

Advances in Sustainable Nutrient Recovery for the Management of Nitrogen-rich Residue Streams (REFERT)

Authors: Bart Bonsall, Donncha Haverty, Corine Nzeteu, Pádraic Ó hUiginn and Vincent O'Flaherty



Environmental Protection Agency

The EPA is responsible for protecting and improving the environment as a valuable asset for the people of Ireland. We are committed to protecting people and the environment from the harmful effects of radiation and pollution.

The work of the EPA can be divided into three main areas:

Regulation: Implementing regulation and environmental compliance systems to deliver good environmental outcomes and target those who don't comply.

Knowledge: Providing high quality, targeted and timely environmental data, information and assessment to inform decision making.

Advocacy: Working with others to advocate for a clean, productive and well protected environment and for sustainable environmental practices.

Our Responsibilities Include:

Licensing

- > Large-scale industrial, waste and petrol storage activities;
- > Urban waste water discharges;
- > The contained use and controlled release of Genetically Modified Organisms;
- > Sources of ionising radiation;
- > Greenhouse gas emissions from industry and aviation through the EU Emissions Trading Scheme.

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- > Audit and inspection of EPA licensed facilities;
- > Drive the implementation of best practice in regulated activities and facilities;
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- > Assess and report on public and private drinking water quality;
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- > Prosecute those who flout environmental law and damage the environment.

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- > Implement and enforce waste regulations including national enforcement issues;
- > Prepare and publish national waste statistics and the National Hazardous Waste Management Plan;
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- > Engage with national and regional governance and operational structures to implement the Water Framework Directive;
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- > Publish Ireland's greenhouse gas emission inventories and projections;

- > Provide the Secretariat to the Climate Change Advisory Council and support to the National Dialogue on Climate Action;
- > Support National, EU and UN Climate Science and Policy development activities.

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- > Design and implement national environmental monitoring systems: technology, data management, analysis and forecasting;
- > Produce the State of Ireland's Environment and Indicator Reports;
- > Monitor air quality and implement the EU Clean Air for Europe Directive, the Convention on Long Range Transboundary Air Pollution, and the National Emissions Ceiling Directive;
- > Oversee the implementation of the Environmental Noise Directive;
- > Assess the impact of proposed plans and programmes on the Irish environment.

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- > Coordinate and fund national environmental research activity to identify pressures, inform policy and provide solutions;
- > Collaborate with national and EU environmental research activity.

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- > Monitoring radiation levels and assess public exposure to ionising radiation and electromagnetic fields;
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- > Provide independent evidence-based reporting, advice and guidance to Government, industry and the public on environmental and radiological protection topics;
- > Promote the link between health and wellbeing, the economy and a clean environment;
- > Promote environmental awareness including supporting behaviours for resource efficiency and climate transition;
- > Promote radon testing in homes and workplaces and encourage remediation where necessary.

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The EPA is managed by a full time Board, consisting of a Director General and five Directors. The work is carried out across five Offices:

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2. Office of Environmental Enforcement
3. Office of Evidence and Assessment
4. Office of Radiation Protection and Environmental Monitoring
5. Office of Communications and Corporate Services

The EPA is assisted by advisory committees who meet regularly to discuss issues of concern and provide advice to the Board.

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Identifying pressures

Across the EU, anthropogenic sources of nitrogen (N) threaten water and air quality, while greenhouse gas (GHG) emissions drive climate change.

Agriculture is a significant source of N and GHG emissions. N for crops is sourced from mineral fertilisers, livestock manures and anaerobic digestion digestates. Periodic application can result in nutrient concentrations that exceed the uptake capabilities of crops, leading to:

- N losses to surface or groundwater, leading to problems including eutrophication;
- ammonia emissions that contribute to fine particulate pollution and acid rain;
- the release of nitrous oxide, a potent GHG.

GHG emissions also arise from fertiliser-related practices, for example:

- Mineral fertiliser production is a high-intensity source of GHG emissions.
- Livestock manures release the potent GHG methane during storage.
- Tractors generate transport-related GHGs.

Other ammoniacal effluents, such as landfill leachates, also generate N and GHG emissions.

The aim of this project was to identify cost-effective methods to mitigate the release of N and GHGs from ammoniacal streams.

Informing policy

A community-based “hub and spoke” business model was proposed to aggregate slurries from the fragmented farm supply chain. Introduction of a sustainable nutrient certification scheme was proposed to engage farmers, remunerating them for supply of slurry for processing, consistent with new EU Common Agricultural Policy objectives. Implementation of a sustainable slurry management system is consistent with other EU and national policy objectives related to agriculture, the environment, energy, climate change and the economy.

The project concluded that this sustainable slurry management concept offered a route to improve the economic and environmental sustainability of agriculture. Validation of the potential benefits, however, requires further development of process technology, which to date has been undertaken only

at prototype level. Integration of technology and business model concepts in a multi-cycle, end-to-end, commercial-scale demonstration is required to validate economic and environmental assumptions and to determine acceptance among stakeholder groups. Development of regulatory roadmap(s) will be critical to allow for more comprehensive economic and impact assessments to be undertaken, and for progression of the proposed system towards full-scale application.

Developing solutions

This project identified cost-effective methods to mitigate the release of N and GHGs from ammoniacal streams. The development of suitable technological solutions and a business model for their widespread adoption were both in the scope of the project.

Technological outcomes included:

- the demonstration of a scalable, modular treatment system;
- demonstrating that effluents could be treated to substantially reduce N, relative to current practices;
- demonstrating that solids could be conditioned to recover renewable energy;
- demonstrating that spent filter media and lime could be recovered as precursors to saleable soil-conditioning products;
- demonstrating that ammonia could be recovered as a precursor to a saleable fertiliser product.

The technology suite can be flexibly configured to deal with different ammoniacal effluents.

The project also identified a potential business model that:

- engages stakeholders by making it convenient and economical for farmers to participate in the slurry supply chain;
- uses technology for efficient aggregation of a widely dispersed slurry resource to achieve economies of scale;
- generates revenues from sales of productised fractions that can be applied to offset processing costs, required to underpin economic viability;
- is readily replicated, promoting commercialisation and progress towards sustainability objectives.

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This report is based on research carried out/data from mid-2019 to mid-2021. More recent data may have become available since the research was completed.

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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Executive Summary

Context

Across the EU, anthropogenic sources of nitrogen (N) threaten water and air quality, while greenhouse gas (GHG) emissions drive climate change.

Agriculture is a significant source of N and GHG emissions. N for crops is sourced from mineral fertilisers, livestock manures and anaerobic digestion digestates. Periodic application can result in nutrient concentrations that exceed the uptake capabilities of crops, leading to:

- N losses to surface or groundwater, leading to problems including eutrophication;
- ammonia (NH₃) emissions that contribute to fine particulate pollution and acid rain;
- the release of nitrous oxide (N₂O), a potent GHG.

GHG emissions also arise from fertiliser-related practices, for example:

- Mineral fertiliser production is a high-intensity source of GHG emissions.
- Livestock manures release the potent GHG methane during storage.
- Tractors generate transport-related GHGs.

Other ammoniacal effluents, such as landfill leachates, also generate N and GHG emissions.

Objectives

The aim of this project was to identify cost-effective methods to mitigate the release of N and GHGs from ammoniacal streams. Technological objectives included:

- the demonstration of a scalable, modular treatment system for different ammoniacal effluents;
- demonstrating that ammoniacal effluents could be clarified and conditioned to substantially reduce N, relative to current practices;
- demonstrating that solids could be recovered and conditioned to recover renewable energy;
- demonstrating that spent filter media and lime could be recovered as precursors to saleable soil-conditioning products;

- demonstrating that NH₃ could be recovered as a precursor to a saleable fertiliser product.

The experimental phase of the project in 2020 and early 2021 was severely affected by the COVID-19 pandemic, which led to travel restrictions and laboratory closures. This compromised elements of the work programme and hence the overall outcomes of the project. The technology components were, however, tested on a pilot scale using different ammoniacal effluents with the following results:

- **Landfill leachates.** Struvite technology reduced NH₃ concentrations from 1500 parts per million (ppm) to c.100 ppm. Deployment of the technology at scale would facilitate more efficient use of on-site wastewater treatment plants. Supplemental application of reverse osmosis reduced NH₃ concentrations to <5 ppm.
- **De-watered pig slurries.** Slurries were screened and filtered to remove solids. NH₃ stripping via liming generated a recoverable NH₃ stream, reducing NH₃ concentrations from c.2500 ppm to <500 ppm. Struvite precipitation reduced residual NH₃ concentrations further, to <50 ppm. The clarified liquor could be distributed via umbilical pump, thus avoiding transport to fields.

The technology suite can be flexibly configured to deal with ammoniacal effluents from different sources. In the case of high-solids effluents, such as livestock slurries and digestates, further development work is required to optimise solids removal. Further work is also required to develop, test and optimise technology to generate slow-release fertiliser products from NH₃ and solids fractions, and to advance the concept towards commercial deployment.

The project was also tasked with identifying a business model that:

- engages stakeholders by making it convenient and economical for farmers to participate in the slurry supply chain;
- uses technology for efficient aggregation of a widely dispersed slurry resource to achieve economies of scale;

- generates revenues from sales of productised fractions that can be applied to offset processing costs, required to underpin economic viability;
- is readily replicated, promoting commercialisation and progress towards sustainability objectives.

A community-based “hub and spoke” business model was proposed to aggregate slurries from the fragmented farm supply chain. Introduction of a sustainable nutrient certification scheme was proposed to engage farmers, remunerating them for supply of slurry for processing, consistent with new EU Common Agricultural Policy objectives. Implementation of a sustainable slurry management system is consistent with other EU and national policy objectives related to agriculture, the environment, energy, climate change and the economy.

Finally, a techno-economic review concluded that this sustainable slurry management concept offered a route to improve the economic and environmental sustainability of agriculture. Validation of the potential benefits, however, requires further development of process technology, which to date has been undertaken only at prototype level. Integration of technology and business model concepts in a multi-cycle, end-to-end, commercial-scale demonstration is required to validate economic and environmental assumptions and to determine acceptance among stakeholder groups. Development of regulatory roadmap(s) will be critical to allow for more comprehensive economic and impact assessments to be undertaken, and for progression of the proposed system towards full-scale application.

1 Introduction

Nitrogen (N) is present in the environment in a variety of chemical forms, including as dinitrogen (N_2) gas, which makes up 78% of the Earth's atmosphere, ammonia (NH_3), ammonium (NH_4^+), nitrates (NO_3^-), nitrites (NO_2^-) and other oxygenated N species, such as gaseous nitric oxide (NO) and nitrous oxide (N_2O), which together are known as nitrogen oxides (NO_x). Atmospheric N_2 is the ultimate source of N for living organisms, and it is fixed or converted to inorganic forms by bacteria in soils and waterbodies. N is then incorporated into proteins and other cellular structures and, when organisms die, these organic N forms are incorporated into soil humus or other matrices, before decomposition returns N_2 to the atmosphere. This process is known as the N cycle (Bernhard, 2010).

Industrialisation, internal combustion engines and the intensification of agriculture have affected the global N cycle, resulting in imbalances arising from:

- fertiliser applications, resulting in run-off NO_3^- losses to surface waters and groundwaters, fugitive NH_3 and other gaseous N emissions;
- storage and land spreading of livestock manure, resulting in atmospheric N emissions and run-off losses;
- municipal and industrial waste management causing N losses to surface waters and groundwaters;
- combustion of fossil fuels generating gaseous emissions of oxygenated N species.

The principal forms of N-related pollutants arising from anthropogenic activities include:

- **Atmospheric particulate matter (PM).** This aggravates respiratory problems and lung diseases. PM can be classified as coarse (PM_{10} ; particles with an aerodynamic diameter of $< 10 \mu m$) or fine ($PM_{2.5}$; particles with an aerodynamic diameter of $< 2.5 \mu m$). Residential and commercial sources account for c.55% of $PM_{2.5}$ emissions in Ireland, with 7.4% derived from agriculture (EPA, 2020).
- **Nitrogen oxides.** NO_x arise from combustion, animal manure/slurry and fertiliser management.

NO_x harm health and contribute to acid rain. N_2O is also a potent greenhouse gas (GHG).

- **Ammonia.** NH_3 arises mainly from manure/slurry and fertiliser management, but also from industry and transport. NH_3 reacts with gases such as sulfur dioxide (SO_2) and NO_x to form NH_4^+ -containing aerosols, a major source of $PM_{2.5}$. Atmospheric NH_3 and NO_x also drive the formation of acid rain (Bobbink *et al.*, 2010).

1.1 Effects of Nitrogen Imbalances on the Environment and Human Health

N imbalances have significant environmental and human health implications, threatening water quality, air quality, vegetation and soil quality, and driving climate change, as outlined below.

1.1.1 Water quality

Excess N causes overstimulation of the growth of photosynthetic microorganisms and algae, leading to the oxygen depletion and eutrophication of waterbodies (Clislock *et al.*, 2013). Algal blooms affect the use of water for recreational activities, and excessive NO_3^- in drinking water can be harmful (Clislock *et al.*, 2013; EPA, 2021a).

A range of policy measures aim to protect water quality from N and its impacts:

- The Water Framework Directive (2000/60/EC as amended) is given legal effect in Ireland by S.I. No. 722/2003 – European Communities (Water Policy) Regulations 2003. The Water Framework Directive requires EU Member States to monitor, protect and improve water quality in all rivers, lakes, groundwater and transitional coastal waters, and to prepare river basin management plans designed to achieve good ecological status by 2027 at the latest (Department of Housing, Local Government and Heritage, 2022a).
- The Drinking Water Directive (98/83/EC as amended) is given legal effect in Ireland by S.I. No. 122 of 2014 European Union (Drinking

Water) Regulations 2014 (Irish Statute Book, 2014a) and obligates Member States to maintain minimum quality thresholds in water intended for human consumption.

- The Urban Waste Water Treatment Directive (91/271/EEC as amended) is implemented in Ireland pursuant to S.I. No. 214/2020 – European Union (Waste Water Discharge) Regulations 2020 (Irish Statute Book, 2020a). The directive requires discharges from municipal wastewater treatment (WWT) plants to be licensed or certified, and sets standards for the collection and treatment of wastewater.

A recent Irish water quality indicators report (EPA, 2021a) highlights that environmental indicators for water are deteriorating despite these policies, investments and actions. It notes that:

- 47% of river sites have unsatisfactory NO_3^- concentrations, with 38% of sites showing an increasing NO_3^- trend for the period 2013–2020, and only 3% showing a decreasing trend;
- 22% of estuarine and coastal waters have unsatisfactory dissolved inorganic N levels;
- marine N discharges from rivers have increased by 26% (14,574 tonnes) since 2012–2014;
- 24% of groundwater monitoring sites have high NO_3^- concentrations ($> 25 \text{ mg L}^{-1} \text{ NO}_3^-$) and three sites exceed the drinking water standard ($50 \text{ mg L}^{-1} \text{ NO}_3^-$);
- 47% of all groundwater sites had increasing NO_3^- concentrations for the period 2013–2020.

1.1.2 Air quality

Atmospheric NH_3 has effects on the local and international (transboundary) scales. The transport lifetime of NH_3 is relatively short (10–100 km), but NH_4^+ may be transferred over much longer distances (from 100 to > 1000 km) (Asman *et al.*, 1998). Hence, NH_3 emissions are addressed by the 1979 United Nations Economic Council for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution (UNECE, 1979), which includes a requirement for countries to cooperate in managing air pollution. In addition, the UNECE Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (UNECE, 1999) established national ceilings on certain air pollutants and set reduction targets for pollutant

concentrations that should have been achieved by 2020 (relative to a 2005 baseline).

The World Health Organization (WHO) publishes air quality guidelines, with recommended limit values for specific air pollutants, aiming to achieve air quality that protects public health. In the EU, a range of policy measures target the management of air quality in an increasingly industrialised and agriculturally intensive environment. These measures are incorporated into the Irish Clean Air Strategy (An Taisce, 2022) and include the following:

- Directive 2008/50/EC on Ambient Air Quality and Cleaner Air for Europe (CAFE Directive) as amended (European Union, 2008), which has been transposed into Irish legislation by the Air Quality Standards Regulations 2011 (S.I. No. 180 of 2011). It establishes background limits and target values, and obligations to monitor and report on levels of pollutants in ambient air, including NO_x and $\text{PM}_{2.5}$ and PM_{10} .
- The National Emissions Ceilings Directive 2016 (Directive 2016/2284) (European Union, 2016) establishes emissions caps and obligations to reduce emissions (relative to 2005 baselines) by 2020, including for NO_x (49% reduction) and NH_3 (1% reduction).

The EPA monitors and reports on air quality, as regulated and informed by EU and WHO standards. A recent report on key air pollutants (EPA, 2021b) indicates that, while Ireland is compliant with the emissions reduction commitments for 2020 under the National Emissions Ceilings Directive for NO_x , NH_3 emissions are non-compliant with the 2020 commitments. The report highlights that, although NH_3 emissions decreased slightly in 2020, they have been non-compliant for 8 of the past 9 years.

1.1.3 Vegetation

Atmospheric NH_3 is taken up by leaves, with significant negative effects on cellular functions (Pitcairn *et al.*, 2006; Sheppard *et al.*, 2009). Mosses and lichens are most at risk because of their limited detoxification capacity and large surface area relative to mass compared with other species. Specific effects of NH_3 on vegetation include increased sensitivity to frost and drought stresses, increased incidence of pest and pathogen attacks, reduced root growth and

mycorrhizal infection leading to reduced nutrient uptake (Krupa, 2003).

1.1.4 Soil quality

N fertilisers have an acidifying effect on soils. At excessive levels, soluble NH_4^+ can interfere with cation uptake by plants (Hoppen *et al.*, 2010). Acidic soils require periodic neutralisation by liming to ensure soil fertility and to mitigate N loss via soil emissions to the atmosphere. Liming rates in Ireland declined by approximately 50% from the 1980s to the 2000s, but are now increasing (Teagasc, 2022a).

An EU soil strategy for 2030 (European Commission, 2020a) is under development. Its vision and objectives are to achieve healthy soils by 2050, commencing with concrete actions by 2030. It will contribute to the objectives of the European Green Deal via a new soil health law, proposed to be introduced in 2023, which will set standards to support health protection, climate neutrality and a clean and circular economy, halting desertification and land degradation.

1.1.5 Climate change

Increased concentrations of GHGs, including N species such as N_2O , in the atmosphere are caused by human activity and increase the Earth's temperature. The World Meteorological Organization (2022) reported that GHG levels in the atmosphere reached a new record in 2020 and were increasing at an annual rate above the 2011–2020 average. The Intergovernmental Panel on Climate Change (2022) reports that climate change is causing dangerous and widespread disruption in nature, resulting in increases in the frequency and intensity of heatwaves, drought and floods.

The 2030 Energy and Climate Change Framework (European Commission, 2022a) proposes raising the 2030 GHG emission reduction target, including emissions and removals, to at least 55% compared with 1990, which will enable the EU to move towards a climate-neutral economy and implement its commitments under the European Green Deal and the Paris Agreement. Key targets of the 2030 Energy and Climate Change Framework include a minimum:

- 40% cut in GHG emissions (relative to 1990 emission levels);

- 32% share of renewable energy;
- 32.5% improvement in energy efficiency.

