cropping (cereals, sugar beet and maize) and to grassland, with the remaining 30% applied in the autumn and winter to grassland, winter cereals and other crops such as cover crops and stubble turnips (Figure 4-4b).

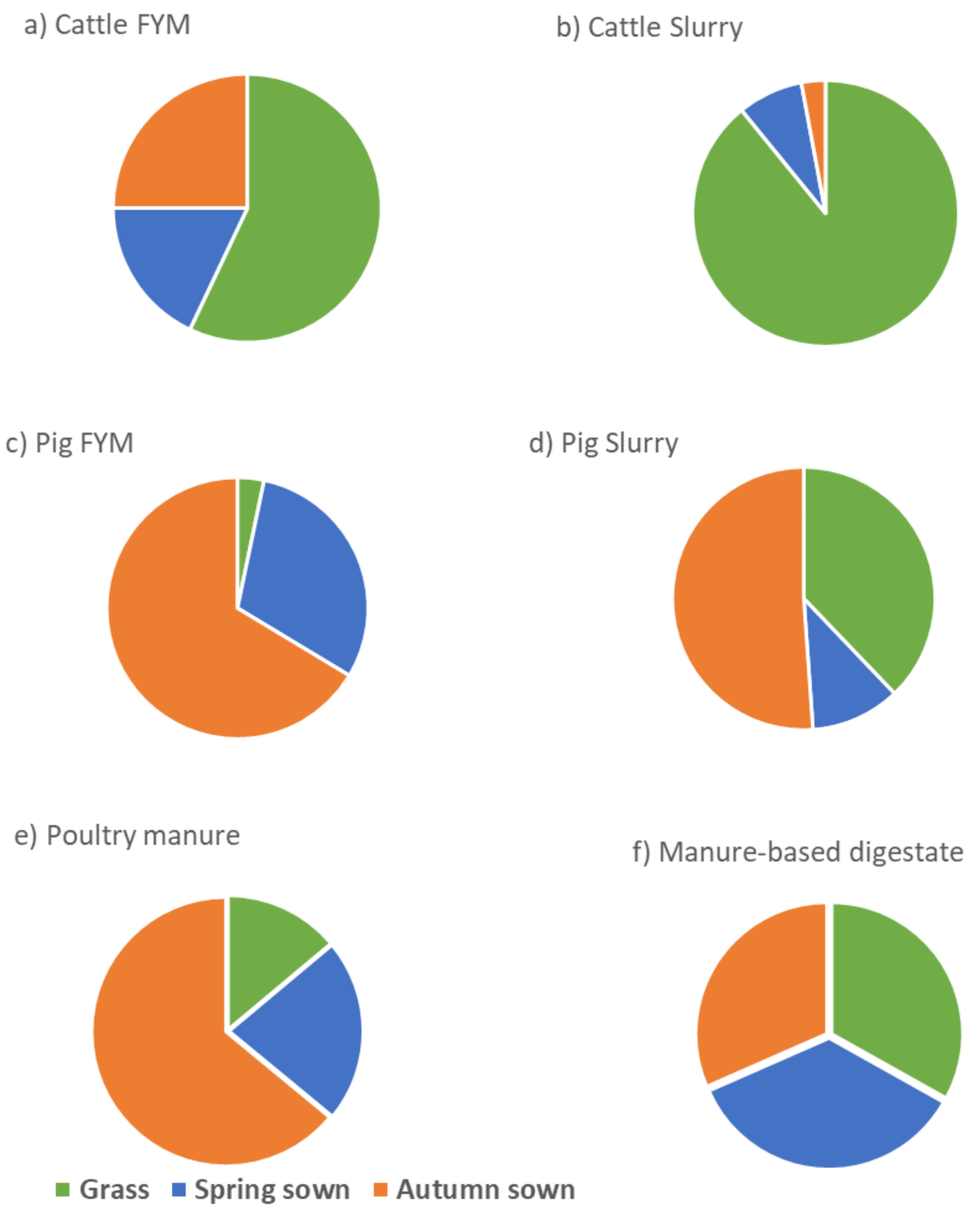


Figure 4-2 Destination of livestock manure applications

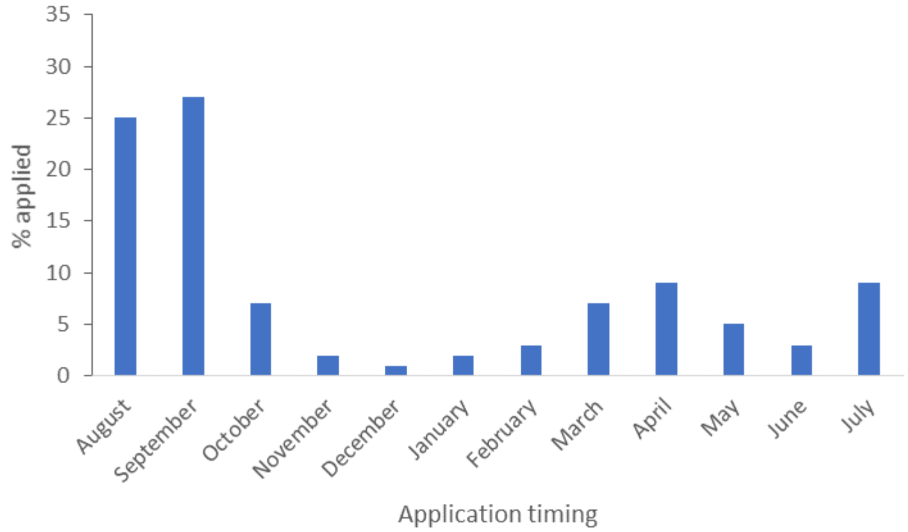


Figure 4-3 Biosolids application timings (Data supplied by 9 Water Companies; M. Tayor, pers.comm);

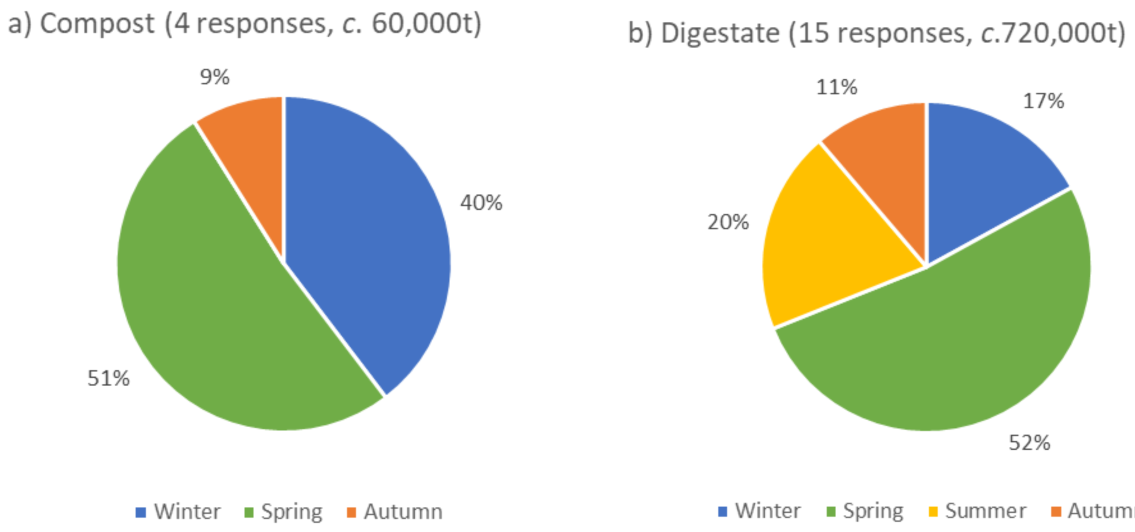


Figure 4-4 Compost and Digestate (food and farm-based) application timings (J. Grant, pers. comm.); Winter (Dec-Feb); spring (Mar-May); Summer (June-Aug); Autumn (Sept – Nov). Note Summer applications include the month of August.

5. FRfW Rule 1 Scenario

Rule 1 of the FRfW states:

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

It was agreed at the initial inception meeting that the current interpretation of Rule 1 of the FRfW by the Environment Agency, does not allow the application of an organic material that contains readily available N (RAN) in the autumn or winter unless the crop has a nitrogen fertiliser requirement in that season.

Nitrogen (N) is present in organic materials 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.

The Nutrient Management Guide (RB209) provides data on the 'typical' nutrient contents of organic manures based on samples analysed as part of research projects and other national sampling programmes. Table 5.1 shows the typical N contents of the commonly applied organic materials from RB209, with data from the previous edition of RB209 (Anon, 2010) used to show the typical RAN content of these materials (Table 5.1).

Table 5.1 Typical total N content (fresh weight basis) of organic materials applied to agricultural land

Organic material	Dry matter (%)[1]	Total N (kg/t)[1]	RAN (% of total N)[2]
Cattle FYM	25	6.0	10-20
Pig FYM	25	7.0	15-25
Poultry manure	60	28	35
Cattle slurry	6	2.6	46
Pig slurry	4	3.6	70
Biosolids (digested cake)[3]	25	11	15
Green compost	60	7.5	<3
Green/food compost	60	11	5
Food-based digestate	4.1	4.8	80[4]

[1]The Nutrient Management Guide (RB209)

[2]The Fertiliser Manual (RB209) 8th Edition (Anon, 2010)

[3]Recent data supplied by 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) report virtually identical dry matter, total N and RAN contents for digested cake (11 kg/t total N of which 13% is RAN). These companies also reported RAN contents for lime stabilised and advanced AD biosolids at 5 and 20% total N content, respectively.

[4]Digestate and Compost in Agriculture: Good Practice Guidance; WRAP 2016

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 (SI, 2015). As can be seen from Table 5.1, this includes livestock slurry, poultry manures and whole/liquid anaerobic digestate (also liquid digested sewage sludge). As a result these organic material types are subject to closed period spreading rules under the NVZ Action Plan. By contrast, Table 5.1 demonstrates that Rule 1 of FRfW applies to all livestock manures, digestates, green/food compost and biosolids as they contain RAN ranging from 5-80% of the total N content. The only organic material listed in RB209 which is not subject to timing restriction is green compost which contains 'negligible' RAN (at <0.2 kg RAN/t, equivalent to <6 kg/ha RAN from a 30 t/ha application).

The Nutrient Management Guide (RB209) also provides guidance on recommended timings for N fertiliser application for individual crops, whilst acknowledging that NVZ rules relating to closed periods should take priority:

- For winter sown wheat and barley there is no requirement for fertiliser N in the seedbed.
- Autumn applications of 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 fertiliser N should be applied in spring or early summer when sward demand is greatest.

NVZ rules allow closed-period applications of manufactured fertiliser N to winter oilseed rape, grass and some field vegetable crops (Table 5.2), recognising that these crops have a small fertiliser N requirement at this time. For organic materials, RB209 recommends to make best use of their N content, they 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 where N leaching is the main loss pathway, the fertiliser N replacement value of the organic material will be increased. This is particularly important for organic materials with a high RAN content and where applications are made to sandy/shallow soils. Where low RAN materials are applied to medium/heavy textured soils, application timing has only a small impact on organic material fertiliser N replacement value as N leaching losses following autumn application timings are usually similar or less than ammonia emissions from spring application timings (RB209, section 2).

Table 5.2 NVZ crop types permitting manufactured fertiliser applications in the autumn

Crop	Maximum manufactured fertiliser nitrogen rate (kg/ha)
Winter oilseed rape	30 (up until 31st October)
Grass	80 (up until 31st October, maximum of 40kg/ha per application)
Brassica	100 (50 kg/ha every 4 weeks until harvest)
Over-winter onions, parsley	40
Asparagus	50

Assuming the NVZ permitted autumn manufactured fertiliser N applications in Table 5.2 reflect those crops which have an autumn N requirement, it can be seen that the EA's interpretation of FRfW Rule 1 restricts the post- harvest (late summer/autumn) application of **all** organic materials (except green compost) to winter oilseed rape and grassland and only a very small land area of field vegetables.

The scenario developed therefore assumed:

- Organic materials could only be applied ahead of OSR and to grass in August and September; materials which are currently applied to other crop types in the autumn or to grass and OSR after 30th September (regardless of the type of organic materials & RAN content), were moved to the spring in the order:
    - August application dates → Applied in February
    - September application dates → Applied in March
    - October & November application dates → Applied in April
- Applications were to the same crop type which changed the method of application and incorporation in some circumstances, with the following assumptions:
    - Spring cropping – solid and liquid organic materials broadcast and incorporated within 24 hours of application
    - Winter cropping – solid organic materials top-dressed, with no incorporation; liquid manures applied by band-spreader (trailing hose), with no incorporation.
    - Grass – solid organic materials broadcast, with no incorporation, liquid manures applied by band-spreader (trailing shoe)

See Figure 5-1 for a schematic of these assumptions.