The 40% reduction in GHG emissions will be implemented via a combination of an increase in the share of renewable energy incorporated into the energy mix, an increase in EU Emissions Trading System obligations, an increase in binding obligations on Member States under the Effort Sharing Regulation and changes to the Land Use, Land Use Change and Forestry (LULUCF) Regulation. These obligations are given effect in Ireland via S.I. No. 32 of 2021, the Climate Action and Low Carbon Development (Amendment) Act 2021 (Irish Statute Book, 2021). EU land use has historically contributed a carbon sink that offsets GHG emissions. To secure this contribution on an ongoing basis, the European Commission adopted changes to the LULUCF Regulation, obligating each Member State to ensure that LULUCF emissions are entirely compensated for by removals of carbon dioxide equivalent ($\text{CO}_{2\text{-eq}}$) (the “no debit” rule). Proposals (Herold *et al.*, 2021) under new rules are intended to provide Member States with a framework to incentivise more climate-friendly land use, which may in future include application of the “polluter pays” principle to agricultural emissions (European Commission, 2021a).

1.2 Sources of Nitrogen and Related Greenhouse Gas Emissions in Ireland

In Ireland, the largest source of anthropogenic N and related GHG emissions is agriculture (Mockler *et al.*, 2017), although N species are also emitted from transport, landfill sites, sewage works, anaerobic digestion (AD), the composting of organic materials, combustion and industrial wastes and residues (Sutton *et al.*, 2000).

1.2.1 Agriculture

Irish agriculture is composed predominantly of a grass-based livestock system, with family-sized dairy and beef farms predominating. Ryan (2014) documents that surpluses of grassland N arise because of the high level of fertiliser inputs relative to the net outputs of N in milk and meat. These surpluses result in N losses that will be exacerbated by the intensification envisaged in Food Vision 2030 (Department of

Agriculture, Food and the Marine, 2021a). N loss arises from NH_3 volatilisation, NO_3^- leaching and NO_3^- denitrification. The rate of loss is affected by factors such as crop variety, the form, method or timing of fertiliser application, soil properties, rainfall and temperature (Ryan, 2014). In some areas, N emissions exceed acceptable Air Quality Framework standards (Doyle *et al.*, 2017) and give rise to nutrient run-off that triggers eutrophication and other pressures on inshore water quality.

Teagasc reports that Irish agriculture is responsible for 98% of Irish NH_3 emissions (Figure 1.1) (Teagasc, 2022b) and almost 90% of Irish N_2O emissions (Teagasc, 2022c). There is a direct relationship between NH_3 concentrations and N_2O emissions included in the national GHG inventory, which is currently based on the sum of:

- direct soil emissions resulting from N inputs from synthetic fertiliser, animal manure, sludges, fixation by N-fixing crops, crop residues returned to soils and cultivation of organic soils;
- emissions associated with N excreted by grazing animals;
- indirect emissions due to N deposition resulting from emissions of NH_3 and NO_x in agriculture and from NO_3^- leaching.

Agricultural NH_3 arises from the land application of manures or chemical fertilisers, such as calcium ammonium nitrate (CAN), monoammonium phosphate (MAP) or diammonium phosphate (DAP) or urea.

Volatilisation losses range from negligible amounts to >50% of the applied N, depending on the source of N (e.g. losses from manure exceed losses from urea, which in turn exceed losses from CAN). Application practice also influences loss levels (e.g. soil incorporation results in less volatilisation than surface application). Solubility and dissolution processes affect NH_3 and N_2O emissions, which are higher in warm, dry conditions and lower in cool, wet conditions (Peoples *et al.*, 1995). It is estimated that c.15% of the N in animal manures and 2% of the N contained in chemical fertilisers is lost to the atmosphere as NH_3 (Government of Ireland, 2019).

NH_3 and NO_3^- are highly soluble, while organic forms of N are incorporated into solids matrices or adhere to small sediment particles. During rain, excess N species (i.e. those not immediately taken up by crops) make their way into surface waters via **run-off** or can be lost via **leaching** into groundwaters (O'Mara *et al.*, 2007; Doyle *et al.*, 2017). Lawniczak *et al.* (2016) report that the relative balance between N, phosphorus (P) and potassium (K) in fertilisers has a significant bearing on N losses, as plants' capacity to

Agricultural emissions

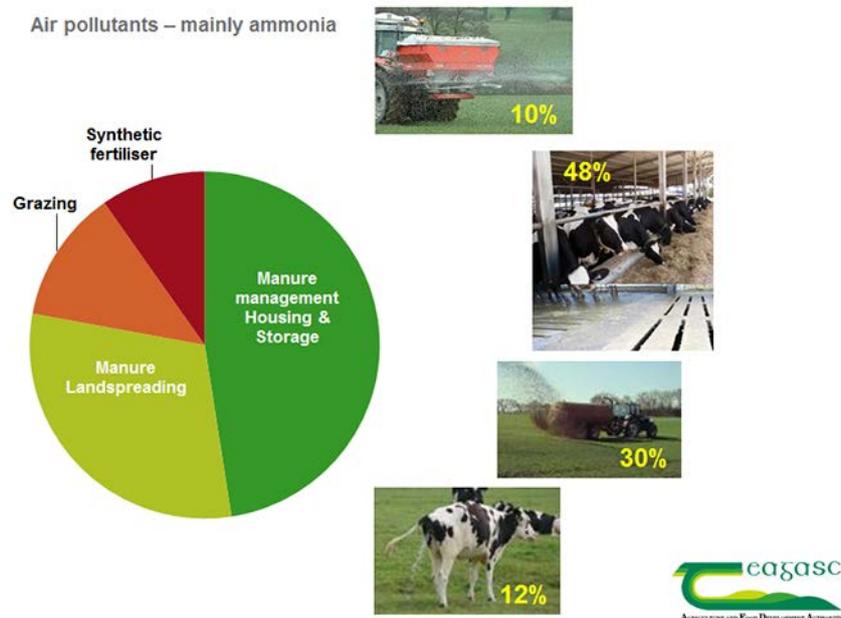


Figure 1.1. Sources of ammonia emissions in Irish agriculture (Source: Teagasc, 2022b).

utilise N is governed in part by the availability of the other nutrients.

Freeney (2005) notes that the efficiency of fertiliser N can be increased and losses reduced by matching supply with crop demand, optimising split application schemes, changing the form to suit the conditions, and using slow-release fertilisers and urease inhibitors. Ryan (2014) references methods such as the co-application of enzymatic or chemical inhibitors to mitigate N loss, but warns that the cost at small-farm level can be prohibitive. Bertora *et al.* (2008) found that separation of animal slurry solids from liquors facilitates different methods of application for each fraction, which can significantly reduce N₂O emissions.

Manure management

Given the predominance of livestock farming in Ireland, land spreading of manure/slurry is widely utilised to offset requirements for inorganic fertilisers, and leverages the availability of solubilised nutrients, and nutrients bound in cellular matrices and released over time by soil microbes (Dai *et al.*, 2016). An estimate of the approximate levels of organic and inorganic N in collected livestock slurry and farmyard manures was conducted (Table 1.1). The estimates were extrapolated from livestock headage as reported in the Central Statistics Office farm livestock reports 2019 and industry association literature for poultry and horses (Irish Food Board, 2019; Central Statistics Office, 2020; Statista, 2020), and estimates of

available N levels were then compared with values referenced in the “Nutrient Content in Livestock Excretions” tables incorporated into Irish Nitrates Directive legislation (S.I. 113 of 2022). Average collection percentages were based on Teagasc reports for cattle and pigs and various literature and media reports for other livestock. The estimates were compared with the estimated ammoniacal emissions from slurries and manures projected as of 2010 by Hyde *et al.* (2003).

The actual levels of N and related GHG emissions from slurries and farmyard manures not only depend on the soil pH, weather and relative balance between N, P and K, but are also influenced by the type of manure storage system, the length of the storage period and the timing and practices of land spreading (Timmons *et al.*, 1973). Liquid slurries give rise to lower N₂O emissions than solid farmyard manures, although losses of NH₃ to run-off may be higher. Use of closed containers reduces fugitive atmospheric discharges during storage. Nutrient losses are reduced when fertilisers are incorporated into soils as opposed to surface broadcast (Timmons *et al.*, 1973; Teagasc, 2022d).

In Ireland, land spreading of manures is governed by the Good Agricultural Practices for Protection of Waters Regulations 2022, which generally limit the application of manure-derived N to 170 kg per hectare (ha). This establishes a limit on stocking intensity. Ireland’s nitrates derogation provides farmers with

Table 1.1. Estimate of organic and inorganic N levels in collected Irish slurries and farmyard manures

Herd	Herd size (head)	Estimated average organic and inorganic N content – excreted manure (kg year ⁻¹)	Estimated portion collected (%)	Estimated organic and inorganic N resource in collected manures (t)
Dairy cows	1,425,800	85	25	30,298
Other mature cattle: >2 years	1,430,200	65	25	23,240
Other cattle: 1–2 years	1,751,400	57	25	24,957
Other cattle: ≤1 year	1,962,400	24	25	11,774
Sheep: breeding ewe and lambs	2,723,100	10	10	2723
Other sheep	1,185,200	5	10	593
Pigs: breeding unit and progeny	145,000	87	100	12,615
Pigs: finishing units	1,468,300	9.2	100	13,508
Poultry: broiler chickens	70,000,000	0.24	90	15,120
Poultry: laying hens	2,000,000	0.56	90	1008
Poultry: turkeys/ducks	4,000,000	1	75	3000
Horses	92,300	45	25	1038
Combined total				139,874

an opportunity to farm at higher stocking rates, subject to additional conditions designed to protect the environment. Almost 7000 intensively stocked farms availed of the derogation in 2018 (Department of Housing, Local Government and Heritage, 2020). Derogation farms covered c.445,200 ha (c.10%) of Irish agricultural lands, with the average farm size being 65 ha and housing an average 162 livestock units. While agricultural intensification is increasing, in general, Ireland’s grassland livestock system is still considered quite moderate in terms of kg manure N applied per ha. Over 60% of cattle and dairy farms have a stocking rate of under 85 kg livestock manure N ha⁻¹. At the end of 2018, the national herd was 6.6 million head, an increase of 0.74 million head since 2010 (+10.3%). The distribution of stocking intensity as reported by the Department of Agriculture, Food and the Marine (DAFM) can be seen in Table 1.2.

Table 1.2. Distribution of livestock farms by stocking rate for 2018

Stocking rate (kg bovine livestock manure N ha ⁻¹)	Farms	
	No.	%
Extensive (≤85)	80,919	61.0
Moderately intensive (86–130)	26,602	20.0
Intensive (131–170)	17,420	13.1
Very intensive (>171)	7855	5.9

Source: Nitrates section, as reported in Department of Housing, Local Government and Heritage (2020).

Chemical fertilisers

Chemical fertilisers such as urea and CAN are applied to grassland and tillage crops to supplement livestock slurries and manures. While the increasing use of chemical fertilisers has plateaued compared with the very rapid escalation seen in the later part of the 1990s and early 2000s, chemical fertiliser sales increased by over 10% in 2018, following a similar increase in 2017 (Figure 1.2) (Central Statistics Office, 2021). To manage application rates, Teagasc is widely promoting the use of stabilised urea (which provides plant nutritional needs while reducing NH₃ emission levels). As a result, sales of stabilised urea have increased, although they still account for only 1% of N fertilisers sold.

Different fertilisers provide plant nutrition via different (bio)chemical routes. Urea (NH₂)₂CO is converted to NH₃ or, when soil moisture conditions are good, to NH₄⁺. When soil moisture conditions are marginal or poor, NH₃ persists in the soil, causing NH₃ toxicity to germinating seedlings. Potential volatilisation of NH₃ becomes a concern when soil pH increases after the surface application of urea. A temporary increase in soil pH promotes the formation of NH₃ gas at the soil surface, leading to increased NH₃ volatilisation. Volatilisation is therefore potentially greatest on sandy soils and lowest on fine-textured clay soils (Herold *et al.*, 2021).

Chemical fertilisers, such as urea or CAN, are most commonly applied in granular or pill form. Crops take up nutrients through their roots, so surface application

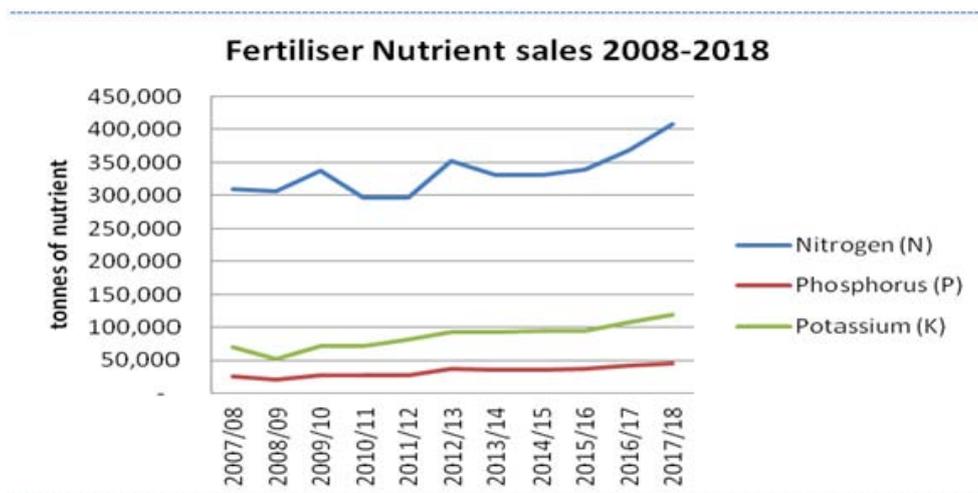


Figure 1.2. Fertiliser sales in Ireland.

of granular fertilisers relies on moisture to dissolve the fertiliser and transfer the nutrients to plants' subsurface root systems. Alternatively, chemical fertilisers can be applied in a liquid form using a bar sprayer that projects larger liquid droplets on top of the plants. Proponents of this method believe that it offers better control over the distribution pattern, especially in windy conditions, as well as more rapid penetration of nutrients to the subsurface root systems, promoting more rapid uptake by plants. Liquid fertilisers can also be applied using a fine particle sprayer that sprays pressurised liquids with a wide distribution pattern; however, this type of system is less common than granule or liquid bar sprayers (Department of Agriculture, Food and the Marine, 2021a).

Fertiliser use is one of the largest variable costs on Irish farms, currently accounting for over €400 million annually, calculated from the US dollar value of imports as per "Ireland imports of fertilizers" (Trading Economics, 2023) using an exchange rate of €1.000 to US\$1.142. The dominant N fertiliser used in Ireland is CAN. Increasing the efficiency of N use is important for achieving the Food Vision 2030 targets of ensuring sustainability both financially and environmentally. Urea amended with urease inhibitors can decrease NH₃ emissions by up to 70%, while generating similar yield responses to CAN (Richards *et al.*, 2016). Use of a nitrification inhibitor could further increase yields and reduce losses. Reducing national CAN use by 36%, via substitution of lower-cost stabilised urea and inhibitors, could ultimately save Irish farmers €9.5 million annually (Department of Agriculture, Food and the Marine, 2021a).

Agricultural sustainability in relation to nitrogen

The Common Agricultural Policy (CAP) is a mechanism used to underpin a stable supply of food, to support farm incomes and to promote environmentally sustainable farming within EU Member States. In June 2021, the European Commission reached an agreement on CAP reforms to be implemented in 2023 (pending final agreement among Member States). The agreement incorporates a significant focus on agricultural sustainability, climate and biodiversity. The agreement includes provisions that 40% of the CAP budget will have to be climate relevant and strongly support the general commitment to dedicate 10% of the EU budget to biodiversity

objectives (European Commission, 2022b). It requires each Member State to develop a strategic implementation plan, which in Ireland's case includes an "Eco-Scheme" that integrates conditional payments for defined agricultural practices that are beneficial to the climate and the environment, and that go beyond established baselines. Examples of the types of activities supported include voluntary limitations on stocking intensity and limits on the application of chemical N fertiliser (Department of Agriculture, Food and the Marine, 2022).

The EU's new Fertiliser Product Regulation (FPR) (Regulation (EU) 2019/1009) came into effect in July 2022 (European Union, 2019). This harmonises specifications for all CE-labelled fertilisers (new organic fertilisers as well as conventional fertilisers) intended to be traded in the EU and European Economic Area. The FPR is intended to create a harmonised internal market for new fertiliser products derived from organic materials, such as biodegradable waste from the food industry, from non-animal agricultural by-products, and from biochars and ash, where historically an end-of-waste designation may have been difficult to achieve, as product specifications were less clear. Under the FPR, if a fertiliser product meets the specifications and is CE marked, it is no longer considered waste.

The FPR establishes specifications for product function categories (PFCs), such as an organic fertiliser, a growing medium, a liming agent or soil amendment, a plant growth inhibitor or stimulant, or a combination product. PFC specifications generally address minimum performance criteria and limits on heavy metals and toxic chemical composition, and also limits on microbial loads. They establish specifications for component material categories (CMCs) from which the fertiliser products can be sourced (e.g. virgin materials, plant and plant extract materials, composts, digestates, microorganisms, nutrient polymers, derived products and by-products, and thermal oxidation materials, such as biochar or ash). Currently, organic fertilisers from animal by-products (ABPs) such as slurries or meat and bonemeal ash are explicitly excluded from the scope of the new FPR unless they have been designated to meet "end-of-process" definitions governed separately under the Animal By-products Regulation (Regulation (EC) No. 1069/2009) (European Union, 2009), which are given legal effect in Ireland by S.I. No. 187 – European

Union (Animal By-products) Regulations 2014 (Irish Statute Book, 2014b). The regulations define ABPs as materials of animal origin that people do not consume, such as skin, bones, horns and hooves, blood, fat and offal, including catering waste and manures. The ABP Regulations classify material into three categories. The highest risk material is category 1, which includes tissues from infected animals and imported catering waste. Category 2 materials are intermediate-risk materials, which include manures. Category 3 materials are low-risk materials, such as hides, hooves, bones or other parts sourced from healthy animals. The ABP Regulations establish acceptable procedures for management or recovery that are related to the classification of material. For example, category 1 material can be disposed of or subjected to energy recovery only via combustion, while material in categories 2 and 3 can be used for biogas generation or land spreading as soil conditioners, provided that measures are taken to protect animals intended for human consumption against disease. The ABP Regulations define the “end of process” concept, whereby derivatives of category 2 and category 3 materials can be placed on the market, provided that the end of process has been defined by the European Food Safety Authority and that the derivative products meet the specifications and regulatory requirements governing such derivatives.

Actions are under way within the scope of these regulations to define “end of process” for a range of ABP materials, and the outcome of this work will determine the authorisation pathway for the fertilising compounds recovered by the modular technologies developed during this project. This is likely to be through DAFM, the licensing body for these regulations in Ireland. Potentially toxic materials such as municipal wastewater sludges are precluded as an acceptable CMC, although precipitated salts derived from such materials may be accepted, which is also of relevance to this project.

The environmental impact of chemicals and manure on farms is governed by the Nitrates Directive (91/676/EEC), which in Ireland is implemented by S.I. No. 113/2022 – European Union (Good Agricultural Practice for Protection of Waters Amendment) Regulations 2022 (Irish Statute Book, 2022). This directive incorporates regulatory limitations on the application of N to farms and other measures for the protection of water and air quality, and also obligates

each Member State to periodically prepare a national nitrates action programme (NAP) that incorporates some other (non-regulatory) measures, such as codes of good practice that promote the use of good farming practice to manage the application of livestock manures and other fertilisers. Some of the regulations incorporated into Ireland’s 2022 NAP (Department of Housing, Local Government and Heritage, 2022b) include:

- an annual limit of 170 kg of nitrogen equivalents (N_{eq}) per ha, unless the landowner has been approved for a derogation allowing spreading of up to 250 kg N_{eq} ha⁻¹ under certain manure management conditions;
- an obligation to use low-emission slurry-spreading equipment (Department of Agriculture, Food and the Marine, 2021b);
- setting periods when land spreading is prohibited because of risk;
- setting capacity levels for the storage of livestock manure;
- an obligation to avoid spreading organic fertilisers when land is waterlogged or flooded or when heavy rain is forecast within 48 hours, when land is frozen or covered with snow or where the ground slopes steeply and there is a risk of water pollution;
- an obligation to maintain specified distances from waterways and water supplies.

Under the 2022 NAP, the national N allowances are to be reduced by 10% nationally from March 2022 and by a further 5% from 1 January 2024, pending review, scientific advice and water quality trends.

1.2.2 Other sources of nitrogen species and related greenhouse gas emissions in Ireland

Wastewater and wastewater treatment

Over 1000 WWT plants operate in Ireland, with a population equivalent of 4.6 million being treated in 2017, and an average treatment efficiency of c.56% for N. There are also > 500,000 individual package treatment plants or septic tanks, which generate over 5800 tonnes of N per annum, with an average treatment efficiency rate of 67% for N. Septic tank systems have the potential to cause certain impacts

in small streams, particularly in areas of poor permeability during low-flow periods, where there is little baseflow for dilution. In areas with a high density of septic tank systems ($> 19 \text{ km}^{-2}$), annual nutrient emissions contributions can be up to 13% of the total for N. Finally, there are over 1500 combined sewer overflows in Ireland that convey the wastewater and stormwater generated from urban areas and drained by combined sewer systems, estimated to account for 1% of N emissions (10% of wastewater N emissions) (Mockler *et al.*, 2017).

Landfill

Landfill leachates generate N discharges that put pressure on inshore water quality. The project team discussed landfill operations with different operators in Counties Limerick, Tipperary, Clare, Cork and Mayo, which highlighted that effluent management practices vary depending on individual circumstances. Some landfills that generate leachates from new waste cells with high NH_3 concentrations (e.g. $> 1500 \text{ ppm}$) deployed on-site WWT to reduce the N load prior to discharge. Other landfills generating leachate from more mature cells have lower N loads (e.g. 35–150 ppm). These operators discharge leachates directly or collect them in local lagoons and transport them to municipal WWT plants. Discussions highlighted that an individual operator can incur annual costs as high as €250,000 for collection and transport. The principal driver of transport to WWT plants is the requirement for N removal. There may also be requirements to reduce levels of dissolved metals, hydrocarbons or other contaminants. Anecdotal discussions with industry stakeholders indicate that, in many cases, plants are operating at or near capacity, periodically constraining the ability to accept deliveries of landfill leachates, which can give rise to storage pressures and the risk of discharging untreated leachates.

Several policy and regulatory initiatives govern the management and discharge of N waste streams. The EU Circular Economy Action Plan 2020 (European Commission, 2020b) promotes Europe's agenda for sustainable growth and targets stages along the entire life cycle of products – including sustainable design, integration of manufacturing and circular recycling/re-use/recovery processes – fostering sustainable consumption and aiming to ensure that resources

used are kept in the EU economy for as long as possible. The action plan introduces legislative and non-legislative measures to reduce waste, targeting sectors with high potential for circularity, including the food, water and nutrients sectors.

The Industrial Emissions Directive (IED) (Directive 2010/75/EU) is the main EU instrument setting environmental standards for large-scale industrial activities (European Union, 2010). It is given effect in Ireland via a series of instruments, including S.I. No. 148 – European Union (Industrial Emissions) Regulations 2013 – and S.I. No. 138 – Environmental Protection Agency (Industrial Emissions Licensing) Regulations 2013. It aims to protect human health and the environment by reducing harmful industrial emissions via a licensing system that establishes maximum binding emission limit values (ELVs) for discharges based on the application of best available techniques and a corresponding inspection, monitoring and reporting protocol to underpin compliance. The objective is an integrated approach to prevent and control pollution in a coherent manner. All installations engaged in activities covered by the IED (e.g. incineration plants, large landfills and intensive agriculture) are required to hold either an industrial emissions or integrated pollution control licence issued by the EPA, which regulates industrial discharges to water, air and soil (EPA, 2022). Emissions from activities that do not qualify as licensable industrial emissions or integrated pollution control activities are regulated by local authorities under this framework.

The waste management policy (European Commission, 2022c) incorporates several pieces of relevant legislation that aim to protect the environment and human health, and enable the EU's transition to a circular economy. Components of the policy include the Landfill Directive (Council Directive 1991/31/EC) and the Waste Framework Directive (WFD) (Directive 2008/98/EC as amended by Directive (EU) 2018/851), which is given legal effect in Ireland by S.I. No. 323 of 2020 – the European Union (Waste Directive) Regulations 2020 (Irish Statute Book, 2020b). The WFD sets definitions of waste and requires that waste be managed without endangering human health; without risk to water, air, soil, plants or animals; without causing a nuisance through noise or odours; and without adversely affecting places of special interest. The WFD lays down the accepted waste processing hierarchy that prioritises (i) waste prevention

followed by (ii) preparation for re-use, (iii) recycling, (iv) recovery and, as a last resort, (v) disposal. The WFD establishes Member State processing targets and monitoring and reporting protocols for various categories of waste, and it also introduces the “polluter pays principle” and the “extended producer responsibility” concept.

The WFD defines a by-product as a substance or object resulting from a production process, the primary aim of which is not the production of that item. It also establishes the “end-of-waste” concept that defines when waste ceases to be waste and becomes a product or secondary raw material. An “end-of-waste” status is achieved when waste has undergone a recovery operation and complies with the following specific criteria:

- The substance or object is commonly used for specific purposes.
- There is an existing market or demand for the substance or object.
- The use is lawful, and the substance fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products.
- The use will not lead to overall adverse environmental or human health impacts.

1.3 Project Objectives

Managing undesirable N and related GHG discharges to waters and the atmosphere is required to maintain water quality and air quality fit for humans, flora and fauna. The European Green Deal, published by the European Commission in December 2019 (European Commission, 2022d), is an overarching sustainability agenda that promotes the efficient use of resources, including nutrients like N, by moving to a clean, circular economy, while cutting pollution and restoring biodiversity. It establishes targets for reductions in carbon footprints and GHG emissions (e.g. in the transport, buildings and industry sectors), while also targeting job creation and climate action, including reforestation and enhancing the contribution of soils to carbon sequestration. The Deal outlines the

investments needed and financing tools available in the context of a socially just and inclusive transition.

The Irish Department of the Environment, Climate and Communications (2021a) Climate Action Plan sets out the actions that the Irish Government intends to take to address the sustainability objectives outlined in the European Green Deal and the Circular Economy Action Plan, and other elements of a broader sustainability initiative. Ireland’s Climate Action Plan provides that the government will lead the transformation from waste management to circular economy practice through the delivery of a new national policy, including through the identification and introduction of measures to address key regulatory barriers to the development of the bioeconomy, for instance by exploring opportunities to establish “end-of-waste” criteria for certain biowastes.

This project focused on the development of a flexible method to improve the management of ammoniacal streams of varying characteristics by the recovery of solids and NH_3 to reduce concentrations in effluent discharges. Economic sustainability was addressed by the use of processes that require low capital and energy inputs and by recovering solids and NH_3 as saleable renewable energy and nutrient products, thus offsetting incremental processing costs. The proposed method aimed to contribute to achieving:

- air and water quality objectives by reducing N species in effluent discharges to meet relevant licence or regulatory requirements and to mitigate detrimental effects;
- waste management and circular economy objectives by reducing requirements for waste disposal in favour of recovery;
- Climate Action Plan and European Green Deal objectives by decreasing soil GHG emissions arising from intensive fertiliser practices, reducing methane emissions via improved slurry management protocols, and reducing carbon intensity in the energy mix and in fertiliser production;
- CAP objectives by maintaining the productivity while improving the sustainability of agriculture.

2 Technology to Address Nitrogen and Greenhouse Gas Emissions

The project focused on identifying a flexible, economically sustainable method to improve the management of ammoniacal streams of varying characteristics by recovering bulk solids for renewable energy generation and NH_3 for producing sustainable nutrient products. It considered the use of existing technologies as well as the development of novel technologies that could be integrated into a modular system, capable of being adapted to fit the varying processing demands of different ammoniacal streams.

2.1 Nutrient Accumulation, Release and Recovery

Existing technologies used to mitigate N and GHG emissions from ammoniacal waste streams incorporate a combination of different functional steps, such as nutrient accumulation, release and/or recovery (Mehta *et al.*, 2015). The requirement for this multi-phase approach is driven by low N concentrations in waste streams ($1\text{--}200\text{ mg L}^{-1}$) or the presence of N in forms not readily available, which affects recovery efficiency.

When nutrients are highly diluted, they first need to be concentrated, and then release techniques are used to mobilise and recover them as concentrated products for beneficial use. Biological, physical and chemical techniques have been used for nutrient accumulation to recover soluble N, P and K at low nutrient concentrations ($2\text{--}20\text{ mg L}^{-1}$) from waste streams (Neethling *et al.*, 2010; Sengupta *et al.*, 2015). Physical technologies, for instance using evaporation, membranes or air stripping, sometimes have high energy and material input requirements and therefore have a negative environmental impact (Bradford-Hartke *et al.*, 2015).

2.1.1 Bioaccumulation

Common nutrient-accumulating microorganisms include Proteobacteria, such as polyphosphate-accumulating organisms (PAOs), purple non-sulfur bacteria and cyanobacteria (Yuan *et al.*, 2012). Yuan

et al. (2012) reported P removal of up to 90% from various industrial wastewaters using Proteobacteria in the enhanced biological phosphate removal (EBPR) process. The EBPR requires alternating anaerobic, aerobic and anoxic conditions, so that the uptake of P by microorganisms is above normal metabolic requirements. In more recent years, Keating *et al.* (2016) reported P removal of 78% during high-rate, low-temperature AD. Phosphate-rich sludge with PAOs can be separated from the wastewater by settling, and nutrients can then be released and recovered from the settled sludge. The bioaccumulation of nutrients using PAOs is perceived to be not viable for some applications because of the low ratio of carbon to other nutrients of some streams. In these cases, the use of autotrophic microorganisms, such as algae, for the accumulation of nutrients may be more suitable.

Algae remove both N and P from wastewater streams via assimilation for their growth without oxygen consumption (Campos *et al.*, 2019). Dry algal biomass can contain up to 2% N and 3.3% P (Powell *et al.*, 2008). They are also valuable sources of biomass for energy because of their high growth rates compared with terrestrial plants and their ability to capture large quantities of atmospheric carbon dioxide (CO_2). After accumulation, the nutrient-rich algae can be further processed biologically or thermochemically to recover energy and nutrients or may be used directly as an animal feed or a fertiliser. Supplementation of livestock feed with some forms of algae has been reported to reduce enteric methane emissions (Kinley *et al.*, 2016). Removal of nutrients from wastewater using algae is usually carried out in a raceway pond because of the low capital costs compared with using a photobioreactor. However, a barrier to this strategy is the large footprint of these ponds, which makes their use conditional on the availability of land.

2.1.2 Physico-chemical accumulation

In this approach, soluble nutrients and nutrients bound to colloids (mixtures with particles ranging between 0.01 and $1\ \mu\text{m}$) are precipitated as solids

and separated by settling in clarifiers. The most used coagulants for accumulation of P from dilute wastewater are aluminium- or iron-based coagulants. Other coagulants, such as calcium, natural and synthetic organic polymers, and pre-hydrolysed metal salts, such as polyaluminium chloride and polyiron chloride, are also used, but these generally have a relatively high cost. Once suspended particles have flocculated into larger particles called sludge, they can usually be separated from the treated water by sedimentation. This approach is heavily used as part of a multi-point dosing process for controlling P discharge from sewage treatment plants, for example. In addition to nutrient removal, the chemical coagulation process can also remove organic matter, bacterial pathogens, viruses and inorganic species, such as arsenic and fluoride. High operating costs due to the use of chemical substances, high amounts of sludge generation and downstream management costs are reported as the main disadvantages of chemical accumulation by precipitation (Mehta *et al.*, 2015). The downstream processing of the sludge through AD could also be problematic because of the inhibitory effects of some of the coagulants or flocculants used for nutrient accumulation. Biological treatment of the sludge is, however, necessary to release nutrients, such as N and P, that are not directly bioavailable for plant uptake.

Adsorption can be used as a method for the accumulation of NH_3 from wastewaters or gaseous emissions. For example, Halim *et al.* (2010) report that organic contaminants and ammoniacal N are two of the problematic parameters in landfill leachate treatment. Common features of stabilised leachate are high concentrations of ammoniacal N ($3000\text{--}5000\text{ mg L}^{-1}$), moderately high levels of chemical oxygen demand (COD) ($5000\text{--}20,000\text{ mg L}^{-1}$) and low ratios of biochemical oxygen demand (BOD) to COD (<0.1). Biological treatment of landfill leachates has been shown to be very effective in the removal of organic matter from leachates produced in the initial time period after landfill closure (Berrueta and Castrillon, 1992), when the BOD-to-COD ratio in the leachate is high. This ratio decreases with the age of the landfill; the process is less effective with time because of the presence of refractory organic matter (Li *et al.*, 1999). At this point, physico-chemical processes may become more appropriate than biological treatment.

Some of the adsorbents utilised include activated carbon, zeolite, activated alumina or low-cost adsorbents such as limestone, rice husk ash and peat. Combinations of organic and inorganic pollutants that exist in streams, such as landfill leachate, require adsorbents with the ability to remove both organic and inorganic species. Zeolite is widely used as a natural ion exchanger to remove NH_3 and inorganic pollutants from leachate or other wastewaters. Gao *et al.* (2005) found new zeolite–carbon composite materials, which combine the excellent properties of both zeolites and carbon. The surface of zeolite is hydrophilic with regularly aligned molecular-level pores and cationic exchange ability, which makes it a good adsorbent for metallic ions and catalysts. A composite adsorbent medium may be more effective in removing all contaminants from media such as landfill leachates.

Reverse osmosis (RO) is a water purification process that applies pressure to an aqueous solute on one side of a partially permeable membrane, overcoming osmotic pressure and enabling purified water to pass through to the other side (Shin *et al.*, 2020). The separation process concentrates ions and larger molecules, bacteria and suspended particles in the retained solute. RO is used in both industrial processes and the production of potable water (e.g. desalination applications), and a range of selective membranes are available for different applications. Shin *et al.* (2020) investigated the use of RO to separate NH_3 from treated AD digestates using an energy-saving polyamide membrane at various pH levels and pressures ranging from 5.2 to 19.0 bar. They reported that hydraulic operating conditions (i.e. pressure and flux settings) did not affect total ammoniacal nitrogen (TAN) rejection efficiency, but pH had major impacts due to changes in $\text{NH}_4^+/\text{NH}_3$ speciation and membrane surface charge. They also reported that, by maintaining the pH at 6, TAN was reduced from 47.3 mg L^{-1} in the solute to $\leq 1\text{ mg L}^{-1}$ in the permeate, a standard that meets typical potable water guidelines. At pH 6, TAN rejection efficiency was optimal, at 99.8%. At higher or lower pH, TAN rejection rates decreased (rejection efficiency at pH 5.4 was 98.4% and at pH 3.9 was 96.6%). These findings suggest that RO may be a useful technology for concentrating NH_3 to levels that can be processed and for “polishing” low-concentration ammoniacal liquors in circumstances where a very high standard of discharge effluent is required.

Nutrient release methods are used either to release nutrients that have been accumulated or to directly extract them into a recovered product. Biological, chemical and thermal techniques are commonly used for nutrient release.

2.1.3 Nitrification and denitrification

This is a two-stage biological process and is a mature and well-proven biological technology widely deployed in WWT plants.

First, nitrifying bacteria convert $\text{NH}_4^+/\text{NH}_3$ to NO_2^- and then to NO_3^- . These nitrification reactions are coupled and proceed rapidly to NO_3^- ; accordingly, NO_2^- levels at any point are usually low. Nitrifying bacteria are strict aerobes, requiring free dissolved oxygen at levels of $\geq 1.0 \text{ mg L}^{-1}$. At dissolved oxygen concentrations of $< 0.5 \text{ mg L}^{-1}$, their growth rate is minimal. Nitrification requires a long retention time, a low food-to-microorganism ratio, a high mean cell residence time and adequate buffering (alkalinity). Optimum pH for nitrifiers ranges between pH 7.5 and pH 8.5. Most treatment plants can operate at a pH ranging between 6.5 and 7.0; however, at a pH of < 6.0 , nitrification stops. Nitrification consumes 7.1 mg L^{-1} of alkalinity (usually as CaCO_3) for each mg L^{-1} of $\text{NH}_4^+/\text{NH}_3$ oxidised. Ideal water temperatures range between 30°C and 35°C . At temperatures of $> 40^\circ\text{C}$, nitrification rates fall to near zero, while at temperatures of $< 20^\circ\text{C}$ nitrification proceeds at a slower rate but will continue at temperatures of $\geq 10^\circ\text{C}$. Nitrification can be inhibited by the presence of toxic compounds, such as cyanide, thiourea and phenol, and heavy metals, such as silver, mercury, nickel, chromium, copper and zinc. Nitrification can also be inhibited by excess nitrous acid and NH_3 .

Denitrification is the second stage of the process. During this stage, nitrates are biologically reduced to nitrogen gas (N_2) by facultative heterotrophic bacteria. Denitrifiers need a carbon source, which is normally organic matter in the wastewater stream provided that the stream's C availability is sufficient to drive the denitrification reaction. Supplemental carbon can be provided by the addition of methanol or acetic acid, which are sometimes used for this purpose. Denitrification occurs when oxygen levels are depleted and NO_3^- becomes the primary electron acceptor. Therefore, the process is performed under anoxic conditions, that is, when the dissolved oxygen

concentration is $< 0.5 \text{ mg L}^{-1}$, and ideally $< 0.2 \text{ mg L}^{-1}$. NO_3^- is reduced to N_2O and, in turn, to N_2 , which escapes into the atmosphere.

Conditions that affect the efficiency of denitrification include NO_3^- concentration, the extent of anoxic conditions, the presence of organic matter, pH, temperature, alkalinity and the effects of trace metals. Denitrifying organisms are generally less sensitive to toxic chemicals than nitrifiers and can recover from toxic shock loads quicker than nitrifiers. Temperature affects the growth rate of denitrifying organisms, with greater growth rates at higher temperatures. Denitrification can occur at temperatures ranging between 5°C and 30°C .

The nitrification/denitrification process is designed to convert problematic N species to inert N_2 gas, and it can be energy intensive and precludes valorisation of the N fraction.

2.1.4 Ammonia stripping via pH and/or temperature shifts

NH_3 is present in aqueous solutions as two N species that are in equilibrium: an uncharged (unionised) NH_3 species and an NH_4^+ ion, which is positively charged. The equilibrium is highly dependent on solution temperature and pH (Figure 2.1). By increasing temperature to between 60°C and 80°C and/or increasing pH to between 8 and 12, the thermodynamic equilibrium of NH_4^+ is shifted to the gaseous phase (i.e. to NH_3), enabling NH_3 to be stripped from the solution (Jamaludin *et al.*, 2018).