Change Timing of Manure Application

Baseline	Scenario
August	February
September	March
October to December	April

If Timing Changed then Change Method of Manure Application

Manure Type	Baseline	Scenario
SLY	Broadcast	Band Spread

If Timing Changed then Change Manure Incorporation Delay

Crop Type	Scenario
Spring Sown	Within 24 Hours
Autumn Sown	Not Incorporated

Figure 5-1 Scenario development

Initially the scenario assumed that all liquid manures were applied by band-spreading equipment, as this is a future requirement of the Clean Air Strategy. However, currently only *c.* 20% of slurries are applied by band spreading (or injection) to grassland and *c.* 35% to cropland, reflecting the availability of spreading equipment and contractors with suitable machinery (Figure 5-2). To reflect the current situation, a second scenario was run that assumed the current method of slurry application i.e. autumn broadcasting → spring broadcasting; autumn bandspreading → spring bandspreading (Table 5.3).

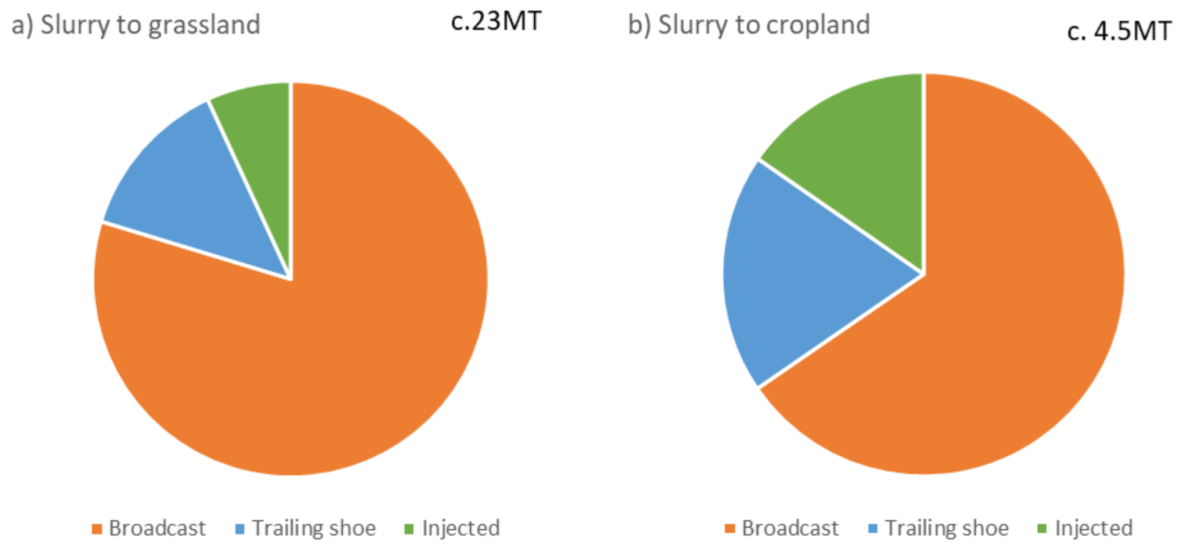


Figure 5-2 Current method of slurry application to grass and crop land (England)

Table 5.3 Scenarios evaluated

Scenario	Details
BASELINE	Current practice, as determined from BSFP
OPTIMISED SCENARIO	Applications moved to the same crop type in the spring (unless applied to grass and OSR in August & September); All slurries applied using a bandspreader; applications ahead of spring crops rapidly incorporated within 24 hours
CONSTRAINED SCENARIO	Applications moved to the same crop type in the spring (unless applied to grass and OSR in August & September); Current method of slurry application maintained; applications ahead of spring crops rapidly incorporated within 24 hours

6. Implications for farm practice

Moving organic material applications from autumn to spring poses several practical and logistical challenges to the management of organic materials, particularly storage and spreading.

6.1. Storage of organic materials

Implementation of the scenario outlined in section 5 (Figure 5-1 and Table 5.3), would result in the movement of approximately 7 million tonnes of solid manures and 3 million m^3 of slurries from autumn/winter applications to spring (Table 6.1). This is equivalent to over 60% of pig FYM, 40% of poultry manure and 35% of cattle FYM, the majority of which (80-90%) are currently applied and incorporated ahead of autumn sown crops. Only *c*. 10% of cattle slurry would be affected, as this is more commonly applied to grassland, however over 2 million m^3 of extra cattle slurry storage would be required. Moreover, up to 20% of pig slurry currently applied to cropland would be moved from autumn to spring. For solid organic materials an increase in spring applications is likely to result in an increase in the number of temporary field heaps, and the length of time they are left in fields. For slurries this will require an increase in the storage capacity of slurry tanks or lagoons.

Table 6.1 Potential movement (and storage requirement) of organic materials as a consequence of implementing the scenario

Manure type	Total applied (MT)	% Applied Aug-Dec		Quantity moved to spring	
		Baseline	Scenario	MT	%
Cattle FYM	14.2	43.6	8.2	5.1	35.5
Cattle Slurry	24.9	11.4	2.0	2.3	9.4
Pig FYM	2.1	71.9	9.2	1.3	62.8
Pig Slurry	2.9	33.6	13.0	0.6	20.6
Poultry Manures	1.9	67.8	25.5	0.8	42.2
Farm-based Digestates	2.2	18.6	7.9	0.2	10.7
Total	**48.2**	**27.5**	**6.0**	**10.3**	**21.5**

6.1.1. Storage of solid manures in temporary field heaps

Storage of solid manures, including high RAN poultry manures in temporary field heaps is common practice, and permitted within the NVZ rules as long as the manures are solid enough to be stacked in a free-standing heap, and they do not give rise to free drainage from within the stack. Poultry manures without bedding/litter should also be covered with an impermeable sheet.

A Defra funded 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). Similarly for biosolids, Brettell et al. (2013) found that leachates from 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 solid manures 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 rules (and the Biosolids Assurance Scheme - BAS) 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 should minimise losses from field heaps within NVZs. Indeed, Williams et al (2015b) 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.

6.1.2. Slurry storage capacities

Current NVZ and SSAFOs (Silage, Slurry and Agricultural Fuel Oil) regulations state that any livestock slurry must be stored in a tank, lagoon or other suitable facility, with these stores having the capacity to store both the slurry and any rainfall/ wash water that enters during the storage period. SSAFO (2010) regulations require a minimum of 4 months' storage capacity, whereas NVZ regulations require stores to be large enough to enable 6 months (October to April) storage for pig slurries (and poultry manures) or 5 months storage (October to March) for other livestock manures. The EA's interpretation of Rule 1 of FRfW is likely to increase slurry storage requirements by two fold for pigs and for cattle housed all year and 50% for cattle housed during the winter. For example, where there is access to grass, it is likely that a minimum of 6 months storage will be required, however, where slurry is only applied to arable land at least 9 months storage is likely to be necessary. This will require significant capital investment on many farms. A slurry investment scheme is due to be offered by the UK Government from 2022, which aims to help farmers invest in new slurry stores and provide cover for existing stores in line with new regulations being implemented as part of the Clean Air Strategy (Defra, 2020). The initial focus for these grants will be on those locations where environmental impact will be greatest and for businesses seeking to invest in improved storage, so it will take time for the benefit of this scheme to be realised.

Nicholson et al (2011) estimated the cost of extending slurry storage capacity from the 2007 baseline (3 and 4 months storage capacity for cattle and pig slurries, respectively) to the current NVZ regulations (5 and 6 months storage, respectively), to be in the region £555 million if applied to the whole of England and Wales. It was considered this would be partly offset by savings in fertiliser usage (£104 million over a 20 yr period) and ecosystem damage costs (£259 million), with a cost-benefit ratio of *c.* 1.5:1. However, extending the closed periods further (by 1 or 2 months) did not result in proportional reductions in fertiliser use or ecosystem damage costs (cost-benefit ratio of 3.4:1 for whole of England and Wales).

6.2. Spreading organic materials

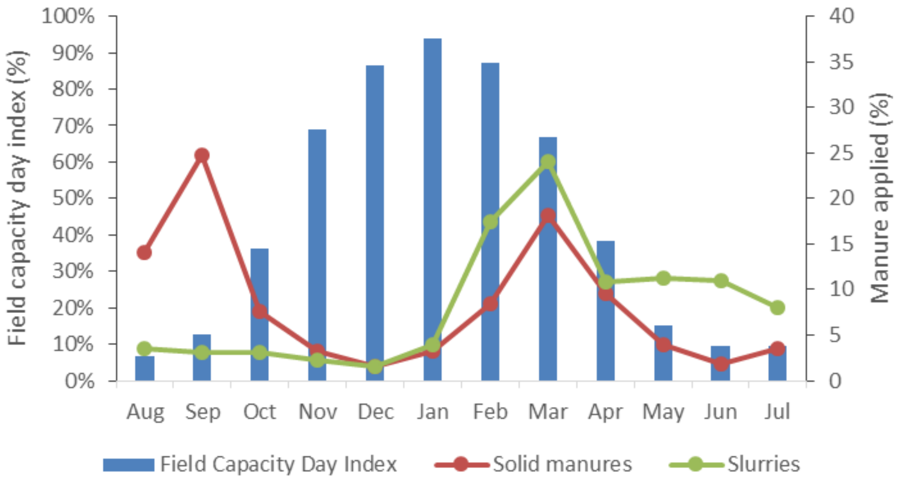
There are intrinsic differences between manufactured fertiliser and organic manures. Manufactured fertilisers typically contain nutrients which are in a highly available form (e.g. 100% water soluble). They also have a high nutrient density, in that they contain a high quantity of nutrient per tonne (e.g. ammonium nitrate is 34.5% N), and are manufactured in liquid or granule (prill) forms which can be easily stored and transported. These properties allow manufactured fertilisers to be top-dressed in spring using comparatively light machinery, from tramlines without driving over and damaging the growing crop. By contrast, organic materials, particularly low RAN FYM, biosolids and composts, contain nutrients in a less available form (<30% of total N is readily available) and have a low nutrient density (e.g. approximately 1% of the fresh weight is N). Solid manures are also bulky and require different equipment to spread them accurately which often cannot fit tramline spacings. For slurries, the use of band spreaders (compared to conventional 'splash plate' application) allows for more accurate top-dressing of materials across full tramline widths, without causing significant crop damage or contamination (as long as the crop is not too advanced). However, for both solid and liquid manures the weight of equipment is typically in excess of 20 tonnes. Therefore, ensuring that organic material applications are made to soils when they are strong enough to withstand the weight of the spreading equipment, is an important practical consideration. Travelling on 'wet' soils with heavy spreading equipment 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 (e.g. Figure 6-1). 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.

Figure 6-1 Soil damage following slurry spreading on 'wet' clay soils

The number of days per month a medium textured soil would be at or close to field capacity (SMD≤ 5mm) was estimated from 30 year average rainfall data (1981-2010). This was then 'weighted' according to how much manure was applied in the month, in order to derive a 'field capacity day index' – i.e. the proportion of days in a month when there is a risk of soil damage by, and runoff following, manure application. This index can also be used as an indication of the number of days available for spreading when soil conditions are suitable. The lowest risk (and hence highest number of 'spreading days') is in the summer and early autumn (May – September), which is when c.50% of solid manures and 35% of slurries are currently applied (Figure 6-2a). By contrast, in February and March, the soils are at field capacity for c.70-80% of the month and the number of available spreading days are lower, but c. 25% of solid manure and 40% of slurry applications already occur at this 'high risk' time (Figure 6-2a). The EA's interpretation of rule 1 of FRfW will lead to even greater pressure on the number of days available for safe spreading (i.e. where the risk of soil damage or run-off is low), at a time when there is also a higher risk of soils returning to field capacity shortly after application (Figure 6-2b). This pressure will vary across the country depending on rainfall, soil type, cropping and current manure application, with the greatest impact (change from baseline) seen in the dry arable areas of eastern England, where a large proportion of manures are currently applied at 'low risk times' in the autumn (Figure 6-3). Note, this index is based on average annual rainfall, and the risk will vary depending on the actual rainfall.

a) Baseline (current practice)



b) Scenario

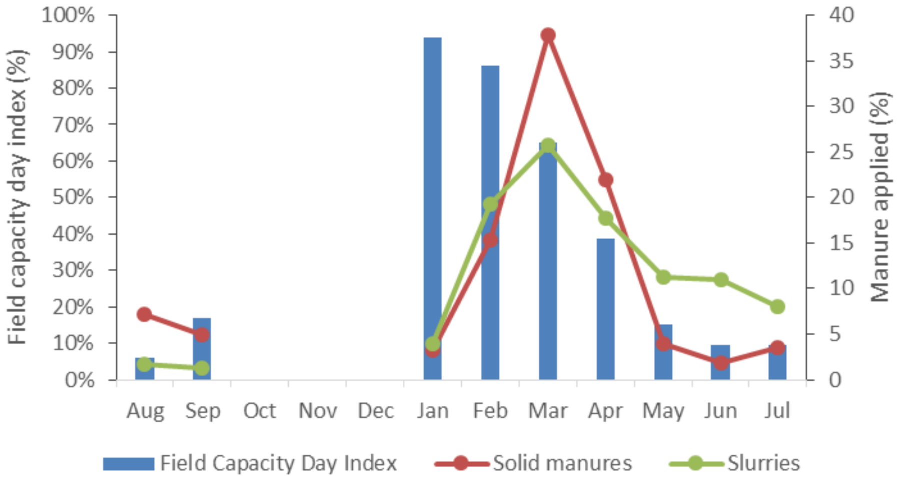


Figure 6-2 Volume-weighted proportion of days in a month where soils are at field capacity, compared to the distribution of manure application in England; a) baseline (current practice); b) Scenario (restricted autumn applications).