Several variants of the stripping approach exist but generally all involve flowing a gas (usually air or

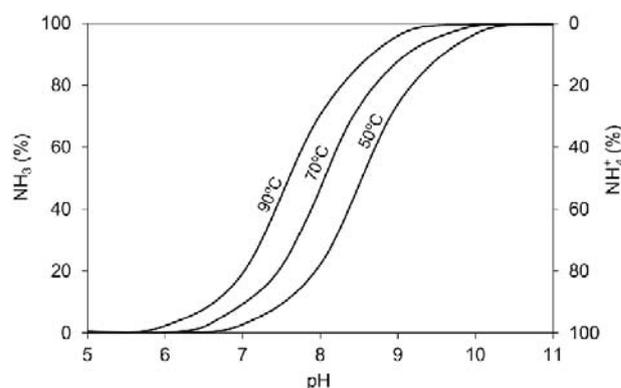


Figure 2.1. Effects of pH and temperature on ammoniacal equilibrium in water.

steam) counter to the ammoniacal liquor current, with the gas being either circulated in a spray tower or supplied through diffusers in a liquid column to maximise surface area contact between the carrier gas bubbles and the liquid phase. At moderate intensity, air stripping achieves typical NH_3 removal rates of 20–40%, depending on the operating conditions. At very high intensity (e.g. well-filtered liquors stripped at ratios of air flow to liquid feed of $>2000\text{ m}^3\text{ air m}^{-3}$ liquid feed and a high sodium hydroxide (NaOH) load, elevating pH to >10.5), Baldi *et al.* (2018) achieved NH_3 removal rates of 60% over a 9-hour hydraulic residence time (HRT). Exiting NH_3 concentrations were c. 1000 mg L^{-1} from an influent concentration of c. 3000 mg L^{-1} .

2.1.5 Ammonia recovery via membrane separation and acid scrubbing

In stripping applications, the vaporous NH_3 that evolves is highly diluted in the stripping medium, usually to a few mg m^{-3} , given the high air-to-liquid ratios used. This limits its value unless the resultant vapour is routed for recovery, typically by contacting the ammoniacal vapour with an acid solution to yield a dilute solution of the corresponding ammonium salt. One example involves increasing the pH and the temperature in an ammoniacal liquor to shift the equilibrium towards gaseous NH_3 , which is passed through a gas-permeable membrane and diffused into a sulfuric acid solution, where it is trapped as

soluble NH_4^+ and precipitated as an ammonium sulfate fertiliser. The process can potentially be deployed on a moderate scale and the energy and/or reagent inputs are relatively modest (Kinidi *et al.*, 2018). de Leon Perez and Eriksson (2002) highlight that this system is used to treat wastewaters outside Stockholm, and Blue-Tec in the Netherlands advertises the use of this process to treat different ammoniacal wastewaters (Figure 2.2) (Blue-Tec, 2022).

2.1.6 Ammonia recovery in aqueous solution

As an alternative to recovery of vaporous NH_3 in an acid solution, NH_3 can be recovered in a separate aqueous solution at low concentration. At 25°C , the solubility limit of NH_3 in water is 32 mg L^{-1} (Flinn Scientific, 2022), and aqueous NH_3 solutions (with NH_3 concentrations ranging from 10% to 15%) are regularly used in commercial applications and are common ingredients in products such as NH_3 -based household cleaning agents (Young, 2003). To determine the feasibility of capturing vaporous NH_3 in an aqueous solution at low concentration, the project team rigged up a small vacuum pump (Figure 2.3) to extract vapour from a rotating mixer, into which 0.5 kg of calcium hydroxide ($\text{Ca}(\text{OH})_2$) (slaked lime) was added to 50 L of pig slurry liquor, sparging the resultant NH_3 vapour through 1 L of water. It was noted that, over 90 minutes, the dissolved NH_3 concentration grew to 2%. While this was a scoping exercise to determine the ease or difficulty of capturing

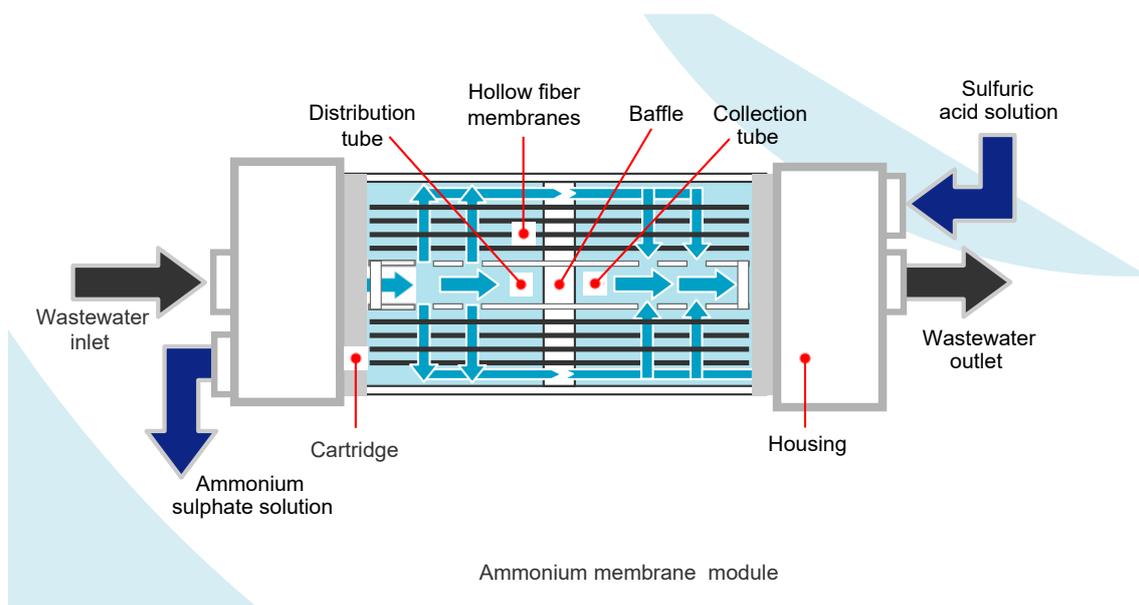


Figure 2.2. An NH_3 -stripping process (www.blue-tec.nl).



Figure 2.3. Sparging ammonia vapour through water column.

vaporous NH_3 in solution rather than a rigorous scientific test to quantify the volumes captured, the result indicates that it is feasible to capture vaporous NH_3 in a low-concentration solution, and no obvious barriers to achieving a 10–15% concentration, with the appropriate scaling and engineering, were noted.

The project team also considered alternative methods for the recovery of vaporous NH_3 in a more highly concentrated form. NH_3 liquefaction is currently undertaken commercially via pressurisation and condensation (Chemical Guide, 2013). The pressure–temperature phase diagram for NH_3 indicates that, at c. 10 bar and moderate temperature (c. 10°C), NH_3 gas can be compressed into a liquid state, whereas air (the primary component of the vaporous stream) remains in a gaseous state. It was hypothesised that the liquefaction procedure could be applied to recover vaporous NH_3 in more highly concentrated form, and this concept was discussed with engineers from O’Dwyer Refrigeration Ltd in County Tipperary (who have experience with several large-scale NH_3 -based refrigeration installations). O’Dwyer Refrigeration indicated that it may be possible to liquefy NH_3 from a

gaseous stream via compression and condensation, but noted that (i) there would be health and safety considerations because NH_3 is combustible and methods would have to be implemented to ensure that it did not ignite in the presence of air; (ii) NH_3 is highly corrosive and so compressor heads would have to incorporate specialist materials to avoid corrosive degradation and excessive maintenance; and (iii) specialised equipment would be required to collect, store and transport NH_3 in a liquid state (e.g. those currently used for anhydrous NH_3). The engineers highlighted that the presence of moisture in the ammoniacal vapours may result in coincidental condensation of moisture and limit the concentration that could be achieved, and therefore that a method for moisture removal may be required.

Dinh and Kim (2021) reported on various methods for the removal of moisture from gasses, noting that condensation techniques that physically separate condensed moisture from the gaseous medium can reduce levels of soluble fractions (e.g. NH_3) that condense with the moisture. Removal of moisture in a solid state (e.g. adsorption on to a desiccant) minimises the coincidental condensation of the soluble fractions. Calcium oxide (CaO) can be used to remove moisture from NH_3 vapour because CaO reacts vigorously in an exothermic reaction with water to form $\text{Ca}(\text{OH})_2$ and does not react with NH_3 .

Permeation techniques are also available for moisture removal. These involve deploying a water-permeable membrane with moist gas on the supply side and dry gas on the receiving side to generate a moisture flux across the membrane, reducing the moisture content in the retentate. It may be possible to achieve a higher NH_3 concentration (Wooram *et al.*, 2021) by generating an NH_3 flux across an NH_3 -gas-permeable membrane, leaving the moisture and other gaseous constituents in the feed stream.

While a variety of techniques for the recovery of NH_3 concentrates from gaseous streams are potentially available, additional work will be required to identify a specific technique that can be adapted and engineered into a system for nutrient recovery.

2.1.7 Struvite precipitation

The precipitation of NH_3 by the crystallisation of struvite is a further method for the recovery of

NH_3 (and phosphates) that offers many potential advantages, not least that it can theoretically achieve very low NH_3 equilibrium concentrations in solution. Struvite is one of the few NH_3 salts that is insoluble in aqueous solutions, facilitating recovery as a mineral crystalline substance containing equal molar amounts of magnesium, ammonium and phosphate ($\text{NH}_4\text{MgPO}_4 \cdot 6\text{H}_2\text{O}$). It has a measurable nutrient value, with the NPK value (which reflects the relative amounts of N, P and K) being 6-29-0, and has a potential application as a slow-release fertiliser.

Struvite precipitation is widely studied as a chemical precipitation reaction for nutrient recovery (Campos *et al.*, 2019). The speciation chemistry, however, is complex, being highly dependent on achieving an appropriate stoichiometric relationship between the reactants, on the type of magnesium (Mg) and phosphate ions supplied, and on pH, temperature and the avoidance of inhibitory effects caused by process media contaminants. In practice, struvite precipitation from ammoniacal streams, such as livestock slurries, is complicated by stoichiometric imbalances between the reactants, and the presence of colloidal solids in the liquor and dissolved CO_2 , both of which complicate and reduce the availability of NH_4^+ ions for reaction in solution. For example, Li *et al.* (2012) report an 85% recovery of ammoniacal N from dilute wastewater by adding reactants at a level that keeps the ratio of Mg to NH_3 to phosphate at 1:1:1, whereas Le Corre *et al.* (2009) report the recovery of 90% of soluble phosphates but only 30% of soluble ammonium from waste streams because of the stoichiometric imbalance caused by a large molar excess of ammoniacal N that was not taken up by the addition of extra P and Mg, causing excess NH_4^+ to remain unrecovered in solution.

Generally, the concentration of Mg is limited in ammoniacal streams such as wastewater, leachates or slurries. It needs to be added to ensure maximum efficiency of N and P recovery (Sheets *et al.*, 2015) and needs to be presented to the working liquor fully solubilised, which complicates processing and limits the choice of Mg to soluble Mg salts such as chloride or nitrate salts. While these reagents have been successfully used to precipitate struvite from municipal wastewaters and filtered livestock slurries, their use introduces counter-ions that may not be desirable in the treated liquor depending on the sensitivity of the discharge environment to chlorides or nitrates. Use

of magnesium oxide (MgO)/hydroxide may provide an alternative source of Mg, but solubility is limited at moderate pH. An acceptable method for ensuring solubility is to use MgO as the Mg source for struvite precipitation.

Even with soluble sources of Mg, the high CO_2 concentrations in pig slurry result in a substantial portion of the Mg forming its carbonate and/or hydrogen carbonate salts, which manifests as there being (typically 1.5 times) higher stoichiometric Mg levels than are required to yield struvite. This increases the reagent requirement if struvite crystallisation is to be used for NH_3 remediation unless CO_2 is first removed and/or NH_3 concentrations are reduced.

Struvite solubility is pH sensitive, as are the speciation and solubility of its three reactants in solution. Phosphate is typically supplied as phosphoric acid, the fully dissociated tribasic phosphate form of which is favoured at high alkaline pH. By contrast, Mg salt solubility is usually favoured by low pH, as is the conversion of NH_3 to NH_4^+ ions. The choice of operating pH for struvite crystallisers is therefore based on a trade-off between these equilibria and minimising struvite solubility. Typically, a pH at or about the pK_a of the NH_3 - NH_4^+ equilibrium in aqueous solution is optimal (i.e. a pK_a of 8.5–10). At this level, an alkali source is required to maintain an optimum pH for struvite precipitation. Sodium or potassium hydroxide solutions are typically used for such purposes.

As with all crystallisation systems, the kinetics of crystal growth and the settling properties of the resulting crystals are enhanced by seeding. In a continuous system, the throughput, or HRT, will be influenced by the operating concentration of the magnesium/phosphate stream input to the crystalliser.

Where NH_3 concentration is the limiting factor in achieving a liquor fraction acceptable for discharge, struvite crystallisation as a means of NH_3 removal is a viable option if starting ammonium levels are sufficiently low. In liquors with high NH_3 concentrations (e.g. ammoniacal streams such as livestock slurries, where NH_3 concentrations can exceed 3000 ppm), recovery via struvite is rarely applied in practice, as the requirement for an external source of P and Mg (supplied at minimum at a molar equivalent to NH_3) results in inhibitory costs. In these circumstances,

a supplemental step to first moderate ammoniacal concentrations can make the struvite route a viable option, especially when the liquor has been preconditioned to maximise the efficiency of phosphate recovery and when struvite can be sold as a fertiliser product to offset the input costs.

The recent introduction of the new FPR (European Union, 2019) offers prospects for valorisation of recovered struvite as a commercial fertiliser. The mineralised struvite crystals degrade over time, releasing N and P slowly, which better matches crop

take-up requirements, minimising loss to atmospheric emissions or leaching. The crystalline character facilitates either broadcast application to grassland or incorporation into tillage soils, which may facilitate compliance with best practice application techniques promulgated in the 2022 NAP. A number of commercial technologies are advertised, including Aiprex, Crystal Green, PHOSPAQ, ANPHOS and PEARL, which are available around the world for nutrient recovery using struvite precipitation from the sludges or wastewaters (Li *et al.*, 2012).

3 A Modular System to Manage Ammoniacal Streams

The project focused on the design and testing of a flexible, integrated system that could be adapted to process different types of ammoniacal streams to achieve an acceptable standard of discharge that is compliant with the regulations relevant to receiving waterbodies of different sensitivity. Technology commercialisation and deployment, which are required to deliver environmental objectives, are dependent on economic sustainability. The approach prioritised the integration of technologies that recover energy and nutrient fractions in saleable forms, facilitating the offsetting of processing costs with revenues from product sales. To the extent possible, the project attempted to integrate mature and market-proven components with developmental technology to develop a system that could be evaluated at technology readiness levels 5–7 (European Commission, 2015). The project was undertaken during a period when COVID-19 restrictions were in force. In some cases, this precluded completion of all necessary tasks, especially those relating to productising process outputs, which, by their nature, are highly technical and require access to third-party expertise and resources.

Ammoniacal effluents can be broadly divided into low- and high-solid-load streams. The project selected landfill leachates and livestock slurries as representative examples the former and latter group, respectively. A modular system was designed, integrating technology that first separates bulk and colloidal solids (as required) and then conditions bulk solids for renewable energy recovery while partially clarifying and broadly standardising the physical character of the aqueous liquor effluents. These liquor effluents are then suitable for NH_3 stripping (using $\text{Ca}(\text{OH})_2$ to generate a pH shift), struvite precipitation and filtration and RO (as required), with the aim of achieving a standard acceptable for discharge.

3.1 Ammonia Recovery from Liquors with Low Solid Loads

Landfill leachates comprise the liquor effluents that percolate down through landfill cells, collecting soluble components from the buried waste as they go.

Organic solids are anaerobically digested, generating landfill gas, and bulk solids are filtered out as the liquors percolate, avoiding the need for bulk solid removal. NH_3 levels, metals composition and other characteristics of landfill leachates vary depending on the material in the landfill cell, the age of the cell, the levels of rainfall and other factors. As a rule, the level of residual organic matter is low, and solid content is relatively dilute and composed of small suspended particulate matter or colloidal solids. This low organic solid load precludes further energy recovery.

Depending on their character, leachates are sometimes directly discharged into receiving waterways or can be treated on-site or collected in lagoons for transport to municipal WWT plants. Discharge of landfill leachates is governed by industrial emissions authorisations, which incorporate ELVs for NH_3 and other compounds that vary depending on the specific circumstances of individual sites, taking into consideration the classification and sensitivity of the receiving area.

Technologies tested for treatment of low-solid-load landfill leachate streams include:

- liming (and lime settlement) to partially remove suspended solids;
- struvite precipitation to recover NH_3 and further reduce the level of colloidal solids;
- filtration and RO to “polish” liquors to a dischargeable standard.

A leachate sample was sourced from an active landfill cell in a County Limerick landfill exhibiting high NH_3 concentrations that required mitigation prior to discharge. At site, the leachate samples from new cells were collected in a lagoon and routed together with leachates from mature cells to a WWT plant deploying nitrification–denitrification technology prior to discharge into a local waterway. Periodically, excessive NH_3 loads exceeded the plant’s treatment capacity, which led to the risk of discharges exceeding licensed ELVs. High NH_3 loads generally resulted in an excessive demand for treatment reagents, giving rise to excessive WWT costs.

The operator's analytical services company prepared a monthly analysis of the (combined) leachate streams, which provides the input for periodic reports to the EPA. The July 2020 report showed that reported metal concentrations were within industrial emissions licence parameters (Table 3.1).

NH₃ levels ranged from 1700 to 1800 ppm during summer months, decreasing to 1400–1500 ppm during rainy winter months. To determine if the metals or organic load would interfere with the struvite reactions, bench-scale struvite precipitation tests were performed on each of the leachate samples. The bench-scale tests were run by adding Mg (via a MgP solution prepared by blending 320 g of formic acid in 1667 g of water with 396 g of phosphoric acid for every 140 g of MgO) at a 1:1 ammoniacal molarity. To overcome any potential process inhibitions, crystallisation pH was initially maintained at a level of 9.5 by adding sufficient NaOH solution (16% by weight). No obvious process inhibitions were observed. Over 24 hours, NH₃ concentrations were reduced from 1495 to < 150 ppm.

Subsequently, a pilot-scale struvite reactor was moved to the County Limerick site, where a continuous test

was run over a period of 2 weeks on blended leachate from the active cells (Figure 3.1).

The test protocol collected leachate from the lagoon in a 15 m³ buffer tank. Leachate was routed through an in-line mixer that injected an MgP solution into the leachate (prepared using the bench-scale test protocol described previously), adding Mg, P and ammonia at an ammoniacal molarity of 1:1:1. Process liquors were routed to an agitated nucleation tank, where NaOH solution was injected (16% NaOH by weight) to generate a pH shift. Process liquor from the nucleation tank then overflowed into a stirred tank reactor to complete struvite crystallisation. The flow of caustic solution was adjusted periodically to maintain pH within a range of 8.8–9.4 in the crystallisation tank. A pH probe provided constant pH monitoring. Struvite-laden process medium was routed to two sequential passive clarifiers, where the solid struvite was gravity settled and recovered (Figure 3.2), with partially clarified process liquors (at reduced NH₃ concentrations) routed back into the landfill holding lagoons. Each day, the system inlet valve was opened during working hours to introduce untreated leachate into the nucleation and crystallisation tanks. Subsequently, the inlet valve was closed, and agitation was maintained in the nucleation/crystallisation tanks overnight. This resulted in processing on a scale ranging between 1 and 1.5 m³ day⁻¹ and with an average HRT of 24 hours.

Each day, readings from the Pro Cal 2500 analyser (fitted with an SI AI162 NH₃ sensor) were calibrated against control samples with known NH₃

Table 3.1. Analysis of leachate streams from the County Limerick landfill site

Component	Concentration
BOD	521 mg L ⁻¹
COD	2980 mg L ⁻¹
TAN	1762 mg L ⁻¹
SO ₂	497 mg L ⁻¹
Cl	1820 mg L ⁻¹
TOC	897 mg L ⁻¹
CN ⁻	0.14 mg L ⁻¹
B	4917 µg L ⁻¹
Cd	< 0.5 µg L ⁻¹
Ca	70 µg L ⁻¹
Cr	168 µg L ⁻¹
Fe	2989 µg L ⁻¹
Pb	< 5 µg L ⁻¹
Mg	65 µg L ⁻¹
Mn	670 µg L ⁻¹
Hg	< 1 µg L ⁻¹
Ni	146 µg L ⁻¹
K	784 mg L ⁻¹
Na	1612 mg L ⁻¹
F	< 6 mg L ⁻¹
Z	26 mg L ⁻¹



Figure 3.1. Continuous struvite tests at the County Limerick landfill site.



Figure 3.2. Struvite collected from County Limerick landfill tests.

concentrations of 1, 10, 100 and 1000 ppm. Over 10 days, the NH_3 concentrations in the leachate inflow (as measured each morning) were compared with NH_3 concentrations in samples of processed media taken from the clarifier (as measured each morning and afternoon). All measurements were taken in triplicate. The processing protocol generated an averaging effect with respect to the character of the process media at discharge. The NH_3 concentrations in discharge media ranged from 96 to 100 ppm, while the concentrations in the input leachate ranged from 1476 to 1523 ppm. The results were validated by the landfill operator's external laboratory, which analysed a number of treated samples and confirmed that the NH_3 levels were reduced to ≤ 100 ppm. It was concluded that the struvite protocol successfully reduced the ammoniacal load to c.100 ppm, potentially underpinning greater stability in the WWT plant and more consistent compliance with licensed discharge ELVs, and enabled effective struvite recovery.

A second leachate sample was sourced from a mature (inactive) cell from a County Tipperary landfill that was not accepting new deposits of waste for disposal. Monthly NH_3 levels periodically reported to the EPA by the operator ranged between 80 and 150 ppm over the year, depending on the level of seasonal rainfall. Visually, liquor samples from the mature cells were a clear light-brown colour (suggesting low levels of colloidal organic matter) and dissolved metal levels were within the parameters specified in the industrial emissions licence. Leachates at this site were collected in a storage lagoon and transported for treatment to a municipal WWT plant prior to discharge. Periodically, owing to capacity constraints at the local WWT plant, loads had to be transported great distances to other WWT plants with available capacity. Transport and treatment gave rise to significant costs. Leachate samples were collected from the leachate lagoon and the bench-scale struvite protocol (as described above) was applied. Over an HRT of 4 hours, the NH_3 concentrations were reduced from 135 to < 50 ppm.

To further develop a treatment protocol for "light" leachates to facilitate local discharge, small commercial-scale water treatment filtration and RO units were procured to evaluate the use of these techniques as supplemental methods to further reduce NH_3 concentrations in landfill leachates (Figure 3.3). A leachate sample was sourced (2 m^3) in intermediate bulk containers from the wellhead on a mature (inactive) County Cork landfill site. Using the test protocols described above, NH_3 concentration was measured at an average of 38 ppm.

Slaked lime ($\text{Ca}(\text{OH})_2$) was applied to each sample at a rate of 0.5 kg m^{-3} and after agitation via manual stirring, and samples were left to settle for 1 day.

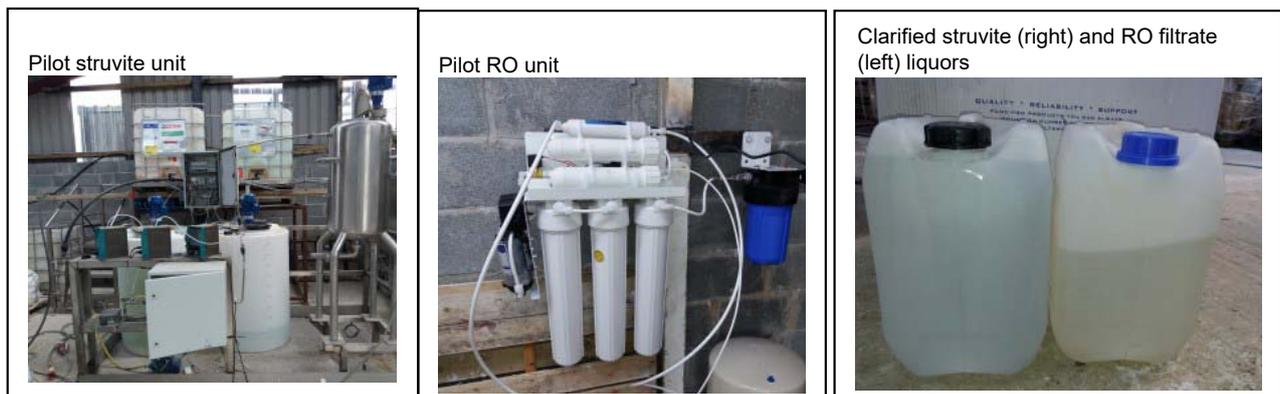


Figure 3.3. Images of test units and process results.

Process observation confirmed that liming and settlement removed much of the turbidity, resulting in a clear liquor with a very light yellowish tint. Limed liquor was subjected to the struvite protocol described above, targeting an average residence time of 3 hours. MgP solution was mixed in to supply Mg and P at an ammoniacal molar relationship of 1:1:1. NaOH was added to achieve a pH shift that was maintained at between 8.8 and 9.4 in the crystalliser. Subsequent settling of the struvite media for a period of 3 hours resulted in a clear process liquor (Figure 3.3), with the NH_3 concentration reduced to < 10 ppm. Precipitated struvite was recovered.

Clarified struvite liquor at an alkaline pH was neutralised by adding 130 g of sulfuric acid per m^3 of process liquor, shifting the pH to 6.0. Neutralised liquor was subjected to filtration at 2 bar through a 5- μm spun-polypropylene sediment filter. The filter sufficiently removed residual struvite and suspended/colloidal solids, enabling continuous RO at 7 bar using a thin-film composite membrane (rated at 100 GPD) for a period of 2 hours. The RO streams were split 60–40% between permeate and retentate. The NH_3 concentrations in the permeate were measured at < 1 ppm, while the residual NH_3 concentrated in the retentate was measured at < 28 ppm.

While the permeate achieved a discharge standard likely to be acceptable to most receiving bodies, a method may be required for retentate treatment, given its higher concentrations of NH_3 and other ionic fractions. In the landfill leachates analysed, the ammoniacal concentration seemed to be the rate-limiting factor, rather than the concentrations of dissolved metals or other fractions, which appeared to consistently comply with licence conditions. Accepting that the regulatory standard to be achieved is a function of a specific licence/permit application, it was hypothesised that a continuous process might achieve an acceptable discharge standard by routing retentate back to the process head, repeating the liming and struvite processing, and then blending the struvite-clarified liquors at NH_3 concentrations of < 10 ppm with RO permeates with NH_3 concentrations of < 1 ppm to consistently achieve an average NH_3 concentration of < 5 ppm. Given the low levels of metals and other ionic substance concentrations in the original leachate, and the fact that these were already being discharged into the local waterway by the municipal WWT plant, it was considered that this was unlikely to be problematic

from a licensing perspective. Time precluded iterative testing to determine the full impact of this protocol on ammoniacal and other ionic substance concentrations. Additional investigation is required to determine whether or not this concept might be developed and integrated into a system capable of meeting a wide range of regulatory standards.

3.2 Energy and Nutrient Recovery: Ammoniacal Streams with High Solid Loads

Livestock slurries and AD digestates are examples of high-solid-load ammoniacal streams. The solid load potentially creates obstacles to the recovery of NH_3 , and so solid separation is required to facilitate downstream treatment of ammoniacal liquors. Separation and aggregation also facilitate the efficient recovery of renewable energy, which contributes to the mitigation of the environmental impacts that would otherwise result from land spreading. Technologies evaluated in respect of sustainable energy and nutrient recovery include the following (Figure 3.4):

- bulk solid separation by vibrating rundown screen or screw press, followed by bulk solid liming to drive off NH_3 , expedite drying and condition solids for energy recovery; linked with the development of vaporous NH_3 recovery technology;
- suspended/colloidal solid recovery by filtration using ash filter aid; linked with the development of recovery technology to facilitate filter aid valorisation;
- renewable energy recovery via AD;
- NH_3 stripping via liquor liming and partial liquor clarification with lime settlement to facilitate lime recovery and valorisation; linked with the development of vaporous NH_3 recovery technology;
- struvite precipitation to reduce NH_3 levels in partially clarified liquor filtrates and to further remove residual colloidal solids, conditioning liquor to a standard suitable for discharge;
- liquor polishing via filtration and RO (if required) to remove residual dissolved solids.

Livestock slurries were selected as high-solid-load streams for the evaluation of renewable energy and nutrient recovery. The test samples were sourced from the Slurries pilot project of the Sustainable Energy Authority of Ireland (2017a).

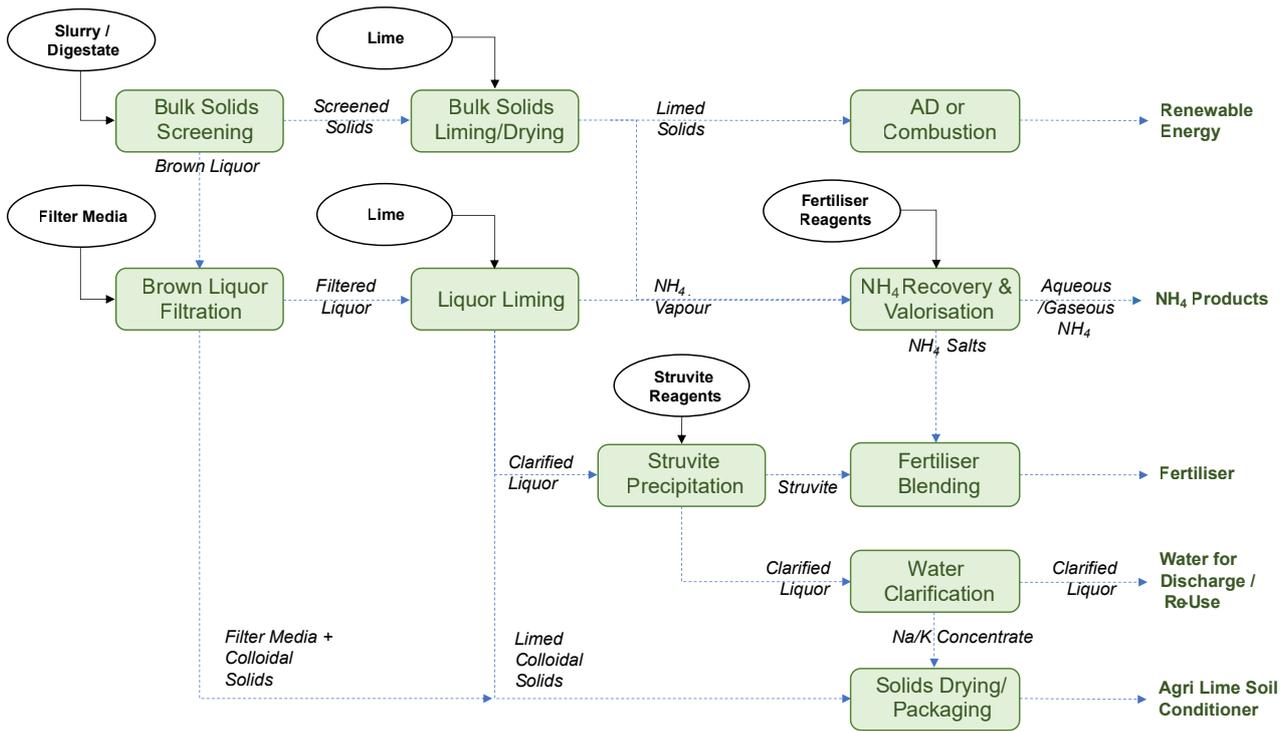


Figure 3.4. Schematic of the process concept to be developed, tested and evaluated.

3.2.1 Bulk solid recovery

Pig and cattle slurry samples were subjected to solid separation and conditioning for purposes of evaluating renewable energy recovery applications.

Teagasc (2022d) reports that pig slurries incorporate on average 4% dry matter (DM), while cattle slurries incorporate on average 6% DM. Vibrating rundown screens (Figure 3.5) were identified as a low-cost technology that leverages gravitational force to separate bulk solid particles from liquors. A vibrating wedge wire rundown screen, configured with 250- μm openings, was used to screen both pig and cattle slurry samples. Visual observations confirmed that pig slurry samples were lighter and much less viscous than cattle slurry samples, with a clearer delineation between the solid particulates and surrounding liquor fractions within the pig samples. Pig slurry bulk solids comprised relatively small granular particles, whereas cattle slurry was heavier and highly viscous, with a bulk solid content comprising longer undigested grass fibres.

Pig slurry bulk solids were readily separated using the vibrating rundown screen. The separated bulk solids accounted for 12% of the input slurry. Table 3.2 shows the partitioning of DM and ammoniacal N achieved by the vibratory screen separator. Separation

indices for the individual components are defined as the percentage of the incoming component that was remediated to the solid fraction, as given by Equation 3.1, where C_i is the concentration of component i in the solids, liquid or raw slurry,



Figure 3.5. Vibrating gravity screen for bulk solids dewatering.

Table 3.2. Pig slurry solid composition (average from analysis of three samples)

Component	Raw slurry	Screen filtrate	Solids	Separation index (%)
DM	3.7% w/w	1.2% w/w	22.03% w/w	71.46
N NH ₃	2600 ppm	1956 ppm	7590 ppm	33.12
N organic	1777 ppm	291 ppm	12,755 ppm	86.13
N oxidised	57 ppm	60 ppm	39 ppm	8.17
N total	4584 ppm	2307 ppm	20,384 ppm	53.36
P	1500 ppm	86 ppm	11,869 ppm	94.95
K	1885 ppm	1200 ppm	6903 ppm	43.95

respectively, and M is the mass of solids, liquid or raw slurry.

$$SI_j = \frac{C_{isolids} \times M_{solids}}{C_{iraw} \times M_{raw}} = 1 - \frac{C_{iliquid} \times M_{liquid}}{C_{iraw} \times M_{raw}} \quad (3.1)$$

The vibratory screen achieved separation indices of 71% and 33% for solids (DM) and ammoniacal N, respectively, yielding a liquid fraction (screen filtrate) with ammoniacal N and DM contents of 1956 mg L⁻¹ and 1.2% w/w, respectively (Table 3.2).

Cattle slurry required a longer residence time on the screen bed for the fractions to separate. It did not separate as effectively as pig slurry, resulting in a higher residual moisture content in the separated solids fraction. Accordingly, it was concluded that an alternative technology would be needed to optimise separation of bulk cattle slurry solids at scale. Supplier presentations and reference sites (e.g. GEA Farming, 2022, and Trident TNZ, 2022) advertise commercially available technologies, ranging from rotary drum screens to roller presses and small-scale screw presses, that are designed to de-water cattle slurry solids. Manufacturers' literature indicates that cake moisture levels of c.20% DM can be achieved, and, given that such technology is market tested, it was determined that the use of this existing technology would be preferable to gravity rundown screens in respect of cattle slurry solid separation.

Separated bulk solids, even when de-watered to a cake, comprise only 10–25% DM. Further moisture removal is advantageous if solids are to be transported for aggregation or productised as a packaged growth medium or solid fuel. Moisture removal via thermally assisted drying technology is very energy intensive. Therefore, liming was investigated in this project as a method to enhance moisture removal and generate a pH shift to drive off NH₃.

Slaked lime (Ca(OH)₂) was added to screened cattle and pig slurry solids (at 17% and 22% DM, respectively) at different concentrations as a proportion of total mass: 4%, 8%, 12% and 24%. Lime was mixed with slurry solids using an open-faced rotary mixer for 5 minutes. Olfactory evidence indicated that, very quickly on introduction of the lime, the release of vaporous NH₃ emissions commenced. Recovery of vaporous NH₃ for valorisation requires an alternative mixer design, an approach that is addressed in the following discussion on limed liquor treatment. Post mixing, limed solid samples were air dried by spreading the moist samples at a depth ranging from 50 to 75 mm on the concrete floor of a covered shed (Figure 3.6). Air drying was assisted by periodically turning and breaking up the samples to increase the surface area exposed to air. Observations from the drying experiments suggested that the addition of



Figure 3.6. Liming of pig slurry (right) and cattle slurry (left) solids.

lime caused moisture to rise and collect on the top of the drying samples, and it was clear from periodic observations that limed samples dried much quicker than untreated control samples.

Over a 3-week period, the limed solids dried to a moisture content ranging from 15% to 25% by mass, while non-limed control samples dried to a moisture content ranging from 47.4% to 53.4%. The limed samples achieved a standard that, with a small amount of supplemental drying, would be suitable for use as a solid fuel (Department of the Environment, Climate and Communications, 2021b). Untreated control samples required supplemental thermally assisted drying. In the limed samples, the relative level of added lime did not directly correlate with drying efficiency (i.e. the samples to which 8% lime were added dried to the same DM range and in a similar time frame as those samples to which 24% lime was added). Given the limited impact derived from adding higher proportions of lime, the samples to which 8% or 12% lime was added (to moist samples on a w/w basis) were selected for testing for energy recovery via AD.

Liming of the slurry solids noticeably reduced the intense odour normally associated with livestock slurries, although odour was not eliminated. It was observed that particle sizes from pig slurry solids (Figure 3.7) were smaller and more granular than cattle slurry solids, and they dried into small chips or

lumps, the largest of which was c.50 mm in diameter. Pig slurry solids also dried quicker than cattle slurry solids, which were more fibrous and dried to a consistency of a packed fibrous mat. The dried cattle slurry solids had to be mechanically chipped into smaller particles of less than 10 mm diameter to facilitate subsequent use as a solid fuel, and the team observed that comminution of the limed cattle slurry solids in a small chipper resulted in significant volumes of dust, which would require abatement/management in a commercial application. The liming then drying of slurry solids may have caused the degradation/disassociation of lignin and other structural components in the biomass (not unlike the degradation that occurs in the kraft pulping process or other alkaline biomass pre-treatment processes), resulting in particle size reductions, especially when the biomass is subjected to mechanical stresses, such as chipping or agitation. This physical degradation and the resulting small, accessible particles may, in part, explain the increase in biogas production when limed slurry solids were anaerobically digested.

Test results indicate that liming slurry solids may be a technically viable method of conditioning solids for valorisation, driving off NH_3 and facilitating low-energy air drying. Test observations suggest that methods would need to be employed to manage NH_3 vapours, odour and dust to comply with planning and licensing requirements, and health and safety regulations, should this process be scaled up.