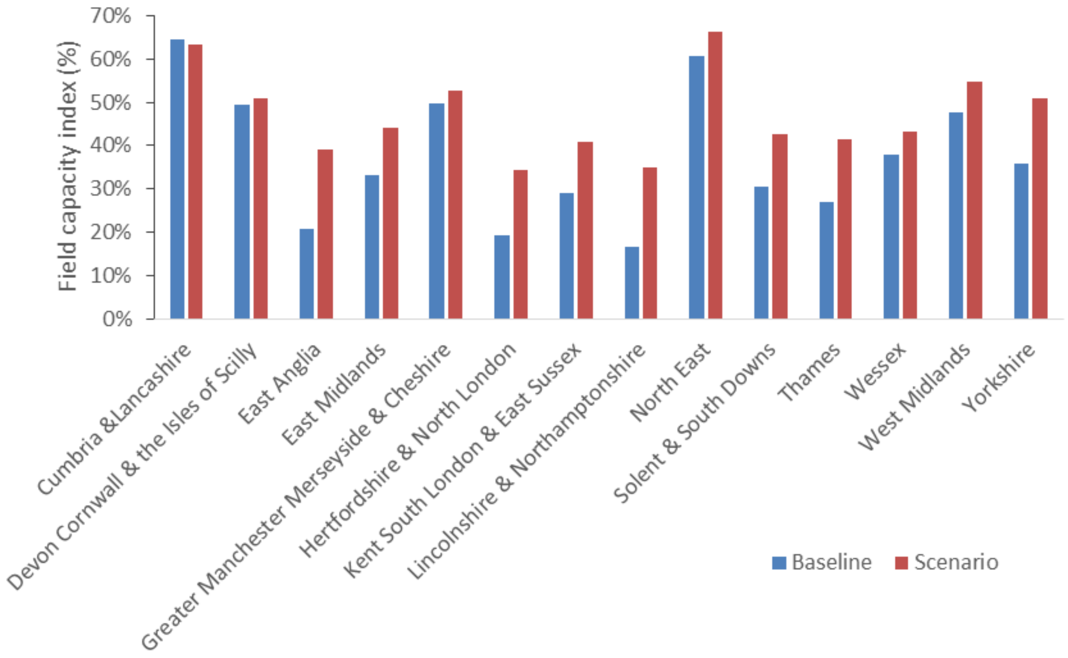


Figure 6-3 Average monthly manure volume weight field capacity index by English region (proportion of days in a month when a medium textured soil is at field capacity).

These results suggest that in most seasons it would not be possible to spread organic materials without significant risk of soil compaction and runoff until April in most English regions. This is likely to delay applications to winter cereal crops until stem extension which typically begins in early April. For slurry and liquid digestate applications made using band-spreading equipment, this may still practically be possible. However, for solid manures, the possible physical damage caused to the plants along with potential crop contamination issues is likely to make topdressing to cereal crops in spring impractical. Topdressing bulky organic materials to growing cereal crops is likely to result in reduced nutrient use efficiency; soil and crop damage; and reduced crop yield and quality compared with autumn applications.

Top-dressing organic materials 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 therefore be required, for example increasing the distance of applications from residential settlements or only applying to small land areas at any one time. Incorporating organic materials 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 these materials to cereal crops, public perception and stakeholder reaction may prevent the practice, particularly in the case of biosolids and food-based composts and digestates.

On grazed grassland, applications should be made at least three weeks prior to grazing to minimize the risk of transferring disease to grazing livestock. This also limits the time available for spreading in the spring; it could also coincide with when the cattle are being turned out for grazing, further limiting the land available for application.

7. Implications for the environment

7.1. Pollutant losses

Applications of organic materials pose a significant risk of diffuse and point source water and air pollution (Figure 7-1). The following sections give a brief review of the scientific evidence base relating to the potential for pollutant losses following organic material applications via these pathways, considering in particular how these loss routes are affected by different management practices (particularly timing of application), as well as soil, environment and climatic conditions.

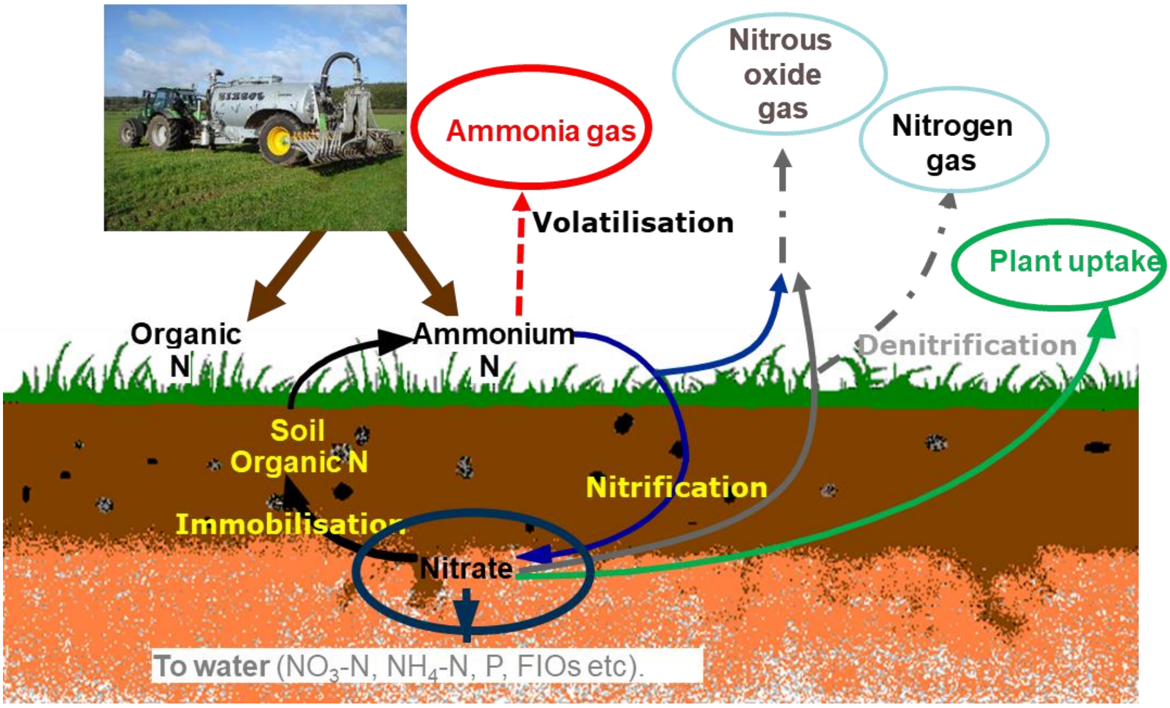


Figure 7-1. Nitrogen loss pathways following organic material application to land

7.1.1. Nitrate - N

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, poultry manures and digestates have 'high' RAN contents and typically contain c.35-80% of total N as RAN (Table 5.1). Cattle and pig FYM, biosolids and green/food composts have 'low' RAN contents and typically contain c.5-25% of total N as RAN, whereas green compost typically contain negligible RAN (<3% of total N; Table 5.1). A small proportion of the organic N applied with organic manures will also become available following application by the process of mineralisation (i.e. the 'mineralisable N' content). However, as the rate of mineralisation is temperature dependent, and drops to low levels at temperatures < 5 ºC, 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 crop establishment, 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. Effective rainfall will be greater in high rainfall areas than in low rainfall areas and following autumn than spring timings.
- ***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 rain 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 (a process known as 'bypass flow'), with transit times influenced by rainfall volume and intensity (Goss et al., 1983).

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 (Defra projects NT1402, NT1410 and OC896; Beckwith et al., 1998, Chambers et al, 2000). This has consistently shown nitrate leaching losses to be highest following applications of slurry and poultry manure which typically have RAN contents greater than 30% of total N. 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 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.

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, nitrate leaching losses from autumn applied pig and cattle FYM were very low (<1% of total N applied), reflecting the low proportion of manure total N present as RAN. 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, reflecting the high RAN content of these manures and low uptake of manure N by the winter wheat crop between application and the start of drainage. Nitrate leaching losses following 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.

The current NVZ manure and fertiliser application rules are based upon these studies, with closed spreading periods in place for organic materials with a high RAN content (>30% of total N content) which vary with soil type (earlier start and end date for shallow/sandy soils) and cropping (grass vs arable); Table 7.1. 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 and consequent risk of nitrate leaching is typically relatively low.

Table 7.1 Closed periods for spreading manure with readily available N contents greater than 30% of total N

	Grassland[b]	**Tillage land**[a,b]
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

[a] 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

[b] Outside these periods, no more than 30m^3/ha of slurry or 8 t/ha poultry manure can be applied in a single application from the end of the closed period until the end of February.

7.1.2. Ammonium-N

Ammonium-N (NH_4-N) losses are important in terms of water quality for fresh water fish, and the Freshwater Fish Directive (FWFD) has set mandatory threshold concentrations for total ammonium-N of 0.78mg/l and guide levels of 0.03mg/l and 0.16mg/l for Salmonid and Cyprinid fish, respectively (EC, 1978). However, NH_4-N is relatively immobile in soils due to adsorption onto the soil exchange complex, and also tends to be rapidly converted to nitrate N via the process of nitrification. Direct leaching of any ammonium-N compounds entering the soil from manures or ammonium-based fertilisers (including following the hydrolysis of urea) is therefore only likely when nitrification is delayed or significant heavy rainfall creating runoff or drainage occurs soon after application and surface runoff or by-pass flow occurs.

For example, concentrations of up to 5 mg/l NH_4-N have been measured in drainage waters following the application of ammonium nitrate fertiliser in both March and May 2000 to a drained clay soil at ADAS Boxworth in Cambridgeshire, and of up to 4.5 mg/l in surface runoff from a clay soil in Devon (Hatch et al., 2004). Similarly, Defra project NT2605 (Macdonald et al., 2006) measured elevated NH_4-N concentrations in drainflow (in excess of the FWFD) following spring applications of ammonium-based fertilisers and urea to both heavy textured drained soils, where water movement is largely via by-pass flow and to undrained heavy textured soils where water movement is largely by surface runoff/subsurface flow; whereas on free draining sandy soils (water movement by matrix flow), virtually 100% of the N leaching loss was in the form of nitrate-N.