Figure 3.7. Chipped cattle slurry solids (left) and non-chipped pig slurry solids (right).

3.2.2 Suspended/colloidal solid recovery

For high-solid-load ammoniacal streams, it may be necessary to not only remove the bulk solids, but also reduce colloidal solid levels to avoid inhibiting downstream NH_3 removal. Two different filtration methods were evaluated to determine the viability of colloidal solid separation from the liquor fraction of screened pig slurries. The first method involved the use of a packed sand bed, where a bed of small pebbles was covered by a layer of conventional builders' sand (particle sizes ranging from 0.425 to 2.0 mm) and topped with a very thin layer of fine crystalline struvite (particle sizes ranging from 40 to 100 μm) (Ronteltap *et al.*, 2010). This unit relied on gravitational pressure to drive the media through the bed and was successful in partially filtering screened pig slurry liquors (Figure 3.8), although viscous cattle



Figure 3.8. Pig slurry filtrate from sand bed filtration.

slurries remained suspended on top of the sand bed and liquors did not readily pass through the filter bed.

A high-pressure filtration mechanism was also evaluated for removal of suspended and colloidal solids. In bench-scale tests, compressed air at c.2–4 bar was used to force liquors through a filter bed comprising different test media (Figure 3.9). Filter aids ranged from biomass, such as sawdust, chopped straw and coffee grounds, to mineralised particles, including biochar and ash residues. Biomass particles and biochar did not retain sufficient structural integrity and blocked the filtration unit when pressure was applied. The mineralised particles (e.g. ash derived from the combustion of AD digestates, poultry litter or meat and bonemeal), however, retained sufficient



Figure 3.9. Bench-scale pressure filtration unit.

structure under pressure to filter colloidal solids from screened slurry liquors for a period of time, until the colloidal solids built up on the filter bed and resulted in a gel-like coagulation that eventually blocked the filtration unit (Figure 3.10). These tests identified suitable candidates for filter aids but highlighted that periodic (or continuous) removal of coagulated solids will be required to avoid blockage.

To evaluate the use of continuous high-pressure filtration at scale, the project deployed a fabricated filtration unit, which comprised a multi-stage high-pressure pump that forced screened slurry liquors through a vertical 5- μm sintered metal filter cylinder housed in a heavy-gauge steel support column periodically perforated to allow filtrate exit (Figure 3.11).

The filter aid, comprising conditioned ash derived from the combustion of poultry litter, was pre-mixed with the process media and, as it progressed through the process, an internal auger removed built-up excess filter aid, maintaining a specified depth of approximately 15 mm on the filter unit wall. Pressure levels were managed via the combination of a pneumatic gate valve, which periodically opened to allow exit of the extracted filter aid solids, together with flow control via the high-pressure pump. In respect of the poultry litter ash filter aid, both fly ash (smaller particulates ranging between 20 and 70 μm (Coutand *et al.*, 2007)) and bottom ash (slightly larger



Figure 3.10. Colloidal solids on filter bed.



Figure 3.11. High-pressure pilot filtration rig.

particulates ranging between 400 and 700 μm) were tested.

Fine silt-like particulates were removed from both fractions by washing to avoid formation of a slick, impermeable silt layer that would significantly inhibit the filtration capacity of the unit. This washing step created an effluent. It may be possible to treat the residual wash water fraction by filtering it back through the spent filter media after it has been used to remove colloidal solids from the brown liquor. Alternatively, a different WWT technology will have to be identified. Time and budget constraints precluded work on a WWT process for this fraction, and further work is thus required to address this.

The pilot-scale filtration unit was used to test the continuous filtration of screened cattle slurry samples. Water mixed with the coarse bottom ash filter media from combusted poultry litter was circulated to first establish the filter bed. Thereafter, screened cattle slurry liquors (diluted 50% by volume with water) were routed through the unit to test separation of suspended/colloidal solids. Dilution of the cattle slurry liquors created a more free-flowing process medium that, from observation, was similar in solid concentration to filtered pig slurry. Fine filter aid medium comprising conditioned poultry litter fly ash was subsequently added to the brown liquor process

medium at a ratio of approximately 1 part filter aid medium to 50 parts process medium by mass until a visible suspension in the recirculated liquor was achieved. Filtration results achieved using solely the coarse filter aid were compared with results achieved with the addition of the fine filter aid (Figure 3.12). In each pair of bottles shown in Figure 3.12, the cloudy liquor is the unfiltered process medium (including the added filter aid), while the clearer liquor is the corresponding filtrate. The pump speed, recirculation valve aperture and timing of the exit gate were adapted to maintain pressure within a range of 12–15 bar in the filtration unit. Initial observations suggested that the exit medium extracted from the top exit gate was in a liquid form rather than in a cake form, as expected. As a result, the operating pressure initially dropped quite dramatically when the exit gate was opened. To address this, the project team introduced a secondary valve in the exit pipe in an attempt to maintain a more consistent, stable pressure. Once these design and operational changes were introduced, it was observed that a stable pressure resulted in clarification of the slurry process medium to a light-yellow colour, as was expected based on the results of the bench-scale filtration tests. The clarity was similar to that achieved in the initial sand bed filtration tests.



Figure 3.12. Filtration results using different blends of filter aid. The two bottles on the left show the results achieved using only a coarse filter aid and those on the right show the results achieved with the addition of the fine filter aid. The first and third bottles from the left contain the unfiltered process medium (including filter aid) and the second and fourth contain the filtrate.

The continuous unit achieved a degree of filtration suitable for downstream liming and struvite processing, but the flow rate was insufficient for a commercial application (approximately 1.5 L min^{-1} or 90 L h^{-1}). To improve the flow rate, operating pressures were progressively increased to >20 bar. While flow rate increased, mechanical failures resulted in contamination of the filtrate. It was concluded that, while high-pressure filter-aided filtration can be used to clarify screened slurry liquors to a standard sufficient to avoid the inhibition of downstream processes, design modifications are required to increase the throughput and reliability to levels required for commercialisation.

3.2.3 Energy recovery via anaerobic digestion

AD is described as the microbial breakdown of organic material in the absence of oxygen to yield a methane-rich biogas and residual digestate (Sustainable Energy Authority of Ireland, 2020). Biogas is mainly a mixture of methane and CO_2 with some other trace gases. The combustible methane content typically ranges between 55% and 60%, which enables combustion as a fuel in boilers or engines to generate heat and/or electrical power. If biogas is further conditioned to remove CO_2 and other contaminants, the upgraded biomethane can replace natural gas and be distributed via the gas grid or used as a gaseous transport fuel. Digestate can be used for land spreading in an unprocessed form (although this generates environmental issues similar to those associated with the land spreading of slurries). However, if it is dried and productised to a standard that complies with specifications enumerated in the new EU FPR (European Union, 2019), it could potentially be marketed across the EU as a CE-marked fertiliser or soil conditioner.

AD can be an effective way of managing farm manure, but installations are capital intensive, requiring the co-digestion of manures with other carbon-rich feedstock (e.g. food waste, digestible green waste, energy crops) to generate methane outputs ranging between 200 and 300 m^3 per tonne DM of feedstock, and economies of scale, required to remunerate the capital investment costs (Sustainable Energy Authority of Ireland, 2020). One advantage of AD is that it typically takes place in a moist environment, avoiding the energy input otherwise required to dry feedstock. Different technology configurations

are available for application of AD, and methane production is optimised if operating parameters, such as microbial culture, the carbon-to-nitrogen (C:N) ratio, pH, temperature, moisture content, multi-staging and agitation, are maintained at specific levels tailored to the individual feedstock/technology design (Sustainable Energy Authority of Ireland, 2020).

While AD is primarily considered an energy technology, it can also have a beneficial impact on nutrient release. The process effluents (solid and liquid) contain inorganic nutrients (Campos *et al.*, 2019), including $\text{NH}_3/\text{NH}_4^+$ and soluble P, which are released during the breakdown of organic compounds. Released nutrients are soluble and tend to form inorganic compounds or adsorb onto solid surfaces in the digestate. Mehta and Batstone (2013) report that, during manure AD, nutrients, including P, Ca and Mg, are released from organic matter but remain bound (or precipitate) as inorganic compounds, which are re-dissolved at low pH (Güngör and Karthikeyan, 2008).

Concentrations of soluble P in most municipally digested waste range between 50 and 500 mg L^{-1} , with about five times more N than soluble P (Le Corre *et al.*, 2009). To reduce solid-handling costs, the digested solids are typically de-watered, generating a soluble, nutrient-rich (predominantly N and K) side stream, which then becomes a feedstock for nutrient recovery. The remaining particulate-bound nutrients and residual organics are recovered as biosolids, which can be used for agricultural purposes provided that mitigation requirements in respect of residual odour, pathogens and heavy metals are met. The economic value of bulk biosolids has historically been difficult to estimate. Productising the biosolids to meet the requirements of the new FPR (European Union, 2019) may offer a route to valorisation. Processing that selectively enhances solubilisation of nutrients can be used to channel a larger portion of nutrients into the valuable, nutrient-rich product stream, instead of into lower-value biosolids. This processing may include the addition of complexing agents, such as ethylenediaminetetraacetic acid, or operation at depressed pH or otherwise modifying the operating conditions to reduce the quantity of nutrients being sequestered with the biosolids/sludge.

In Ireland, AD is a regulated activity, requiring (in addition to planning permission) regulatory

authorisation pursuant to Waste Management (Facility Permit and Registration) Regulations (S.I. No. 821 of 2007 as amended in 2008). The capacity of the facility determines the relevant authorisation route. For small facilities (with < 10,000 tonnes intake per annum), the permit is obtained from the local authority. For larger facilities, the licence is obtained from the EPA. A permit/licence may include limits on, for example, activities, feedstock types, environmental emissions and digestate quality.

If an AD facility processes inputs classified as ABPs, there is a requirement under the EU Animal By-products Regulation (Regulation (EC) No. 1069/2009, as implemented in Ireland by S.I. No. 187 of 2014) to ensure that risks of pathogen and/or disease transmission via digestate are mitigated. ABPs are classified according to their pathogenic risk, with the highest risk being category 1 and the lowest risk being category 3. Different regulatory authorisations apply depending on the type of facility and its inputs. AD digestates derived from high-risk, category 1 feedstocks may not be used for land spreading and must be incinerated. Digestates derived from category 2 feedstock (e.g. livestock manures) and category 3 (e.g. catering waste) may be used for land spreading provided that the digestate has been pasteurised at a minimum temperature of 70°C for at least 1 hour and the particle diameter is no

larger than 12 mm. Compliance with ABP regulations is subject to certification by DAFM.

While livestock slurries are considered a potential feedstock for AD, mono-digestion can be problematic, in part because of the presence of excessive levels of NH_3 and suboptimal C:N ratios. Chen *et al.* (2016) report that the presence of NH_3 at levels exceeding 2 g L^{-1} strongly inhibits methanogenesis and corresponding biogas production. The project investigated whether or not the reduction in NH_3 via the addition of lime to slurry solids would positively influence the C:N ratio and sufficiently reduce NH_3 levels to increase biomethane production relative to the digestion of untreated slurry solids. Samples of the dried pig slurry solids, both untreated and with lime added at, on average, a 10% level (by moist mass), and a sample of macerated grass clippings (for comparison and for testing as a possible co-digestion feedstock) were tested for biochemical methane potential using different feedstock combinations. Biochemical methane potential tests of (i) grass, (ii) untreated slurry solids, (iii) lime slurry solids, (iv) grass and untreated slurry solids and (v) grass and lime slurry solids were undertaken in 500-mL batch AD reactors (Figure 3.13; Table 3.3). The inoculum used for the test was a granular anaerobic sludge (10.65% on a volatile solids (VSs) basis) collected from a dairy WWT facility in County Galway. The

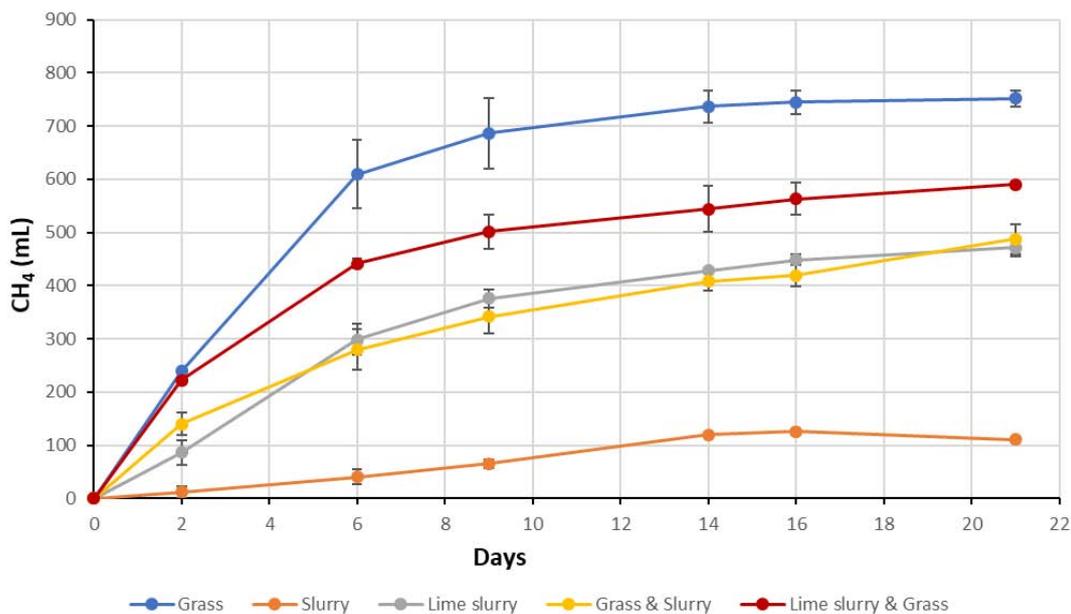


Figure 3.13. Profile of the cumulative methane volume obtained after 21 days of incubation of grass, slurry, lime slurry, grass and slurry, and grass and lime slurry at 37°C.

Table 3.3. Biomethane potential tests: feedstock characteristics

Sample ID	TS content (%)	VS content (%)	VS content/TS content (%)
Grass	23.20	20.16	86.88
Pig slurry solids	31.60	25.27	80.21
Lime pig slurry	23.20	16.75	51.89

working volume for each reactor was 300 mL and the feedstock loading concentration was 10 g VSL⁻¹. A ratio of 1:2 (feedstock to inoculum on a VS basis) was applied. Sodium bicarbonate buffer (300 mL at 10 g L⁻¹) was added to each reactor containing 6 g VS of granular sludge, followed by loading with 3 g VS of feedstock. All test reactors were sealed, flushed with N₂/CO₂ and incubated at 37°C under agitating conditions. Control reactors contained only granular sludge and the anaerobic buffer. A 1-L gas bag was attached to each reactor for biogas collection, which was analysed over the incubation period for the production of methane and its composition.

Experiments were carried out in triplicate. The results highlight that the biomethane production was highest in the grass (only) sample (at 251 mL g⁻¹ VS added). Conditioning the slurry solids with lime, however, appears to have had a significant beneficial impact on biomethane production, enabling the generation of three times more biomethane than non-conditioned slurry solids. Moreover, the co-digestion of grass plus limed slurry solids generated c.25% more biomethane

than that generated from the co-digestion of grass plus untreated slurry solids (Figure 3.14). Furthermore, test results indicate that, in addition to the beneficial impact on biomethane generation, conditioning slurry solids with lime had no significant inhibitory effects on co-digestion with other carbon-rich feedstocks.

Given the maturity of AD technology and the availability of defined routes to market, and provided that operations comply with relevant licensing authorisations and demonstrate required levels of GHG savings, AD can be considered a potentially viable route for renewable energy recovery from limed slurry solids.

3.2.4 Ammonia recovery from slurry filtrate liquors

The Nitrates Directive governs the level of N that can be applied to agricultural land. Given the level of ammoniacal N relative to total N in livestock slurries, NH₃ load is the limiting factor in determining the amount of slurry liquor that can be discharged to a land bank. To evaluate prospects for NH₃ removal from screened and filtered slurry liquors, pig slurry filtrates were subjected to a multi-stage protocol. First, NH₃ stripping was performed via the application of lime to facilitate CO₂ lock-up and generate a pH that resulted in NH₃ elution. Thereafter, a struvite process was applied to polish the liquor to a standard sufficient for localised discharge, in compliance with Nitrates Directive regulations.

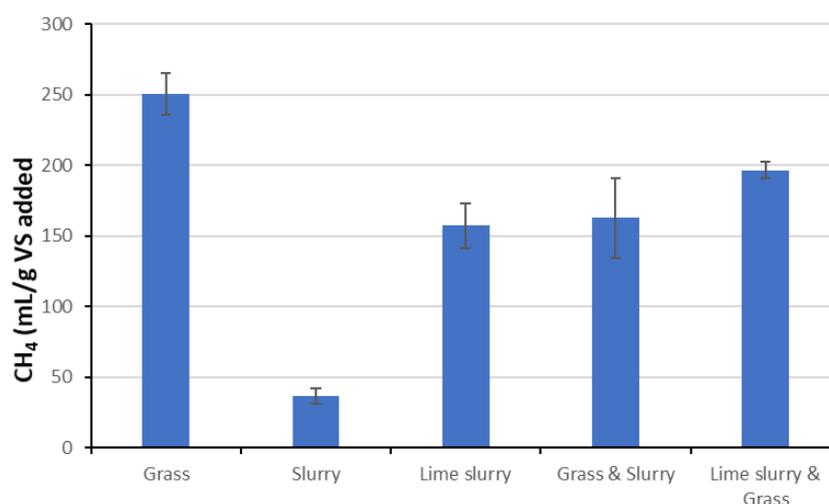


Figure 3.14. Methane yields (g⁻¹ VS added) obtained from grass, slurry, lime slurry, grass and slurry, and grass and lime slurry after 21 days of incubation at 37°C.

NH₃ stripping via liquor liming

The US Environmental Protection Agency (2000) highlights that it is easier and less expensive to remove N from aqueous streams in the form of NH₃ than to convert it to NO₃⁻-N via the nitrification process before removing it as N₂ gas via denitrification. In aqueous solutions, NH₃ (a weak base) reacts with water (a weak acid) to form ammonium hydroxide. To strip NH₃ from an aqueous solution, alkaline reagents can be added to shift the pH to a range of 8–10, which enables the elution of ammonium hydroxide as NH₃ gas.

NH₃ stripping via the application of lime to pig slurry filtrate was investigated as a means to:

- remove the interference of endogenous CO₂ and its carbonate analogues in NH₃ chemistry, facilitating NH₃ removal via struvite precipitation by locking up CO₂ as calcium carbonate;
- facilitate the removal of colloidal solids in pig slurry filtrates by acting as a surface to which the negatively charged colloidal solids can adhere, removing colloidal solids as the lime settles;
- achieve a pH shift that leads to the evolution of appreciable amounts of NH₃ to yield an ammoniacal solution with a concentration that reduces subsequent demand for struvite reagent and reactor capacity to a level that renders stripping technically and economically viable for NH₃ recovery.