Following manure applications, the research evidence in England is limited to data from Defra project WQ0118 'Cracking clays'. Here, elevated NH_4-N concentrations in drainage and surface runoff waters were measured following slurry application to grassland and arable soils with moisture deficits of less than 20 mm, where there was sufficient rainfall soon after application to cause drainflow/surface runoff. On grassland (Rowden, Devon), the highest NH_4-N concentrations were measured in October 2008 where cattle slurry was applied to soils with a moisture deficit of 15 mm and 46 mm of rainfall occurred in the 2 weeks after application; peak NH_4-N concentrations were 28.0 mg/l in drainage waters, 36.9 mg/l in surface runoff from the drained plots and 14.0 mg/l in surface runoff from the undrained plots (i.e. higher than the EC Freshwater Fish Directive limit of 0.78 mg/l NH_4-N). Moreover, at another grassland site (Faringdon in Oxfordshire), drainage waters were discoloured following the application of cattle slurry in March 2008 to soils with a moisture deficit of 7mm and 53mm of rainfall occurred in the 4-10 day period following application, indicating that slurry had moved rapidly from the grassland soil surface through macro-pores to the field drains, with little contact with the soil matrix (Figure 7-2). On arable land (Boxworth, Cambridgeshire), the highest NH_4-N concentrations (6.1 mg/l NH_4-N) were in February 2009 where pig slurry was applied to soils with a moisture deficit of 4 mm and 6 mm of rainfall (7 days after application) resulted in drainflow. Summer cattle slurry application timings had no effect on NH_4-N concentrations in drainage or surface runoff waters; reflecting warm soil conditions at the time of application (which would have encouraged the conversion of NH_4-N to nitrate-N) and low drainage volumes after application. Based on these findings simple risk management guidelines were produced which suggested that the risk of drainage and surface water pollution from NH_4-N (P and E.coli) was *lowest* when slurry was applied to soils with moisture deficits greater than 20mm (Table 7.2), which was usually in the autumn, late spring and summer.

Table 7.2. Risk management guidelines for slurry application timing

Soil moisture deficit (mm)	Risk
>20	Low
10-20	Moderate
<10	High

Figure 7-2 Drainage water samples 10 days after slurry application to grassland at Faringdon in March 2008: slurry treated (left) and untreated (right)

7.1.3. Phosphorus

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 organic material applications play an important role in maintaining the long-term soil fertility. 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.

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.

It is important to understand the forms of P present in organic materials 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).

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 applications has traditionally been based on avoiding N losses to water or the atmosphere rather than controlling P. For almost all crops, manure applications based on N requirements will supply more total P than is taken off in the crop. Ideally, manures 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 minimised. 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 method and timing: 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 organic materials 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.

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.

Effect of soil type: 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 immobilisation 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. 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.

Effect of rainfall: Applying manures 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. 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. Also, increasing the length of time between 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.

7.1.4. Ammonia

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, 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). Indeed, Defra project AC0111 (‘Cracking Clays – Air’) concluded that slurry composition and prevailing soil and weather conditions at the time of application were more important drivers of NH_3 losses than the date of application *per se.*

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. Because the RAN content of FYM is lower than slurry and poultry manure (Table 5.1) 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: EF ranged from 3-5%) than for food-based digestate and slurry (high RAN manures; EF ranged from 24 – 42%), across 3 sites and 2 application timings.

Effect of soil incorporation: 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. 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. Ploughing is the most effective technique, at reducing ammonia loss to 10% of that of surface broadcast manure for FYM and 5% for poultry manure. In the case of slurries and liquid digestate, precision application methods can reduce NH_3 emissions by 40-50% compared to broadcast treatments, with shallow injection more effective than bandspreading techniques such as trailing shoe (grassland) or trailing hose (arable land) e.g. Smith et al., (2000), Misselbrook et al. (2002), Nicholson et al., (2017). The recently introduced Clean Air Strategy (Defra 2019) recognises the importance of soil incorporation in reducing emissions and requires farmers to rapidly incorporate solid manures on tillage land, with all slurries and digestates to be applied using bandspreading or shallow injection equipment by 2025.

7.1.5. Nitrous oxide

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 that 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 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 organic manures and climatic conditions, or whether it would be possible to assign different EFs for different organic 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.

Effect of soil incorporation: Incorporation would be expected to reduce N losses from NH_3 volatilisation, 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 livestock 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.

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 livestock 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.

7.2. Impact of application timings on the balance of N losses to air and water – results of the modelling exercise

The loss of nitrate and phosphorus to water and ammonia and nitrous oxide emissions to the atmosphere as a consequence of baseline (current) farm manure application practices is given in Table 7.3. By far the greatest loss of nitrogen is via ammonia volatilisation to the atmosphere, comprising c.20% of the total ammonia emission from agricultural sources, with nitrate-N losses to water comprising about 2.5% of the total nitrate-N losses from agricultural sources. Total P losses amount to 0.7 kt or 15% of the total P loss from agriculture.

Table 7.3 Baseline N and P emissions from farm manure application compared to the total from agriculture in England

Loss pathway	Baseline (kt)	Total agricultural emission (kt)*	%
NH_3-N	30.6	132	23.3
NO_3-N	9.1	331	2.7
N_2O-N	1.3	18	7.2
Total P	0.7	4	15.2

*Total agricultural emission as reported in the 2019 Agricultural Ammonia and Greenhouse Gas Inventory (AAGHI) and the PSYCHIC model (2004).

As discussed in the sections above, the magnitude of losses will depend on the amount, timing and method of application, with soil type also important, particularly for nitrate and phosphorus losses to water. This can be clearly seen in Figure 7-3a&b, where NO_3-N losses are highest in the autumn on sandy soils (15-30% of N applied), compared to clay soils (c. 5% of N applied), whereas for P, losses are greater on clay soils in the winter and spring. There is little difference in the pattern of ammonia emissions between soil types, with method of application (particularly ability to incorporate/use of band-spreader) and hence crop-type (autumn-sown, spring-sown or grass) a key factor influencing the pattern of emissions. See Appendix Table 11.7 for a full breakdown of the baseline emission factors (by livestock manure type and season).

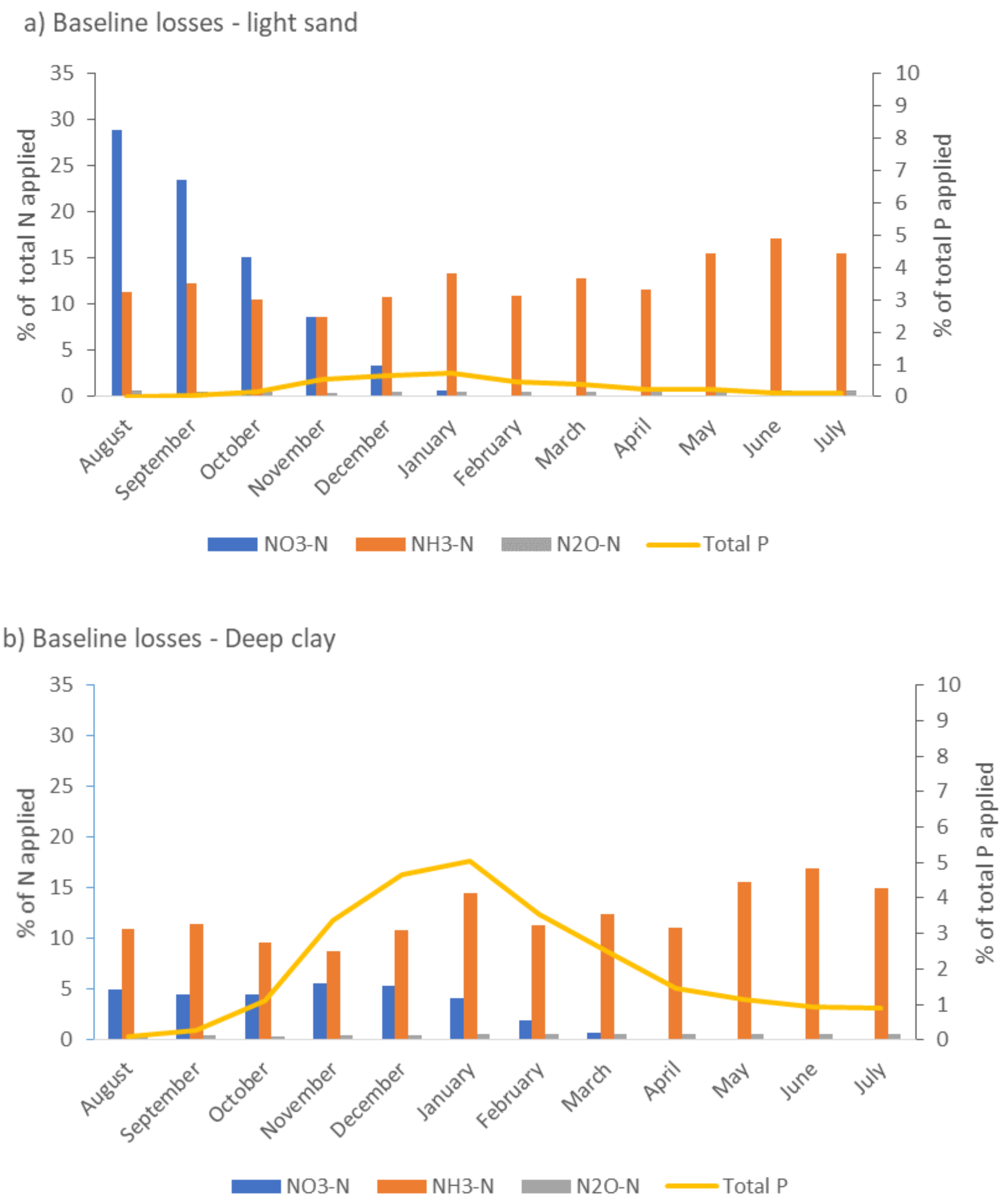


Figure 7-3 Timing of emissions following farm manure applications in England in relation to soil type; baseline (current) practice

Adoption of the optimised scenario outlined in section 5 would result in a *c*. 60% reduction in NO_3-N losses, but a 10% increase in NH_3-N emissions and 30% increase in P losses, with very little change in N_2O emission (Table 7.4). The change in emissions equate to a *c*. 1.5% decrease, but c. 2% and 5% increase in the total agricultural emission of NO_3-N, NH_3-N and P, respectively.

Table 7.4 Implications of a potential move from autumn to spring farm manure applications on the balance of N and P losses to air and water; total for all soil types and English regions

Loss pathway	Baseline	Optimised scenario*		Constrained scenario*	
	kt	kt	% change	kt	% change
NH_3-N	30.6	33.6	9.7	34.1	11.3
NO_3-N	9.10	3.80	-58.2	3.80	-58.3
N_2O-N	1.30	1.291	-1.0	1.288	-1.2
Total P	0.68	0.89	30.7	0.89	30.7

*Optimised scenario – all spring slurry applications applied using a band-spreader (or injection if that is current practice); Constrained scenario – current spreading practices maintained

The constrained scenario is more realistic of what the immediate consequence of the EA's interpretation of rule 1 of FRfW would be, as it is based on current spreading practices which are constrained by the availability of band-spreading equipment. As a result, the increase in NH_3-N emissions following a move to spring applications, is greater under this scenario. This can be clearly seen in Table 7.5 with the optimised scenario showing a reduction in NH_3-N emissions for cattle and pig slurry compared to the constrained scenario reflecting the 100% adoption of band-spreading in the spring (which is not current practice). By comparison if the method of application is maintained (constrained scenario), NH_3-N emissions are greatest. This is particularly the case for pig slurry where currently (baseline) 35% is broadcast and incorporated ahead of autumn sown cereals and there is no opportunity to incorporate following a spring application to a growing crop. Table 7.6 also demonstrates that the impact of a change in application timings is greatest for pig FYM and poultry manures, which are commonly applied in the autumn and incorporated, with a move to spring applications resulting in an 80% and 60% decrease in NO_3-N losses, respectively, but a 1.5 fold and 7 fold increase in P losses, respectively. As a large proportion of pig and poultry units are in the East of England, the greatest change in the balance of losses is seen in these regions (East Anglia, Lincolnshire and Hertfordshire; Table 7.6).

Table 7.5 Percentage change in emissions following different farm manure applications as a result of adopting the optimised ('Opt.') and constrained ('Con.') scenarios.

Organic material	NH_3-N		NO_3-N		N_2O-N		Total P	
	Opt.	Con.	Opt.	Con.	Opt.	Con.	Opt.	Con.
Cattle Slurry	-3	0.2	-41	-41	-0.5	-1	-2	-2
Pig Slurry	0	5	-60	-61	-3	-2	31	31
Manure-based digestate	2	2	-41	-41	-1	-1	14	14
Poultry manures	29	29	-64	-64	1	1	157	157
Cattle FYM	6	6	-68	-68	-6	-6	27	27
Pig FYM	15	15	-83	-83	-10	-10	745	745

Table 7.6 Percent change in emissions (relative to the current baseline) by English region as a result of a shift to spring manure application timings (optimised scenario).