The availability of an agricultural market outlet potentially offers routes for the recovery of spent lime. An initial scoping exercise was undertaken to observe how different varieties of lime performed as a liming agent. Using the filtrates from the sand bed filtration mixed with lime in a rotary mixer, three different varieties of lime were tested to determine their effectiveness in driving off NH₃. These were (i) conventional quarried lime (predominantly CaCO₃), (ii) slaked lime (Ca(OH)₂) and (iii) quicklime (CaO). CaCO₃ was ineffective in driving off NH₃, and CaO, while initially quite reactive, tended to agglomerate and did not maintain a consistent dispersion throughout the process media. Ca(OH)₂ exhibited a particle size that allowed maintenance of even dispersion throughout the process media over an extended period, optimising surface area exposure and resulting in reactivity sufficient to drive off dissolved NH₃. It was concluded that Ca(OH)₂ was the best candidate for testing at scale.

In subsequent pilot-scale tests, hydrated lime (water treatment grade) was added to the liquid fraction from the sand bed separation process in varying amounts and stirred for various times in an open rotary batch mixer to facilitate CO₂ lock-up, partial removal of the colloids and NH₃ elution. Aliquots of the resulting working liquors were removed over time and allowed to settle. The partitioned liquid fraction was analysed for NH₃. Liquid samples were analysed in triplicate for DM and ash by weight difference following drying and combustion at 100°C and 550°C for 12 hours and 1 hour, respectively. Crystallised solid samples were dried in an oven prior to X-ray diffraction analysis. After the liming step, the resulting suspension was placed in a settling drum for 24 hours before decanting the liquid fraction. The solid fraction was readily recoverable via gravity settlement. The liquor fractions from several liming runs were aggregated and mixed for at least 7 hours to supply the process liquors for struvite precipitation.

Three different Ca(OH)₂ concentrations (4.5%, 5.5% and 6.5% w/w) were applied to the pig slurry liquor (Figure 3.15), and in each case significant NH₃ reduction was achieved within relatively short time frames. The shape of the curves is exponential, indicating that the overall process of NH₃ elution is first order in NH₃ concentration. Plotting the NH₃ profiles as the logarithm of NH₃ (mmol) indicated a good fit with a linear function of the form:

$$\ln(NH_3)_t = \ln(NH_3)_0 + k_{[Ca(OH)_2]} t \quad (3.2)$$

The first-order rate constant is dependent on the Ca(OH)₂ concentration applied with dependence expressed in Equation 3.3, where the rate constant unit is hour⁻¹, NH₃ is as above in Equation 3.2 and the unit of Ca(OH)₂ concentration is % w/w (Figure 3.15):

$$k_{Ca(OH)_2} = -0.04835[Ca(OH)_2] + 0.02023 \quad (3.3)$$

As the process facilitated NH₃ concentrations as low as 200 mg L⁻¹ within 10-hour HRT, a comparative run with 0.3% polyacrylate polymer added to the filtrate with no lime was performed for an extended period. After 40 hours of mixing, NH₃ levels in the absence of lime remained above 1000 mg L⁻¹, demonstrating that the rate of NH₃ elution achieved was not a consequence of the mixing regime applied in the batch mixer. In addition, to compare the removal rates achieved with hydrated lime (solid suspension) with a more soluble agent for pH adjustment, further runs

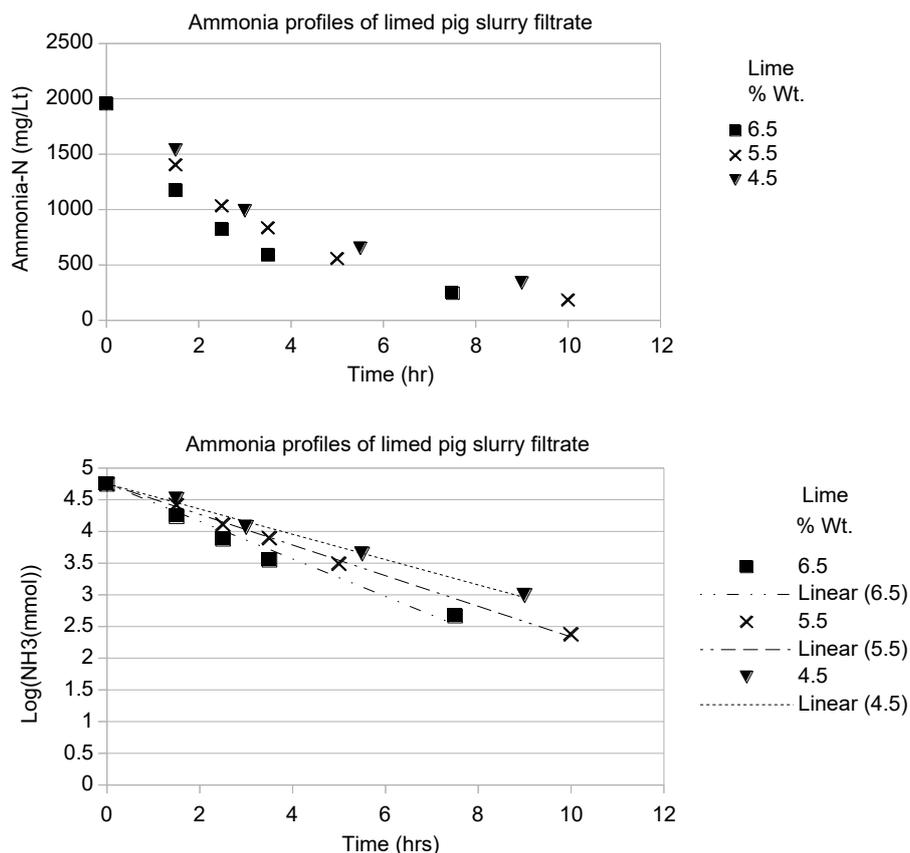


Figure 3.15. NH₃ profiles of limed pig slurry filtrate.

were carried out with 2% and 1% NaOH as the source of alkalinity. A similar trend was observed for the NaOH concentration effect on NH₃ removal rates during mixing, with the rate constant dependence on NaOH concentration given in Equation 3.4:

$$k_{NaOH} = -0.0835[NaOH] - 0.0748 \quad (3.4)$$

The effect of added base on NH₃ removal demonstrates that 2% NaOH is equivalent to approximately 5.5% Ca(OH)₂. In both cases, up to 75% NH₃ removal was achieved within a 5- to 7-hour time frame, with half-life (*t*_{1/2}) values of 2.869 and

2.866 hours for the 5.5% Ca(OH)₂ and 2% NaOH additions, respectively. Temporal elution of NH₃ and pH were measured using a ProLab2500 digital meter and an SI Analytics NH1100 NH₃ ion-selective electrode. The impact of treatments on NH₃ concentration is outlined in Table 3.4.

Given the impact of surface area exposure on the reaction, reaction times may potentially be shortened if the lime particle size can be greatly reduced (to sub-µm or nanoscale); however, this hypothesis would have to be tested to determine its validity.

Table 3.4. Concentrations of ammoniacal N (NH₃-N) in the resulting solids and liquors post treatment

	Raw slurry	Vibratory screen		Filtration and Ca(OH) ₂ (5.5% w/w) ^b	Polyacrylate polymer (0.3% w/w) ^c	Filtration and NaOH (2% w/w) ^b
		Separation index	Solids ^a			
Overall split			12%	88%		
NH ₃ -N	2600 mg kg ⁻¹	35%	300 mg kg ⁻¹	1956 mg kg ⁻¹	291 mg kg ⁻¹	1030 mg kg ⁻¹

^aBy calculation.

^bAfter 8 hours of mixing.

^cAfter 40 hours of mixing.

To evaluate the character of the NH_3 vapour being driven off by liming, a preliminary analysis of CO_2 , NH_3 , hydrogen sulfide (H_2S) and moisture content was performed using a handheld BIOGAS 3000 meter on samples of NH_3 vapour extracted from the (unsealed) rotary mixer. Access to analytical equipment and third-party analytical services was severely curtailed due to COVID-19 restrictions in effect during the test period; accordingly, vapour analysis results provide only a very broad indication of the ammoniacal vapour composition. NH_3 concentration was measured at levels approaching 1000 ppm; however, as this level represents the upper limit of the meter's measurement capability, the actual concentration in the air-diluted vapour could only be estimated to range from c.1000 to 1200 ppm. CO_2 was measured at 383 ppm, H_2S was detected at only trace (unquantifiable) levels and moisture was measured at 12.5 g m^{-3} , all of which are levels that might be expected in ambient air. While the preliminary analysis could not be relied on as an accurate measurement of the absolute fractional values or of the specific vapour character, they do indicate that the addition of lime did not drive off moisture, CO_2 or sulfur in highly concentrated vaporous forms, which suggests that the quality of the NH_3 vapour being driven off by liming may be of a standard that would be suitable for recovery via one of the routes discussed above. Further investigations are, however, required to accurately establish the character of the vapour.

To test the continuous application of the liming process, a rotary reactor was designed and fabricated to continuously process liquor on a scale of c.500 L. During commissioning, olfactory observations indicated that vaporous NH_3 was not being driven off at the same intensity as in the smaller-scale reactor. It was concluded that the internal reactor sub-structure required re-designing to improve agitation and surface exposure of the limed process media to the surrounding gaseous environment. In addition, it was concluded that the point of extraction was not sufficiently sealed from air ingress, resulting in the dilution of the process vapour within the vessel. It was concluded that the re-design of the rotary reactor should incorporate wider blade designs, staged blade locations around the internal diameter of the reactor and sealed ingress/extraction ports. It was hypothesised that incorporation of a vacuum extraction might reduce residence time and generate a more

highly concentrated NH_3 vapour, making recovery easier. Additional testing of the proposed design modifications is required to validate performance of the continuous liming procedure and optimise NH_3 recovery and to validate a protocol for productising and valorising the vaporous NH_3 output.

Ammonia recovery via struvite precipitation

Struvite is one of the only NH_3 salts that is insoluble in water, which makes it useful for removing NH_3 from wastewaters. NH_3 is removed from aqueous solutions by adding dissolved P and dissolved Mg in a 1:1 molar ratio with NH_3 , followed by an increase in pH to precipitate the salt. Current synthesis protocols commonly utilise magnesium chloride as the Mg source because it is readily soluble in water. This, however, results in an increase in the chloride concentration of the effluent liquor, which can potentially be problematic if discharged to agricultural land or sensitive freshwater bodies. To avoid excessive concentrations of chloride in the dischargeable process liquor, an MgP protocol was developed, as described above, for using MgO as the Mg source. While this process is slightly more costly than established protocols, the supply of Mg and P is determined based on the molar relationships with NH_3 . A reduction in slurry NH_3 levels via liming reduces the requirement for struvite reagents.

The pilot-scale struvite unit was adapted for scaled-up testing on treated pig slurry (Figure 3.16) using liquor fractions sourced from several liming runs that were mixed for at least 7 hours and aggregated. A two-stage struvite precipitation process was used with the limed liquor in a continuously stirred tank reactor seeded with 2% struvite crystals. MgO and H_3PO_4 were added at time zero in the form of a 1.0-M acidified MgP solution to yield a stoichiometric ratio of Mg to P to NH_3 of 1:1:1. NaOH was added to achieve an alkaline pH within a range of 8.8–9.4 to stimulate struvite formation. While test protocols used NaOH as the pH-shifting agent, if the struvite process is used to reduce NH_3 concentrations in liquors ultimately discharged to agricultural land, it may be appropriate to utilise KOH as the pH-shifting agent, which, although more costly, may result in residual K being dissolved in the liquor effluent that will contribute to crop nutrient requirements on discharge.



Figure 3.16. Pilot-scale struvite unit.

The crystalliser was operated in batch mode for 24 hours and liquor NH_3 was monitored over time. Solids were collected at the end of the run, dried in an oven at 35°C and analysed by X-ray diffraction. Temporal elution of NH_3 and pH were measured using a Prolab2500 digital meter and an SI Analytics NH1100 NH_3 ion-selective electrode, calibrated in accordance with the manufacturer's instructions using ammonium chloride standards. Working liquor samples were

diluted 1 in 100 with deionised water prior to being analysed in triplicate.

Struvite crystallisation in liquor filtrates in which NH_3 was first reduced with the addition of $\text{Ca}(\text{OH})_2$ was compared with liquor filtrates in which NH_3 was first reduced with the addition of NaOH . It was observed that, while NH_3 removal during the mixing stage was similar for both treatment protocols, the effects of the different agents on NH_3 removal were significantly different post mixing (Figure 3.17). In each case, stoichiometrically equivalent levels of Mg and PO_4 were added to the liquors derived from the mixing step. Comparison of the bases is best revealed in terms of the percentage of the added phosphate that resulted in struvite and consequently NH_3 removal. As NaOH is a soluble base, a large portion remains in solution and associates with the phosphate, most likely in the form of sodium di- or monohydrogen phosphate salts. As these salts are soluble, they remain in solution and do not participate in crystallisation. This is reflected in the high DM and ash contents observed in the (post-struvite) NaOH liquors, measured at 3.41% and 2.91%

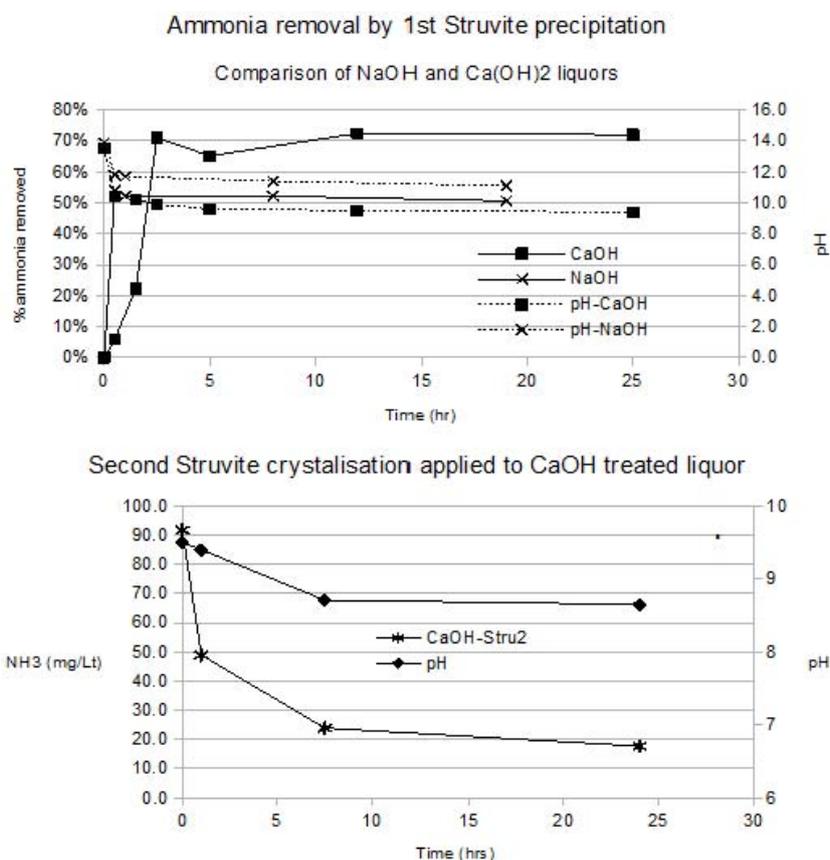


Figure 3.17. Ammonia removal from limed slurry liquor via struvite precipitation.

by weight, respectively. The post-struvite ash content was higher than that measured in the raw slurry.

By contrast, as only a small portion of the $\text{Ca}(\text{OH})_2$ remains solubilised following settlement and removal of struvite, phosphate usage during crystallisation in the lime system is far more efficient, with over 70% of the added phosphate being utilised for struvite precipitation in the hydrated lime system, compared with only 50% in the NaOH system. The phosphate that is not used in precipitating struvite is not lost in the $\text{Ca}(\text{OH})_2$ liquors, but rather the remainder is co-precipitated as insoluble calcium phosphate, which is itself a valuable fertiliser that is recovered in gravity settling. Using struvite precipitation, the NaOH and $\text{Ca}(\text{OH})_2$ mixing treatments resulted in liquors in which NH_3 concentrations could be reduced to 136 and 92 mg L^{-1} , respectively. By comparison, the effluent that resulted from the first struvite precipitation of the limed liquor had only 0.45% ash (most likely dissolved calcium carbonate/hydroxide) and only 0.33% organic solids (Table 3.5).

As the slurry is exposed to several crystalline solids that settle out of the slurry, successive opportunities are provided to the organic solids that remain in the slurry post the initial vibratory screening process to attach to these adsorbing surfaces and drop out with them. Consequently, the organic solid content of the liquors decreases with each subsequent operation to a final value of 0.21%, following the second struvite precipitation. A similar trend was observed in terms of oxidised N and P, demonstrating that the benefit of using $\text{Ca}(\text{OH})_2$ in combination with struvite precipitation is not only that ammoniacal N can be remediated but that other problem components can

also be dealt with. This beneficial effect is summarised in Table 3.5.

Post struvite precipitation, the resulting $\text{Ca}(\text{OH})_2$ liquor was comparatively clean of suspended solids. A second struvite precipitation step was applied to this liquor, that is, further stoichiometric P and Mg were added using the MgP protocol, and a final NH_3 concentration of 17.66 mg L^{-1} was achieved after 24 hours' precipitation. The NH_3 profile is shown in Figure 3.17 (bottom panel) along with the corresponding pH profile. The solid content is shown in Figure 3.18, and Table 3.5 shows the corresponding solid fractions and ammoniacal concentrations at each stage of the progressive treatment.

The progressive application of filtration, stripping and struvite precipitation reduced ammoniacal N from an initial concentration of 2600 ppm to <20 ppm (total N reduction from 4584 to 43.2 ppm). This combined protocol also significantly reduced solid loads, resulting in a clarified liquor that can be discharged

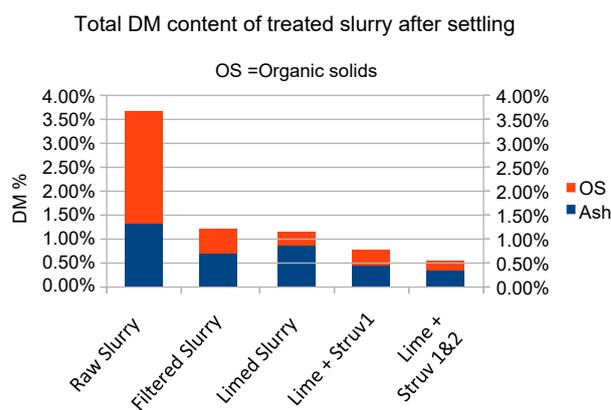


Figure 3.18. Solids content of slurry liquors post treatment.

Table 3.5. Concentration of ammoniacal N ($\text{NH}_3\text{-N}$) in the resulting solids and liquors post treatment

	Raw slurry	Filtration and lime ($\text{Ca}(\text{OH})_2$) (5.5% w/w)	Filtration and NaOH (2% w/w)	Process liquors (post-struvite precipitation)		
				Filtration, lime (5.5%) and struvite 1	Filtration, lime (5.5%) and struvite 1 and 2	Filtration, NaOH (2%) and struvite 1
DM (% w/w)	3.7	1.13	3.85	0.78	0.54	3.41
Ash (% w/w)	1.31	0.85	3.35	0.45	0.33	2.91
Organic solids (% w/w)	2.35	0.28	0.50	0.33	0.21	0.50
$\text{NH}_3\text{-N}$ (mg kg^{-1})	2600	291 ^a	285 ^a	92.00 ^b	17.66 ^b	136.00 ^b

^aAfter 8 hours of mixing.

^bAfter 24 hours of mixing.

Struvite 1, first struvite precipitation; struvite 2, second struvite precipitation.