Public Face Area	% change in emission (optimised scenario)			
	NH_3-N	NO_3-N	N_2O-N	Total P
Cumbria & Lancashire	1.8	-46.8	-1.1	1.8
Devon Cornwall & the Isles of Scilly	2.5	-44	-1.4	7.6
East Anglia	23.2	-72.8	-0.8	220
East Midlands	10.4	-56.1	-0.5	39.5
Greater Manchester Merseyside & Cheshire	1.3	-61.2	-0.9	10
Hertfordshire & North London	19	-70	0	115
Kent South London & East Sussex	16.2	-63.9	-0.2	53.8
Lincolnshire & Northamptonshire	23.9	-60	0.7	198
North East	7.9	-57.2	-0.8	34.2
Solent & South Downs	12.7	-60.9	-2.2	50.7
Thames	13.5	-60.5	-1.2	75.1
Wessex	4.3	-56	-1.4	17.8
West Midlands	10.3	-57.5	-0.2	36.3
Yorkshire	11.5	-63.2	-1.9	55.2

7.3. Soil health and carbon

Soil organic matter (SOM) is a key indicator of soil health. SOM provides a food source and habitat for the soil biological community, drives the cycling of nutrients within soils and is a central component of soil aggregation and the maintenance of structure and water relations, it is also an important carbon store (Kibblewhite et al. 2008). 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). It is widely recognised that the recycling of organic materials to agricultural land is one of the most effective ways of increasing SOM levels (Bhogal et al. 2017). Indeed, the benefit of a range of organic material applications (livestock manures, composts, biosolids) for SOM and soil quality has been widely documented and reviewed (e.g. Edmeades, 2003, Johnston et al. 2009). Studies have evaluated the potential of organic materials as nutrient sources (Schroder et al. 2005) and soil conditioners (Diacono & Montemurro, 2010), as well as a means to sequester carbon in the mitigation of climate change (Powlson et al. 2012). The recycling of organic materials to land therefore contributes to nutrient management and circular economy policy objectives, completing natural nutrient and carbon cycles.

Changing the timing of organic material applications (from autumn to spring) would probably have a minimal effect on their impact on SOM and associated soil improvement. However, if the change in the timing of organic material applications reduces the quantities of organic manures applied to agricultural soils, this will negatively affect soil health and C storage in soils. Moreover, as discussed in section 6.2, if materials are applied when soils are close to field capacity (which is more likely with spring applications), resultant soil structural damage is likely to reverse any potential soil conditioning benefit of the application.

8. Alternative scenarios

8.1. Change farm practice

8.1.1. Restrict applications to spring crops/increase spring cropping area

The predicted increase in ammonia emissions to air and P losses to water as a result of the management changes following the EA's interpretation of Rule 1 of FRfW (Section 7.2) largely reflects the inability to rapidly incorporate manures when they are top-dressed onto established autumn sown crops in the spring. As can be seen in Figure 8-1a, currently c. 95% of solid livestock manures are incorporated by plough, tine or disc following application. A move to predominantly spring applications to the same crop type is predicted to reduce the amount of manure incorporated to less than 50% (Figure 8-1b).

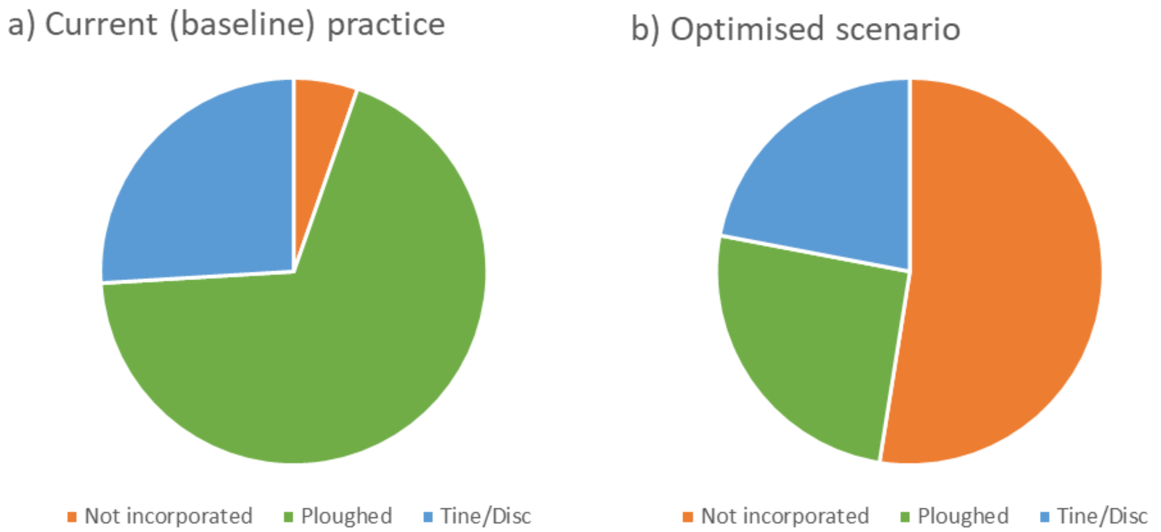


Figure 8-1 Change in method of incorporation as a result of moving to predominantly spring solid manure applications (optimised scenario – 9.8MT solid manure applied across all English regions).

As discussed in section 6 it is only likely to be practical to apply organic manures in spring before the establishment of spring sown crops on light free draining soils as they pose the lowest risk of soil compaction in spring. An increase in spring cropping will also result in increased bare land over winter (unless cover crops are employed) and potentially more spring-sown, late harvested crops (e.g. maize), with associated increased risk of nutrient and sediment losses and soil damage post-harvest. Spring cereals also have a lower gross-margin than winter cereals.

8.1.2. Out-winter animals/increase number of outdoor units

In order to reduce the amount of livestock manure that needs to be stored over winter, farmers could move to out-wintering of animals and increase outdoor pig production. Although this would decrease the amount of 'managed' manure, it would not decrease the overall nutrient loading to soils and the environment. Indeed, this is likely to result in an increase in soil damage and nutrient losses over the winter period. For example, Webb et al. (2005) concluded that although extending the winter grazing season could potentially reduce ammonia emissions (by c. 10% from one month's extra grazing, assuming animals are outside for all the month, day and night), most of the conserved NH_3-N was predicted to be lost by nitrate leaching; for slurry based systems at least 80% of the N conserved could be lost via leaching, for FYM-based systems, the increase in nitrate leaching was always greater than the reduction in ammonia emissions. Moreover, Newell Price et al. (2011) estimated particulate/soluble P and sediment losses would also increase (by up to 10%) as a result of greater poaching damage.

8.2. Organic manure processing to reduce volume and nutrient content

8.2.1. Slurry separation

Mechanical separation of slurry to take out the coarse solid and fibre fraction enables better handling of the liquid fraction. The solid fraction (typically 10-20% of the original slurry volume) can then be stacked, stored and spread in a similar way to FYM, and the liquid fraction spread using band-spreading equipment without causing any blockage problems. However, adoption of this practice would result in an increase in the amount of solid manure to be applied to agricultural land – which is where the impact of a change to spring applications is greatest (not with-standing the greater infrastructure required).

8.2.2. Composting

Compost is made from the controlled biological decomposition of organic materials under aerobic conditions (WRAP, 2016). It produces a stable, sanitised soil-like material which is typically used as a soil conditioner and source of plant nutrients (particularly phosphate and potash). Composting livestock manures will reduce both the volume and RAN content, creating a more stable, consistent material for spreading in the autumn. However, composting manures (without a management system for the gases produced) often results in higher cumulative GHG and ammonia emissions (e.g. Parkinson et al., 2004; Ahn et al. 2011), and also requires infrastructure (concrete pads) and energy (for turning).

Note that all other manure processing technologies (the 'bio-refinery concept' which includes various technologies under development to recover nutrients, chemicals and energy from organic materials e.g. WRAP 2020) are very much in the early stages of development or are too costly to be feasibly practical on a large scale.

8.3. Incineration

Whilst the application of organic materials to land leads to diffuse N and P losses, it does complete natural nutrient and OM cycles and minimises the loss of these valuable resources, contributing to UN Sustainable Development Goals, reducing the need for manufactured fertilisers for optimum crop production and improving soil health and carbon storage. Although incineration has been successfully employed for some poultry manures, these have a high dry matter content. The infrastructure required, logistics and cost of implementing this option is likely to be prohibitive.

9. Overall discussion and conclusions

All agricultural land management can cause pollution of the air and water environments. For example, Goulding et al. (2000) measured nitrate losses from the long-term Broadbalk experiment and saw that even where no fertiliser N had been applied since 1843, c. 10 kg/ha NO_3-N was lost over winter (from the mineralisation of soil organic N), increasing to c. 30 kg/ha at the optimum N fertiliser application rate. Indeed, by far the greatest proportion of NO_3-N leached from agricultural soils in England is from the mineralisation of native soil organic N and crop residues/residual soil N post-harvest, with managed manures currently only contributing to c. 3% of the loss (Table 7.3).

It is recognised that manure management practices that aim to reduce one form of pollution (e.g. nitrate to water) should not exacerbate losses by another route - so called ‘pollution swapping’ (Newell-Price et al. 2011). However, it can be seen from the results of the modelling exercise that a move to predominantly spring applications as a result of the EA’s interpretation of Rule 1 of FRfW, although decreasing nitrate leaching losses by *c.*60%, are likely to increase ammonia emissions by *c.*10% and P losses by *c.*30%. The risk of soil damage as a result of applications to wet soils is also greater in the spring. The impact would be greatest for bulky, low RAN materials that are currently incorporated in the autumn ahead of winter crop establishment, particularly pig and poultry manures, and predominantly for the arable regions in the East of England where these materials are applied. These results have considerable implications for the successful implementation of the Clean Air Strategy and UK Governments ammonia emission reduction targets (i.e. to reduce emissions by 16% in 2030 compared to 2005 levels, in order to meet the National Emissions Ceilings Directive and Gothenburg Protocol target). Moreover, the potential for increased P loss in the spring has implications for compliance with the Water Framework Directive. Indeed, as well as considering the total nutrient loss, it is important to consider the concentration in the drainage water. While it is likely that dilution (and transformation) of N and P in the drainage water leaving a field will reduce the impact on receiving waters, this is dependent on the amount, duration and intensity of rainfall (Goulding et al. 2000). Therefore the risk of non-compliance of water bodies is likely to be in the East of England, with the lowest annual rainfall totals and potential for dilution.

9.1. Draft matrix on the responsible application of organic materials

This impact assessment has shown that the effective management of organic materials to minimise impacts on the environment and farm practice needs to consider the ‘balance of risks’, taking into account not only the type of organic material and when it is applied, but how it is applied (e.g. can it be incorporated to minimise NH_3 emissions and P loss) and where it is applied (soil type and cropping). Light textured soils present the greatest risk of nitrate leaching, and ‘best’ opportunity for travelling in the spring, whereas clay and medium soils present the greatest risk of P losses and soil damage in the spring.

Based on these findings, a matrix on the responsible application of organic materials to agricultural land has been drafted, which aims to guide the industry on when and where a range of different organic materials can most effectively be used in a way where risks to soils, crops, the environment and farm businesses are minimised (Table 9.1).

Table 9.1 Draft matrix on the responsible use of organic materials

Soil type	Light						Medium/heavy					
Organic material (RAN content)	**Low**			**High**			**Low**			**high**		
Application timing	**Autumn**	**Winter**	**spring**	**Autumn**	**Winter**	**spring**	**Autumn**	**Winter**	**spring**	**Autumn**	**Winter**	**spring**
Spreading days	high	low	high	high	low	high	high	low	low	high	low	low
Storage requirements	low	medium	high	low	medium	high	low	medium	high	low	medium	high
Water:												
Nitrate-N	**	*	*	***	**	*	*	*	*	**	**	*
Phosphorus	*	**	**	*	**	**	*	***	***	*	***	***
Air:												
Ammonia - grass	**	**	**	***	***	***	**	**	**	***	***	***
Ammona - arable	*	**	**	*	***	***	*	**	**	*	***	***
Nitrous oxide	*	*	*	*	*	*	*	*	*	*	*	*
Soils:												
Compaction risk	*	**	**	*	**	**	*	***	***	*	***	***

Soil type:

Light: Sand, loamy sand, sandy loam & shallow soils

Medium/heavy: all other soil types (excluding organic/peat soils)

Application timing:

Autumn: August, September & October applications

Winter: November, December & January applications

Spring: February, March & April applications

Risk of losses/compaction:

*** low**

**** Medium**

***** High**

9.2. Knowledge gaps

This impact assessment has not considered the financial implications of the EA's interpretation of Rule 1 of FRfW in terms of cost (££) to farm businesses and organic material producers as well as the value of impacts to the environment (i.e. using an Enabling Natural Capital Approach – ENCA; Defra, 2020b).

9.3. Conclusions

A large proportion (up to 70%) of solid, low RAN materials are currently applied and incorporated in the autumn ahead of autumn-sown cereals. The modelling undertaken as part of this study has shown that the EA's interpretation of Rule 1 of the FRfW will:

- reduce nitrate leaching losses by c. 60% (1.5% decrease in the total loss from agriculture)
- increase in ammonia emission by c. 10% (2% increase in total emissions from agriculture)
- increase in P loss by c. 30% (5% increase in the total loss from agriculture)
- increase solid manure storage requirement by 7 million tonnes and slurry by 3 million m^3

The increase in ammonia emissions and P losses is largely due to the inability to incorporate solid manures in spring, which is an important mitigation method for controlling loss of N by ammonia volatilisation and P via surface runoff. P losses are also likely to be higher from spring applications as soils are usually closer to field capacity than in autumn increasing the risk of surface runoff after application. For livestock manures, these impacts are likely to be greatest in the East of England where most pig and poultry manures are currently applied ahead of autumn cropping.

The FRfW aim to ensure that 'all reasonable precautions' are taken to prevent diffuse pollution following the application of organic manures and manufactured fertilisers, stating that materials should not be applied 'if there is a significant risk of agricultural diffuse pollution'. This impact assessment has shown that the effective management of organic materials needs to consider the 'balance of risks' to the water, air and soil environments, as well as practical considerations. It is important to take into account not only the type of organic material and when it is applied, but how and where it is applied. Light textured soils present the greatest risk of nitrate leaching, and 'best' opportunity for travelling in the spring, whereas clay and medium soils present the greatest risk of NH_3-N emissions, P losses and soil damage in the spring. Clay and medium soil types also have more limited opportunities for spring cropping (and hence the potential for soil incorporation). There are also limited options for further treatment or sustainable alternative uses of these materials if spreading to agricultural land is prohibited.

10. References

Addiscott, T. (1977) A simple computer model for leaching in structured soils. Journal of Soil Science, 28, 554-563.

Addiscott, T. and Whitmore, A. (1991) Simulation of solute leaching in soils of differing permeabilities. Soil Use and Management, 7, 2, 94-100.

Ahn, H. K., Mulbry, W., White, J. & Kondrad, S. (2011) Pile mixing increases greenhouse gas emissions during composting of dairy manure. Bioresour. Technol. 102, 2904–2909.

AFRC (1993) Energy and Protein Requirements of Ruminants. CAB International, 139 pp.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., (1998) Crop Evapotranspiration: Guidelines for Computing Crop Water Requirements. United Nations Food and Agriculture Organization, Irrigation and Drainage Paper 56. Rome, Italy, 300 pp.

Alley, W. M. (1984) On the treatment of evapotranspiration, soil moisture accounting and aquifer recharge in monthly water balance models. Water Resources Research, 20(8), 1137-1149.

Anon (2010). The fertiliser manual (RB209). The Stationery Office, Norwich.ISBN 978-0-11-243286-9

Anthony, S. (2003) The IE model: an event based calculation of rainfall infiltration excess. Unpublished, ADAS internal document.

Anthony, S. (2003) The MCDM model: a monthly calculation of water balance for use in decision support systems. Unpublished, ADAS internal document.

Beckwith, C.P, Cooper, J., Smith, K.A. & Shepherd, M.A. (1998). Nitrate leaching loss following application of organic manures to sandy soils in arable cropping. I. Effects of application time, manure type, overwinter crop cover and nitrification inhibition. Soil Use Manage. 14, 123–130.

Bhogal, A., Williams, J. R., Nicholson, F. A., Chadwick, D. R., Chambers, K. H. & Chambers, B. J. (2016). Mineralisation of organic nitrogen from farm manure applications. Soil Use Manage, 32 (Supplement 1), 32-43.

Bhogal A., Nicholson F.A., Rollett, A., Taylor, M., Litterick, A., Whittingham, M. & Williams, J.R. (2018) Improvements in the quality of agricultural soils following organic material additions depend on both the quantity and quality of the materials applied. Frontiers in Sustainable Food Systems, Waste Management in Agroecosystems 19 April 2018 https://doi.org/10.3389/fsufs.2018.00009

Bhogal, A., White, C. & Morris, N. (2020) Maxi Cover Crop: Maximising the benefits from cover crops through species selection and crop management. Project Report No. 620; AHDB Cereals & Oilseeds.

Boorman, D., Hollis, J. and Lilly, A. (1995). Hydrology of Soil Types: A Hydrologically based Classification of the Soils of the United Kingdom. Institute of Hydrology, Natural Environment and Research Council, Wallingford, 146 pp.

Bourdin, F., Sakrabani, R., Kibblewhite, M. G. & Lanigan, G. J. (2014). Effect of slurry dry matter content, application technique and timing on emissions of ammonia and greenhouse gas from cattle slurry applied to grassland soils in Ireland. Agric. Ecosyst. Environ. 188, 122-133.

Bretell, N., Taylor, M. and Chambers, B. (2013). Pollutant Losses From Biosolids Stored In Temporary Field Heaps.

Brown, P., Broomfield, M., Cardenas, L., Choudrie, S., Jones, S., Karagianni, E. et al. (2019). UK Greenhouse Gas Inventory, 1990-2017: Annual Report for submission under the Framework Convention on Climate Change. Ricardo-AEA, Didcot, UK, April 2019.

Catt, J.A., Howse, K.R., Farina, R., Brockie, D., Todd, A., Chambers, B.J., Hodgkinson, R., Harris, G.L. & Quinton, J.N. (1988). Phosphorus losses from arable land in England. Soil Use Manage. 14, 168-174.

Cayuela, M. L., Aguilera, E., Sanz-Cobena, A., Adams, D. C., Abalos, D., Barton, L. et al. (2017). Direct nitrous oxide emissions in Mediterranean climate cropping systems: Emission Factors based on a meta-analysis of available measurement data. Agric. Ecosys. Environ. 238, 25-35.

Chadwick, D.R., Pain, B.F. & Brookman, S.K.E. (2000) Nitrous oxide and methane emissions following application of animal manures to grassland. J. Environ. Qual. 29, 277-287.

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L, Amon, B. et al. (2011). Manure management: Implications for greenhouse gas emissions. Anim. Feed Sci. Technol. 166– 167, 514– 531.

Chalmers, A. (2001) A review of fertiliser, lime and organic manure use on farm crops in Great Britain from 1983 to 1987. Soil Use and Management, 17, 4, 254-262.

Chalmers, A., Kershaw, C. and Leech, P. (1990) Fertiliser use on farm crops in Great Britain: results from the Survey of Fertiliser Practice, 1969 to 1988. Outlook on Agriculture, 19, 4, 269-278.

Chambers, B. J., Lord, E. I., Nicholson, F. A. & Smith, K. A. (1999). Predicting nitrogen availability and loses following applications of organic manures to arable land: MANNER. Soil Use Manage. 15, 137-143.

Chambers, B. J., Smith, K. A. & Pain, B. F. (2000). Strategies to encourage better use of nitrogen in animal manures. Soil Use Manage. 16, 157-161.

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A. Chantigny, M. H. & Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. Agric. Ecosys. Environ. 236, 88-98.

Cottrill, B. and Smith, K. (2006) Nitrogen and phosphorus output of livestock excreta. Defra project WT0715NVZ, Final Report,

Davison, P., Withers, P., Lord, E., Betson, M. and Stromqvist, J. (2008) PSYCHIC – A process based model of phosphorus and sediment mobilisation and delivery within agricultural catchments. Part 1. Model description and parameterisation. Journal of Hydrology, 350, 290-302.

Defra (2018). Farming rules for water – getting full value from fertilisers and soil. Policy paper. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/695598/farming-rules-for-water-policy-paper-v2.pdf

Defra (2019) Clean Air Strategy 2019. https://www.gov.uk/government/publications/clean-air-strategy-2019

Defra (2020) Farming is Changing. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/986879/farming-changing.pdf

Defra (2020b) Enabling a Natural Capital Approach: Guidance. Crown Copyright 2020. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/869801/natural-capital-enca-guidance_2_March.pdf

Diacono, M., Montemurro, F., (2010). Long-term effects of organic amendments on soil 517 fertility. A review. Agron. Sustain. Dev. 30, 401–422.

EC (1978) Council Directive on the quality of fresh waters needing protection or improvement in order to support fish life. Council directive 78/659/EEC.

Edmeades, D.C. (2003). The long-term effects of manures and fertilisers on soil productivity 522 and quality: a review. Nutr. Cycl. Agroecosyst. 66, 165-180.

Eghball, B., Binford, G.D. & Baltensperger, D.D. (1996). Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. J. Environ. Qual. 25, 1339-1343.

Elliott, H.A., Brandt, R.C. & O'Connor, G.A. (2005). Runoff phosphorus losses from surface-applied biosolids. J. Environ. Qual. 34, 1632-1639.

Firestone, M. K. & Davidson, E. A. (1989). Microbiological basis of NO and N2O production and consumption in soil. In: Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere, eds. M. O. Andeae and D. S. Schimel (Chichester: Wiley), 7–21.

Flynn, N.J. & Withers, P.J.A. (2007). Application of phosphorus in biosolids to agricultural soils. UKWIR Report Ref. No. 07/SL/02/7.

Frossard, E., Sinaj, S., Zhang, L.M. & Morel, J.L. (1996). The fate of phosphorus in soil plant systems. J. Soil Sci. Soc. America, 60, 1248-1253.

Gooday, R.D., Anthony, S.G. & Fawcett, L.E. (2008). A field scale model of soil drainage and nitrate leaching for application in Nitrate Vulnerable Zones. Environmental Modelling & Software, 23, 1045–1055.

Gooday RD, Anthony SG, Chadwick DR, Newell-Price P, Harris D,Deuthmann D, Fish R, Collins AL, Winter M (2014) Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. Science Total Environment 468–469:1198–1209

Goss, M.J., Harris, G.L. and Howse, K.R., (1983). Functioning of mole drains in a clay soil. Agricultural Water Management, 6, 27-30.

Goulding, K.W.T., Poulton, P.R., Webster, C.P. and Howe, M.T. (2000). Nitrate leaching from the Broadbalk Wheat Experiment, Rothamsted, UK, as influenced by fertiliser and manure inputs and the weather. Soil Use and Management 16 244-250.

Green, W. and Ampt, G. (1911) Studies of soil physics, part 1 – the flow of air and water through soils. Journal of Agricultural Science, 4, 1-24.

Hart, M.R., Quinn, B.F. and Nguyen, M.L. (2004). Phosphorus runoff from agricultural land and direct fertilizer effects: a review. Journal of Environmental Quality, 33, 1954-1972.

Hatch, D.J., Chadwick, D.R., Smith, K.A. and Chambers, B.J. (2004). Ammonium-N Losses from Agriculture. In: Agriculture and the Environment – Water Framework Directive and Agriculture. Proceedings of the SAC/SEPA Biennial Conference. (Eds. D. Lewis and L. Gairns), SAC, Edinburgh. pp 165-173.

Hollis, D. and McCarthy, M. (2017) UKCP09: Met Office gridded and regional land surface observation datasets. Centre for Environmental Data Analysis, May 2021, http://catalogue.ceda.ac.uk/uuid/87f43af9d02e42f483351d79b3d6162a

Hough, M. N. and Jones, R. J. A. (1997) The United Kingdom Meteorological Office rainfall and evaporation calculation system: MORECS version 2.0 - an overview. Hydrol. and Earth Sys. Sci., 1(2), 227-239.

IPCC (2019). Chapter 11: N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application. In: Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, eds, K. Hergoualc'h, H. Akiyama, M. Bernoux, N. Chirinda, A. del Prado, Å. Kasimir et al., https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch11_Soils_N2O_CO2.pdf [accessed 20/11/19].

Johnston, A.E., Poulton, P.R., Coleman, K. (2009). Soil organic matter: Its importance in sustainable agriculture and carbon dioxide fluxes. Adv. Agron. 101, 1-57.

Jones, S.K., Rees, R.M., Skiba, U.M. & Ball, B.C. (2005). Greenhouse gas emissions from a managed grassland. Global Planetary Change 47, 201-211.

Jones, S.K., Rees, R.M., Skiba, U.M. & Ball, B.C. (2007). Influence of organic and mineral N fertiliser on N2O fluxes from a temperate grassland. Agr. Ecosyst. Environ., 121, 74-83.

Kibblewhite MG, Ritz K, Swift MJ. (2008). Soil health in agricultural systems Phil. Trans. R. 538 Soc. B 363, 685-701.

Kleinman, P.J.A., Bryant, R.B., Reid, W.S., Sharpley, A.N. and Pimentel, D. (2000). Using soil phosphorus behaviour to identify environmental threshold. Soil Science, 165, 943-950.

Kleinman, P.J.A., Sharpley, A.N., Wolf, A.M., Beegle, D.B. and Moore, P.A. (2002). Measuring water-extractable phosphorus in manure as an indicator of phosphorus in runoff. Journal of Soil Science Society of America, 66, 2009-2015.

Komiskey, M.J., T.D. Stuntebeck, D.R. Frame, and F.W. Madison. (2011). Nutrients and sediment in frozen-ground runoff from no-till fields receiving liquid-dairy and solid-beef manures. Journal of Soil and Water Conservation. 66, 303-312.

Lal, R. (2016). Soil health and carbon management. Food Energy Secur. 5, 212-222.

Lazcano, C., Tsang, A., Doane, t. A., Pettygrove, G. S., Horwath, W. R. & Burger, M. (2016). Soil nitrous oxide emissions in forage systems fertilized with liquid dairy manure and inorganic fertilizers. Agric. Ecosyst. Environ. 225, 160-172.MAFF (1988) Agricultural Land Classification of England and Wales. HMSO, London, 60 pp.

MAFF (1988) Agricultural Land Classification of England and Wales. HMSO, London, 60 pp.

Macdonald, A., Goulding, K., Bhogal, A., Nicholson, F., Chambers, B., Sagoo, L., Dixon, L. and Hatch, D. (2006). Defra project NT2605 Final Report WP4: Nitrogen losses to surface and ground waters.

Mkhabele, M. S., Gordon, R., Burton, D., Madani, A., Hart, W. & Elmi, A. (2006). Ammonia and nitrous oxide emissions from two acidic soils of Nova Scotia fertilized with liquid hog manure mixed with or without dicyandiamide. Chemosphere 65, 1381–1387.

Misselbrook, T.H., Shepherd, M.A. & Pain, B.F. (2006) Sewage sludge applications to grassland: influence of sludge type, time and method of application on nitrate leaching and herbage yield. J. Agr. Sic. Cam. 126, 343-352.

Misselbrook T.H., Smith K.A., Johnson R.A. and Pain B.F. (2002) Slurry application techniques to reduce ammonia emissions: results of some UK Field-Scale Experiments. Biosyst Eng 81(3):313–321

Montes, F., Meinen, R., Dell, C., Rotz, A., Hristov, A. N., Oh, J. et al. (2013). SPECIAL TOPICS — Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J. Anim. Sci. 91, 5070–5094.

Nair, V.D., Portier, K.M., Graetz, D.A. and Walker, M.L. (2004). An environmental threshold for degree of phosphorus saturation in sandy soils. Journal of Environmental Quality, 33, 107-113.

Newell Price, J.P., Harris, D., Chadwick, D.R., Misselbrook, T.H., Taylor, M., Williams, J.R., Anthony, S.G., Duethmann, D., Gooday, R.D., Lord, E.I. and Chambers, B.J. (2011) Mitigation Methods – User Guide. An Inventory of Mitigation Methods and Guide to their Effects on Diffuse Water Pollution, Greenhouse Gas Emissions and Ammonia Emissions from Agriculture. Prepared as part of Defra project WQ0106. 158pp.

Nicholson F.A., Williams J., Thorman R., Gooday R., Harris D., Chambers B., Chadwick D., Doody D. and Foy B. (2011) Pollutant losses following organic manure applications in the month following the end of the closed period. Final report to Defra for project WT0932. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/82413/20111220nitrates-directive-consult-evid6.pdf

Nicholson, F.A., Bhogal, A., Chadwick, D., Gill, E., Gooday, R.D., Lord, E., Misselbrook, T., Rollett, A.J., Sagoo, E., Smith, K.A., Thorman, R.E., Williams, J.R. & Chambers, B.J., (2013). An enhanced software tool to support better use of manure nutrients: MANNER-NPK. Soil Use Manage. 29, 473-484

Nicholson, F., Williams, J. and Sagoo, L. (2016) Pollutant losses from solid manure applications and from solid manures stored in temporary field heaps. Work Package 6: Assessment and update of livestock N and P excretion values. Defra project WT1568, Final Report, 74 pp.

Parkinson, R., Gibbs, P., Burchett, S. & Misselbrook, T. Effect of turning regime and seasonal weather conditions on nitrogen and phosphorus losses during aerobic composting of cattle manure. Bioresour. Technol. 91, 171–178 (2004).

Powlson, D.S., Bhogal, A., Chambers, B.J., Coleman, K., Macdonald, A.J., Goulding, 581 K.W.T. et al. (2012). The potential to increase soil carbon stocks through reduced tillage or 582 organic material additions in England and Wales: A case study. Agri. Ecosyst. Environ. 146, 583 23-33.

Schhroder, J.J., Jansen, A.G., Hilhorst, G.J. (2005). Long-term nitrogen supply from cattle 606 slurry. Soil Use Man. 21, 196-204.

SI (2015). The Nitrate Pollution Prevention Regulations. No. 668. http://www.legislation.gov.uk/uksi/2015/668/contents

SI (2017) The Water Environment (Water Framework Directive) (England and Wales) Regulations 2017. No.407 https://www.legislation.gov.uk/uksi/2017/407/made

SI (2018). The Reduction and Prevention of Agricultural Diffuse Pollution (England) Regulations 2018. No. 151. https://www.legislation.gov.uk/uksi/2018/151/contents/made

Silgram, M., Collins, A.L. and Stevens, C. (2008). Practical mitigation options for phosphorus and sediment loss from winter-sown crops: an overview of contemporary methods. Appendix 4. Final report for Defra project PE0206.

Silgram, M., Jackson, D.R., Bailey, A., Quniton, J. and Stevens, C (2010). Hillslope scale surface runoff, sediment and nutrient losses associated with tramline wheelings. Earth Surface Processes and Landforms 35 699-706

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

Smith, K.A., Jackson, D.R. & Withers, P.J.A. (2001). Nutrient losses by surface runoff following the application of organic manures to arable land. 2. Phosphorus. Environ. Pollut. 112, 53-60.

Smith K.A., Jackson D.R., Misselbrook T.H., Pain B.F., Johnson R.A. (2000) Reduction of ammonia emission by slurry application techniques. J Agric Eng Res 77(3):277–287

Sørensen, P., and G.H. Rubæk. (2012). Leaching of nitrate and phosphorus after autumn and spring application of separated animal manures to winter wheat. Soil Use and Management, 28, 1-11.

Tetegan, M., Niccoullaud, B., Baize, D., Bouthier, A. and Cousin, I. (2011) The contribution of rock fragments to the available water content of stony soils: proposition of new pedotransfer functions. Geoderman, 165, 1, 40-49.

Thomas, C. (2004) Feed into Milk: An Advisory Manual. Nottingham University Press, 88 pp.

Thomas, H. A. (1981) Improved methods for national water assessment. Report No. WR15249270, U.S. Water Resourc. Counc., Washington D.C., 59 pp.

Thorman, R. E., Chadwick, D. R., Boyles, L. O., Matthews, R., Sagoo, E. & Harrison, R. (2006). Nitrous oxide emissions during storage of broiler litter and following application to arable land. In: Proceedings of the 2nd International Conference on Greenhouse Gases and Animal Agriculture. Zurich, Switzerland, 20 - 24 September 2005. International Congress Series 1293, Elsevier, Amsterdam, The Netherlands, 355-358.

Thorman, R. E., Chadwick, D. R., Harrison, R., Boyles, L. O. & Matthews, R. (2007a). The effect on N2O emissions of storage conditions and rapid incorporation of pig and cattle FYM into arable land. Biosyst. Eng. 97, 501-511.

Thorman, R., Sagoo, E, Williams, J. R., Chambers, B. J., Chadwick, D., Laws, J.A. et al. (2007b). The effect of slurry application timings on direct and indirect N2O emissions from free draining grassland soils. In: Towards a Better Efficiency in N Use. Proceedings of the 15th Nitrogen Workshop, Lleida, Spain, 28-30th May 2007, eds. A. Bosch, M.R. Teira & J.M. Villar, 297-299.

Thorman, R. E., Williams, J. R. & Chambers, B. J. (2009). Biosolids Recycling To Agricultural Land: Greenhouse Gas Emissions. In: 14th European Biosolids and Organic Resources Conference and Exhibition, November 2009

Thorman, R., Nicholson, F., Topp, C., Bell, M., Cardenas, L., Chadwick, D., Cloy, J., Misselbrook, T., Rees, R., Watson, C. and Williams, J. (2020) Toward country specific nitrous oxide emission factors for manures applied to arable and grassland soils in the United Kingdom. Frontiers in Sustainable Food Systems, 4, 62, 1-19.

Vadas, P.A., Gburek, W.J., Sharpley, A.N., Kleinman, P.J.A., Moore Jr, P.A., Cabrera, M.L. & Harmel, R.D. (2007). A model for phosphorus transformation and runoff loss for surface-applied manures. J. Environ. Qual. 36, 324-332.

Vadas, P.A., Good, L.W., Jokela, W.E., Karthikeyan, K.G., Arriaga, F.J. & Stock, M. (2017). Quantifying the impact of seasonal and short-term manure application decisions on phosphorus loss in surface runoff. J. Environ. Qual. 46, 1395-1402.WRAP (2016) Digestate and Compost in Agriculture: Good Practice Guide. https://wrap.org.uk/resources/guide/compost-and-digestate-agriculture-good-practice-guide#:~:text=Anaerobic%20digestate%20and%20compost%20can,be%20applied%20to%20the%20field.

Webb, J. and Misselbrook, T. (2004) A mass flow model of ammonia emissions from United Kingdom livestock production. Atmospheric Environment, 38, 14, 2163-2176.

Webb, J., Anthony, S. G., Brown, L., Lyons-Visser, H., Ross, C., Cottrill, B., Johnson, P. and Scholefield, D. (2005). The impact of increasing the length of the cattle grazing season on emissions of ammonia and nitrous oxide and on nitrate leaching in England and Wales. Agriculture Ecosystems & Environment, 105, 307-321.

Webb, J., Thorman, R. E. Fernanda-Aller, M. & Jackson, D.R. (2014). Emission factors for ammonia and nitrous oxide emissions following immediate manure incorporation on two contrasting soil types. Atmos. Environ. 82, 280-287.

White CA, Holmes HF, Morris NL & Stobart RM (2016). A review of the benefits, optimal crop management practices and knowledge gaps associated with different cover crop species. Research Review No. 90. AHDB Cereals & Oilseeds

White, J.W., Coale, F.J., Sims. J.T. & Shober, A.L. (2010). Phosphorus runoff from waste water treatment biosolids and poultry litter applied to agricultural soils. J. Environ. Qual. 39, 314-323.

Williams, J., Munro, D., Sagoo, L & Nicholson, F. (2016a). Review of guidance on organic manure nutrient supply in the fertiliser manual. AHDB Research Review No. 3110149017.

Williams, J., Nicholson, F. & Munro, D. (2016b). Pollutant Losses from Manures Stored in Field Heaps. Final report for Defra project WT1568 (Work Packages 3 and 4).

Williams, J., Sagoo, L., Wright, E. & Lee, D. (2015a). Pollutant losses from solid manure applications and from solid manures stored in temporary field heaps. Final report for Defra project WT1568 (Work Package 1)

Williams, J., Sagoo, L., Wright, E. & Lee, D. (2015b). The impacts of closed-periods for spreading farmyard manure on nitrate leaching losses. Final report for Defra project WT1568 (Work Package 2).

Withers, P.J.A, Clay, S.D & Breeze, V.G. (2001). Phosphorus transfer in runoff following application of fertilizer, manure, and sewage sludge. J. Environ. Qual. 30, 180-188.

Withers, P.J.A. & Bailey, G.A. (2003). Sediment and phosphors transfer in overland flow from a maize field receiving manure. Soil Use Manage., 19, 28-35.

Withers, P.J.A. & Hodgkinson, R. (2009). The effect of farming practices on phosphorus transfer to a headwater stream in England. Agricult. Ecosyst. Environ., 131, 347-355.

Withers, P.J.A. (2011). The Agronomic and Environmental Impacts of Phosphorus in Biosolids Applied to Agricultural Land: A Review of UK Research. UKWIR Report 11/SL/02/10

Withers, P.J.A., Ulen, B., Stamm, C. & Bechmann, M. (2003). Incidental phosphorus losses – are they significant and can they be predicted? J. Plant Nutrit. Soil Sci. 166, 459-468.

Withers, P.J.A., van Dijk, K.C., Neset, T.S.S., Nesme, T., Oenema, O., Rubæk, G.H., Schoumans, O.F., Smit, B. & Pellerin, S. (2015). Stewardship to tackle global phosphorus inefficiency: the case of Europe. Ambio. 44, 193-206.

WRAP (2016) Digestate and Compost in Agriculture: Good Practice Guidance https://wrap.org.uk/resources/guide/compost-and-digestate-agriculture-good-practice-guide

WRAP (2020) A review of technologies to optimise the value of digestate https://wrap.org.uk/resources/report/anaerobic-digestion-and-composting-latest-industry-survey-report-new-summaries

Xia, Y., Zhang, M., Tsang, D.C.W., Geng, N., Lu, D., Zhu, L., Igalavithana, A.D., Dissanayake, P.D., Rinklebe, J., Yang, X. & Ok, Y.S. (2020). Recent advances in control technologies for non-point source pollution with nitrogen and phosphorous from agricultural runoff: current practices and future prospects. Applied Biological Chemistry, 63:8

Zhou, M., Zhu, B., Wang, S., Zhu, X., Vereecken, H. & Brüggemann, N. (2017). Stimulation of N2O emission by manure application to agricultural soils may largely offset carbon benefits: a global meta-analysis. Glob. Change Biol. 23, 4068-4083.

11. Appendices

11.1. Model input data

Table 11.1 Percentage (%) of managed manure quantity spread to each RB209 soil type, within each of the 'Public Face Areas', in the baseline (present day) scenario.

Public Face Area	Deep Clay	Deep Silt	Light Sand	Medium	Organic	Peat	Shallow
Cumbria and Lancashire	11.6	4.9	5.0	59.9	2.8	14.3	1.5
Devon Cornwall and the Isles of Scilly	17.3	1.7	1.0	70.2	3.2	4.0	2.5
East Anglia	22.5	0.8	16.4	43.9	7.1	2.7	6.6
East Midlands	42.0	0.4	6.2	47.0	0.8	1.8	1.9
Greater Manchester Merseyside and Cheshire	40.7	1.6	23.8	30.1	0.9	2.9	0.0
Hertfordshire and North London	29.0	9.8	0.0	55.8	0.8	0.0	4.7
Kent South London and East Sussex	39.5	31.7	0.4	18.3	0.8	0.0	9.2
Lincolnshire and Northamptonshire	42.1	6.8	9.2	29.1	3.5	0.9	8.3
North East	35.0	0.0	1.2	47.3	1.8	14.6	0.2
Solent and South Downs	29.0	17.2	4.0	25.9	1.9	0.8	21.2
Thames	42.9	4.4	1.1	32.2	0.9	0.0	18.5
Wessex	43.7	5.6	0.6	24.6	4.4	2.4	18.6
West Midlands	24.7	13.2	6.3	50.6	0.5	3.5	1.3
Yorkshire	32.0	1.8	5.5	41.5	2.4	11.2	5.5

Table 11.2 Estimated average total and plant available soil water content (%) for top (0 to 300 mm depth) and sub soil (300 to 600 mm depth), and the profile dry albedo and standard percentage runoff (%), for the group of all soil series belonging to each RB209 soil type within England and Wales.

RB209 Soil Type	Top Soil			Sub Soil			Profile	
	Field Capacity	Total Plant Available Water	Easy Plant Available Water	Field Capacity	Total Plant Available Water	Easy Plant Available Water	Dry Albed o	Standard Percentage Runoff
Light Sand	30.9	17.2	11.6	17.4	10.3	7.4	0.3	17.5
Shallow	43.4	23.0	15.3	23.7	12.8	6.9	0.25	12.4
Medium	42.1	20.3	12.1	36.0	14.8	8.2	0.2	26.3
Deep Clay	48.0	20.9	12.6	43.4	13.7	7.4	0.15	42.7
Deep Silt	45.8	22.5	13.2	39.5	16.9	9.9	0.2	32.5
Organic	47.2	25.3	17.2	34.6	17.1	10.9	0.15	19.7
Peaty	60.7	34.1	26.3	53.3	28.4	20.1	0.1	50.7

Table 11.3 Percentage (%) distribution of the total crop and improved grass areas managed by farms between the six crop groups recognised by this study, within each of the 'Public Face Areas' in the baseline (present day; 2015 to 2019) scenario.

Public Face Area	Grass	Forage	Oilseed Rape	Cereal	Sugar Beet	Potato	Other
Cumbria and Lancashire	89.9	0.8	0.3	7.3	0.1	0.5	1.1
Devon Cornwall and the Isles of Scilly	79.3	3.2	0.8	13.2	0.1	0.9	2.5
East Anglia	16.1	1.6	10.0	53.9	5.4	3.1	9.8
East Midlands	48.9	2.1	9.1	33.0	1.1	0.9	4.9
Greater Manchester Merseyside and Cheshire	68.8	4.8	1.8	19.7	0.1	2.4	2.4
Hertfordshire and North London	26.5	1.5	7.4	53.9	0.8	0.3	9.5
Kent South London and East Sussex	47.8	2.2	6.6	31.6	0.1	0.4	11.4
Lincolnshire and Northamptonshire	17.3	1.5	15.4	50.2	3.0	1.9	10.6
North East	66.9	0.6	5.1	25.4	0.0	0.3	1.7
Solent and South Downs	45.1	3.2	7.4	37.5	0.1	0.6	6.1
Thames	43.0	2.3	9.1	39.7	0.1	0.1	5.6
Wessex	61.4	4.8	4.9	25.2	0.1	0.2	3.4
West Midlands	54.6	3.6	5.9	28.5	0.3	1.6	5.3
Yorkshire	47.4	1.2	7.2	37.4	0.7	1.8	4.2

Table 11.4 Reference application rate and nutrient content of managed manures applied to agricultural land.

Manure Type	Total N rate	Total P Rate	Dry Matter	Unit Application Rate	Total N Content	Total P Content
	(kg/ha)	(kg/ha)	(%)	(t/ha or m^3 /ha)	(kg/t or kg/m^3)	
Dairy Slurry	90.8	18.7	8.5	26.7	3.4	0.7
Dairy FYM	99.8	24.1	23	17.2	5.8	1.4
Dairy Digestate	110.4	22.1	3.2	27.6	4	0.8
Beef Slurry	90.8	18.7	8.5	26.7	3.4	0.7
Beef FYM	99.8	24.1	23	17.2	5.8	1.4
Beef Digestate	110.4	22.1	3.2	27.6	4	0.8
Pig Slurry	92.2	14.4	3.3	28.8	3.2	0.5
Pig FYM	178.6	58	24	23.2	7.7	2.5
Pig Digestate	92.2	14.6	3.2	24.5	3.8	0.6
Poultry Litter	198.8	46.9	57	7.1	28	6.6
Poultry Layer Manure	155.4	48.8	48	7.4	21	6.6
Poultry Digestate	198.8	41.4	3.2	27.6	7.2	1.5

Table 11.5 Comparison of the percentage (%) monthly distribution of manure nitrogen applied to crop and grassland in England (2015 to 2019), using all records in the British Survey of Fertiliser Practice, and the quality controlled sub-set used in this study for the baseline (present day) scenario.

Manure Type	Dataset	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Cattle FYM	All	3.3	8.8	18.5	11.1	4.4	2.3	5.3	10.8	21.4	8.3	4.0	1.9
	Baseline	3.6	9.5	19.6	11.5	4.7	2.5	5.0	7.2	21.5	8.5	4.3	2.0
Cattle SLY	All	3.6	15.7	22.1	10.0	12.1	10.8	7.5	6.4	4.4	3.8	2.1	1.5
	Baseline	4.3	18.0	23.8	10.1	12.4	11.8	8.2	1.4	2.5	3.3	2.4	1.8
Poultry Litter (60% DM)	All	3.2	3.4	21.8	5.3	1.5	1.5	3.7	24.1	30.9	4.3	0.3	0.0
	Baseline	3.8	3.9	18.5	5.2	1.8	0.0	0.0	25.8	35.8	5.1	0.0	0.0
Layer Manure (40% DM)	All	0.0	7.2	8.5	7.0	3.5	1.1	0.8	32.9	31.5	5.0	1.6	0.9
	Baseline	0.0	9.1	8.2	8.3	4.0	1.4	0.5	36.0	24.3	5.5	1.5	1.2
Pig FYM	All	1.7	10.1	11.4	4.0	1.0	0.0	0.7	30.0	33.3	6.0	0.8	1.1
	Baseline	2.0	9.3	12.4	3.2	1.1	0.0	0.0	32.0	31.0	6.9	0.9	1.2
Pig SLY	All	0.2	12.1	26.5	16.4	1.0	2.1	6.0	24.6	8.9	1.3	0.9	0.0
	Baseline	0.3	12.7	26.4	17.2	0.7	2.5	6.6	23.5	7.8	1.3	1.0	0.0
Digestate Produced on Farm	All	14.7	5.7	21.9	23.4	8.3	2.3	3.7	6.4	12.3	0.3	0.0	1.1
	Baseline	4.7	7.2	29.5	27.2	5.0	1.3	6.5	10.7	5.6	0.5	0.0	1.8

Table 11.6 Count of farms and individual manure application records in the subset of quality controlled data from the British Survey of Fertiliser Practice (2015 to 2019) used to character baseline (present day) manure management

a) Count of Farms

Manure Type	2015	2016	2017	2018	2019
Cattle FYM	437	370	383	433	477
Cattle SLY	149	117	125	148	157
Pig FYM	31	32	34	32	31
Pig SLY	12	12	15	8	11
Poultry Litter (60% DM)	35	19	21	22	30
Layer Manure (40% DM)	30	21	17	24	19
Digestate Produced on Farm	0	5	4	5	3

b) Count of Applications

Manure Type	2015	2016	2017	2018	2019
Cattle FYM	972	788	829	938	991
Cattle SLY	543	473	472	576	539
Pig FYM	47	56	55	56	57
Pig SLY	27	31	46	22	26
Poultry Litter (60% DM)	56	37	32	35	53
Layer Manure (40% DM)	51	39	33	42	27
Digestate Produced on Farm	0	25	8	15	12

11.2. Modelling results

Table 11.7 Baseline emission factors (expressed as a % of N or P applied)

Organic material	Sand				Clay			
	Aut.	Win.	Spr.	Sum.	Aut.	Win.	Spr.	Sum.
NH_3-N								
Cattle slurry	9.0	11.9	11.4	18.4	9.5	11.9	11.6	18.4
Pig slurry	8.0	10.6	11.6	13.8	8.1	13.1	12.0	13.8
Manure-based digestate	15.4	20.0	16.6	18.3	15.5	20.3	17.1	18.9
Poultry manures	17.1	27.8	22.6	22.7	17.5	29.8	24.8	23.6
Cattle FYM	5.5	6.7	6.0	6.5	5.5	6.7	6.0	6.6
Pig FYM	7.3	7.3	6.9	6.5	7.5	7.8	7.5	6.6
NO_3-N								
Cattle slurry	20.7	6.3	<0.1	<0.1	6.4	7.7	1.5	0.1
Pig slurry	33.3	8.3	<0.1	<0.1	7.6	13.1	1.2	0.1
Manure-based digestate	38.4	2.4	<0.1	<0.1	8.9	10.2	0.9	0.1
Poultry manures	33.0	5.1	<0.1	<0.1	8.1	6.5	1.0	0.1
Cattle FYM	4.5	1.4	<0.1	<0.1	1.2	1.2	0.1	<0.1
Pig FYM	5.1	0.7	<0.1	<0.1	1.3	0.9	0.2	<0.1
N_2O								
Cattle slurry	0.9	0.8	0.8	0.7	0.8	0.8	0.8	0.7
Pig slurry	1.1	0.9	0.9	0.9	0.9	1.0	0.9	0.9
Manure-based digestate	1.2	1.0	0.9	0.9	1.0	1.0	0.9	0.9
Poultry manures	0.8	0.7	0.6	0.6	0.6	0.7	0.6	0.6
Cattle FYM	0.2	0.1	0.1	0.1	0.2	0.1	0.1	0.1
Pig FYM	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Total P								
Cattle slurry	0.2	1.2	0.8	0.1	1.4	5.7	3.7	1.2
Pig slurry	<0.1	0.6	0.7	0.1	0.3	3.7	3.2	0.9
Manure-based digestate	<0.1	1.2	0.4	0.1	0.4	5.9	2.0	0.7
Poultry manures	<0.1	0.9	0.3	0.1	0.1	4.8	1.8	0.7
Cattle FYM	0.1	0.8	0.4	0.1	0.5	3.5	1.7	0.7
Pig FYM	<0.1	<0.1	0.1	<0.1	0.1	0.2	0.6	0.1