Figure 3.19. Slurry liquor samples progressively clarified by screening followed by filtration, liming and struvite precipitation.

via pump (Figure 3.19). Discharge via pump avoids the diesel-fuelled tractor movements required for the land spreading of raw slurry. It is convenient, less costly and readily controlled, potentially allowing for application to land in small increments based on crop demand and weather conditions. Precipitated struvite is readily recovered from process media via gravity settlement. Air drying (with struvite spread at a depth of c.50 mm under cover over 2 days) dries struvite solids to a standard that can be transported.

Effective removal of NH_3 reduces the land bank area required for slurry management. Discharge of 1000 m^3 of struvite-treated liquor (total N concentration of 43.2 ppm) requires only 0.25 ha to comply with the 170 kg ha^{-1} regulatory limit, whereas a land bank of c.27 ha would be required to accept 1000 m^3 of untreated slurry. Once NH_3 concentrations are reduced, K load becomes the rate-limiting factor governing the application of slurry liquors on land. K concentrations depend on the type of slurry and the alkaline additive used to adjust the pH in struvite precipitation (KOH versus NaOH). Teagasc guidelines recommend an annual K application rate of 90 kg ha^{-1} (Teagasc, 2022e). The K concentration in the residual struvite liquor (using NaOH as the pH shift agent) was estimated at 1250 ppm (1.25 kg K m^{-3}). This would



Figure 3.20. Grass sward 2 weeks after application of treated liquor effluent.

suggest a process liquor application rate of c. $72 \text{ m}^3 \text{ ha}^{-1}$ (equating to a land bank requirement of 13.9 ha for discharge of 1000 m^3) before recommended K application levels were reached; or a hydraulic loading rate of 7.2 L m^{-2} , which, given Ireland's average annual rainfall of c.1000 mm, is unlikely to be inhibitory.

Anecdotal evidence from farmers indicates that concentrated land spreading of slurry sometimes results in a short-term inhibition of grass growth (described as “burning” of grassland swards and occurring over a 2- to 3-week period immediately after slurry application). Here, 10-L samples of neutralised limed liquor effluents (pH adjusted to 7 with sulfuric acid) were applied repeatedly over 6 weeks to 1 m^2 of grassland. A control area surrounding the test sward was maintained with no application of liquor or alternative fertiliser. No “burning” or inhibition of grass growth was observed on application of the treated liquor effluent (Figure 3.20). After 2 weeks, a slight increase in grass growth rate within the treated area was noticeable. Over 6 weeks, accelerated grass growth continued. This exercise was a preliminary scoping exercise to evaluate whether or not short-term inhibition of grass growth arose on application of treated effluents. A comprehensive exercise, with multiple samples and multiple growth cycles, will be required to accurately quantify the impact of treated effluents on grass productivity, determine causality and evaluate the impact on soil characteristics.

4 Techno-economic Review

The project tested energy and nutrient conversion technologies that (once optimised and tested) have the potential to be integrated into a flexible multi-stage processing protocol designed to reduce NH_3 concentrations in effluents to achieve discharge targets. Reductions in N discharge levels would reasonably be expected to contribute to improving water and air quality and reducing GHG emissions (although quantifying the net contribution to GHG emissions would require full life cycle analysis and this was beyond the scope of this project).

Achieving measurable contributions to environmental sustainability requires technology commercialisation and roll-out underpinned by economically sustainable business models that together ensure compliance with regulatory requirements. Depending on the specific application, business models vary. Structured conversations with stakeholders, considering different business models, identified a range of commercial and regulatory factors that influence economic sustainability, some of which are discussed in this section.

4.1 Landfill Leachates

In this project, treatment of both “heavy” and “light” leachate streams showed that NH_3 concentrations in effluents could be significantly reduced, removing NH_3 from discharge streams and reducing the risk of non-compliant discharges. Economic sustainability, determined by comparing the net processing costs of the new treatment protocol with the costs of the current treatment protocol, varies because different business models are applicable to different sites.

4.1.1 Heavy leachate streams

NH_3 concentrations could be reduced from c.1500 ppm to c.100 ppm via application of the struvite process, without any significant interference from the organic colloidal solids. From an environmental perspective and based on leachate volumes produced at the County Limerick test landfill site (Chapter 3), processing 260,000 m³ of such leachate per year would remove 364,000 kg of NH_3 from the discharge stream. It would reduce the risk of periodic non-compliant

discharges driven by NH_3 levels from the on-site WWT plant. With respect to GHG emissions, life cycle analysis is required as part of future work to compare the energy intensity of the struvite protocol with the energy intensity of decreasing NH_3 concentrations through the nitrification–denitrification process currently used in the WWT plant. Moreover, it will be important to assess the impact on GHG emissions of replacing carbon-intensive conventional fertilisers with struvite synthesised using recovered NH_3 .

From an economic perspective, on a scale of 260,00 m³ per annum, it was estimated that the net cost for processing (assuming recovery of value from struvite sales) ranged between €5.00 and €7.50 m⁻³, which is unlikely to remunerate the incremental savings in power and WWT processing reagents that would result from treating a lesser ammoniacal load in the legacy WWT plant. Incremental processing costs are influenced by the high NH_3 concentration (which in this illustration is not reduced via low-cost liming), giving rise to a high demand for costly struvite reagents. The reagent cost is only partially offset by the sale of recovered struvite. Considering the incremental power, equipment and other operating costs applicable to the struvite protocol, and further considering the availability of legacy processing capabilities at the existing WWT plant, which avoids any significant requirement for new capital investment to maintain the existing processing protocol, the struvite protocol is unlikely to be economically sustainable in the context of this business model at this specific site.

4.1.2 Light leachate streams

Test results show that a protocol combining light liming, struvite precipitation and filtration/RO reduces NH_3 concentrations from 38 ppm to <5 ppm, which would allow leachates to be discharged to a local waterway (subject to industrial emissions authorisation) and thus avoid costly transport to a municipal WWT plant for treatment.

From an environmental perspective, processing 260,000 m³ of such leachate, as produced by the test County Limerick site, per annum would remove

8580 kg of NH_3 from the discharge stream. More generally, this approach would treat the effluent stream to achieve an ammoniacal discharge standard that could meet the licensed ELV for discharges at most municipal WWT plants. On-site processing would also reduce capacity pressures at WWT plants, reducing the risk of non-compliant discharges. The process generates a small fraction of waste lime/mineral sediment, disposal of which would be managed on-site. On-site processing would avoid the transport-related GHG emissions that would otherwise be incurred in transporting leachate. However, a life cycle analysis would be required to evaluate whether GHG emissions generated as a result of the energy consumed by the combined protocol are greater or less than the GHG emission savings resulting from the reductions in energy intensity and reagent demand arising from the decrease in the ammoniacal load of the WWT plant. In addition, a life cycle analysis would be required to evaluate the impact on GHG emissions of the displacement of carbon-intensive conventional fertilisers with struvite synthesised using recovered NH_3 .

From an economic perspective, transport and processing costs at the County Limerick landfill site WWT plant are significant, ranging from €8 to €15 m^{-3} of leachate, depending on the receiving location. On a scale of 260,000 m^3 per annum, estimates of net processing costs using the combined protocol ranged between €3.00 and €5.00 m^{-3} of leachate, which is less than the estimated cost of transport and processing. This moderate processing cost is driven by a relatively low demand for struvite reagents, which is driven by the low ammoniacal load. A low ammoniacal load would also lead to a lower hydraulic retention time, reducing vessel size and agitation energy requirements. Deployment of a low-pressure RO system (e.g. operating at 8–10 bar) requires much less power (2–3 $\text{kWh}_e \text{m}^{-3}$) than the high-pressure systems normally associated with potable water desalination, for example. Given the high cost of transport and treatment at municipal WWT plants, it is likely that this type of application would be economically sustainable.

4.2 Sustainable Slurry Management

Mitigating the environmental impacts of concentrated slurry land spreading requires technology and changes in slurry management and fertiliser practices that support the displacement of high-loss nutrients from

livestock slurries with low-loss nutrients to maintain crop productivity. In this project, treatment test results show that a combined protocol – incorporating bulk solid removal via screening or screw press and colloidal solid removal using the ash filter aid combined with NH_3 stripping via liming – can reduce slurry NH_3 concentrations from c.2500 ppm to < 500 ppm. Vaporous NH_3 can potentially be recovered for synthesis of low-loss fertilisers, although additional technology development is required to validate this concept. Subsequent application of a struvite protocol can further reduce NH_3 concentrations to < 20 ppm, facilitating recovery of struvite that can potentially be used as a fertiliser. In each case, process effluents are sufficiently clarified to enable discharge to a receiving land bank via pumping, avoiding the diesel-fuelled tractor movements required for slurry land spreading. The size of the discharge land bank is determined by the limit values included in Nitrates Directive regulations, and is influenced by the process configuration (i.e. whether effluent discharge occurs post stripping or post struvite protocol application).

Results show that biomass solids can be recovered as a renewable energy feedstock. Lime conditioning improves biomethane production in AD, although co-digestion with a supplemental feedstock is required to achieve economically viable biomethane outputs. In the recovery of renewable heat energy via thermal technology, lime conditioning assists moisture removal but increases the ash content to a level unsuitable for energy recovery via pyrolysis. Co-combustion with a feedstock of higher calorific value is possible, provided that the calorific value of the combined feedstocks is suitable for sustaining combustion. Both AD and combustion enable the renewable energy derived from biomass solids to be incorporated into the energy mix.

The introduction of a sustainable slurry management protocol would be expected to generate favourable environmental impacts. While quantification of absolute values is dependent on numerous variables, for purposes of illustration consider that the ammoniacal load in 200,000 m^3 of slurry with 2500 ppm NH_3 is 500,000 kg. If ammoniacal N losses to the atmosphere amount to 15% (Nyameasem *et al.*, 2022), the loss would be 75,000 kg. Consider also the following:

- the ammoniacal load from slurry land spreading is replaced with the ammoniacal load from

200,000 m³ of treated effluent with NH₃ concentrations ranging from 500 ppm to 50 ppm (depending on process configuration);

- the NH₃ from treated effluent is recovered as low-loss fertiliser;
- treated effluent discharge is supplemented with low-loss fertiliser applied at rates that achieve full replacement of the ammoniacal N load from slurry application.

If the above conditions were met, then, provided that the atmospheric NH₃ loss rate was no more than 2% from low-loss fertilisers (Teagasc, 2018), the comparable ammoniacal N equivalent loss would range from 23,000 kg to 11,300 kg.

Similarly, in respect of loss to run-off or leaching, if N loss rates from slurry land spreading are 10% of the ammoniacal load, compared with similar loss rates via the application of effluent and 5% loss rates via the application of low-loss fertiliser (Teagasc, 2018), then the ammoniacal N loss to run-off from land spreading would be 50,000 kg, whereas the comparable ammoniacal N equivalent loss from combined effluent and low-loss fertiliser application would range from 30,000 kg to 25,500 kg.

While the above illustrations are conceptual, they indicate that displacement of high-loss sources of N with lower-loss sources of N could make a significant contribution to achieving air and water quality objectives. In practice, the levels of N loss from slurry land spreading vary considerably depending on factors such as application methods, temperature and weather. While loss rates from the application of low-loss fertiliser may exceed the rate used in the above illustration, loss rates from slurry-derived N may be significantly higher than the values used in the above illustration. It is also likely that crop productivity could be maintained by displacing slurry-derived N with lower volumes of mineral fertiliser N. Both of these factors suggest that contributions to achieving air and water quality objectives could exceed the levels suggested by the illustrations above.

In addition to contributing to achieving air and water quality objectives, a sustainable slurry management protocol may also contribute to reducing GHG emissions. While quantification of the net benefit or burden in terms of GHG emissions requires a detailed life cycle analysis (which is beyond the scope of this

project), GHG emission reductions could potentially arise as follows:

- A reduction in soil GHG emissions could result from the displacement of high volumes of high-loss slurry-derived N with lower volumes of low-loss mineral fertilisers.
- A reduction in the carbon intensity of fertiliser production could arise if mineral fertilisers produced via the Haber–Bosch process are displaced with fertilisers produced using NH₃ recovered from livestock slurries with a lower carbon footprint. The carbon intensity of NH₃ recovery and fertiliser synthesis has yet to be established, and this will need to be addressed in the next phase of work. The energy used, for power, transport and lime calcination, in NH₃ recovery will generate some GHG emissions. GHG emissions could be significantly reduced by using renewably sourced electricity and through the development of a lime regeneration technology that uses renewable energy.
- A reduction in methane emissions could result from improved farm slurry management protocols.
- A reduction in energy-related GHG emissions could be achieved by the displacement of energy generated from fossil fuels with renewable energy sourced from slurry solids.
- A net reduction in GHG emissions from agricultural transport could arise from the displacement of the diesel-fuelled tractors used for slurry land spreading with electrically powered pumps discharging treated liquor effluents to small land banks. While reduced tractor use will be partly offset by an increase in the heavy goods vehicle movements required to manage the slurry processing supply chain, carefully choosing the location of processing sites will mean that fewer (high-capacity) road haulage trips can replace numerous low-capacity tractor land spreading trips. Moreover, road haulage vehicles can use renewable fuels, displacing tractor diesel fuel and the associated transport-related GHG emissions.

The achievement of anticipated environmental benefits requires an economically sustainable business model that promotes technology commercialisation and deployment, and changes to the energy mix and slurry management and fertiliser practices. Structured discussions with stakeholders indicate that the

business model will be complex, addressing a range of factors, as described below.

4.2.1 A community-based hub and spoke model to facilitate supply chain development

The Irish agricultural sector is fragmented (Table 4.1). The average farm size in Ireland is 32.4 ha (Department of Agriculture, Food and the Marine, 2020) and the average dairy herd is 84 cows (Teagasc, 2020). The combined technology suite developed and tested here (Chapter 3) requires a capital investment that, while modest, is unlikely to be cost-effective on the individual farm scale. Operationally, it will be more efficient if slurry can be aggregated for processing on a scale ranging between 5000 and 50,000 m³ per annum. Fragmentation in the agricultural sector poses barriers to the efficient introduction of renewable energy and other technologies directed at improving sustainability, especially where scale is required for economic sustainability (Sustainable Energy Authority of Ireland, 2017a). Conceptually, to overcome the potential constraint posed by a fragmented supply chain, it may be possible to organise community-based schemes into a “hub and spoke” configuration (Sustainable Energy Authority of Ireland, 2017b). A model is proposed whereby the slurry resource is collected from individual small farms for aggregation at local agri-spoke processing sites (e.g. local agricultural contractors or piggery operators) that can leverage on the availability of existing personnel, plants and equipment and tap into existing relationships with neighbouring farmers to promote participation.

Clarifying effluents for local discharge could support exploitation of the residual nutrient value to grow grass for silage. This potentially offers high-volume slurry owners methods to sustainably manage their existing slurry supplies at a lower cost (e.g. piggeries can avoid the €4–6 t⁻¹ cost normally incurred to have

slurry hauled away for land spreading). It will support increased investment in localised buffer storage, reducing pressures on small farmers to expand costly on-farm slurry storage (and possibly reduce non-compliant land spreading driven by storage capacity constraints). It may enable the use of collection protocols that minimise agitation requirements, reducing intense atmospheric discharges that would otherwise occur during slurry agitation. It may facilitate the effective introduction of technology to inhibit methanogenesis, avoiding atmospheric discharges of N and methane that would otherwise occur during storage and land spreading.

Localised solid separation and stabilisation, and NH₃ stripping and struvite processing may generate recovered fractions, the valorisation of which will be optimised if they are productised to meet regulatory requirements and consumer expectations. Productising and marketing, however, will most likely require specialist skill sets and high-tech equipment. Unit costs are likely to be reduced, and economic sustainability is likely to be enhanced, by spreading these costs across large product volumes. It is proposed that locally recovered process outputs would be transported to a centralised “hub” that could provide the specialist productising, marketing and support required to generate economies of scale. While this concept requires testing and validation, it emulates the dairy cooperative model that was used so successfully to build the Irish dairy sector.

The REBIOGEN review noted the importance of organisation, as community-based schemes may offer a means to structure ownership such that stakeholders are entitled to participate, creating broad-based community support and mobilising access to feedstock supply (Sustainable Energy Authority of Ireland, 2017b). A community-based centralised management function may facilitate improved reporting and accountability and more effective policy implementation, as processing and management protocols can be standardised. Community-based models may allow access to preferential sources of public finance from programmes such as Ireland’s Climate Action Fund (Department of the Environment, Climate and Communications, 2020) or state-backed loans from the European Investment Bank, which may otherwise be unavailable to private commercial schemes.

Table 4.1. Structure of Irish agriculture in 2016

Parameter	Total
Number of farms	137,500
Utilised agriculture area	4.459 million ha
Average farm size	32.4 ha

Source: Central Statistics Office.

4.2.2 Promoting stakeholder engagement to mobilise transfer of slurry resource to low-loss fertilisers

Farmers have capital invested in slurry management equipment and follow the current nutrient planning and accounting protocol that establishes the N value of cattle slurry at 5 kg t^{-1} and pig slurry at 4.2 kg t^{-1} (Irish Statute Book, 2017). If slurry is removed from nutrient plans (Teagasc, 2023), replacement nutrients will be required, and this will incur an incremental cost. Recent reports (Walsh, 2021) highlight that the cost of CAN was anticipated to reach $\text{€}280\text{--}285 \text{ t}^{-1}$ ($\text{€}1.04 \text{ kg N}^{-1}$), while the cost of urea was expected to range from $\text{€}400$ to $\text{€}450 \text{ t}^{-1}$ ($\text{c.€}0.92 \text{ kg N}^{-1}$) by 2023.

However, slurry management is costly and inconvenient. Pig farmers may haul slurry for distant land spreading unless local tillage farmers have acceptance capacity (mainly over a limited time frame in the spring). Haulage costs can range from $\text{€}4$ to $\text{€}6 \text{ t}^{-1}$, and concerns over soil quality mean that the ability to locate land banks capable of receiving slurry is diminishing. In respect of cattle farms, land spreading is messy, tedious work, requiring multiple vehicle movements and incurring fuel costs (spreading 100 t of slurry with a 2000-gallon slurry spreader requires 11 or 12 tractor movements). For larger dairy farms, obtaining and retaining a Nitrates Directive derogation (allowing spread of 225 kg N ha^{-1} as opposed to 170 kg N ha^{-1}) requires specialist equipment, which can be costly. It is common for derogation farmers to pay agricultural contractors to undertake land spreading, at a cost of $\text{c.€}300$ to spread 100 t of stored slurry.

To incentivise farmers to supply slurry for sustainable processing to low-loss fertilisers, a model is proposed whereby farmers obtain certification for the removal of N (based on slurry volumes collected from farms) and can then redeem their certificate for either a quantity of cash or low-loss fertiliser that is equivalent to the level of N discharges avoided. For illustration purposes, if the redemption value was set at $\text{€}1.30 \text{ kg}^{-1} \text{ N}$ removed (i.e. average $\text{€}6.50 \text{ t}^{-1}$ cattle slurry or $\text{€}5.85 \text{ t}^{-1}$ pig slurry), this might make an acceptable contribution to the cost of slurry removal (e.g. the slurry processing fee is incorporated at $\text{€}4.00 \text{ t}^{-1}$ in the illustrations below) and to the cost of replacing N with lower volumes of low-loss fertiliser. It may be possible to adapt the existing CAP programme to source

funding for this concept. If such a programme was implemented to target 10 million t of slurry annually (slightly less than half of the volume of slurry collected annually in Ireland), the cost to the exchequer would be $\text{€}65$ million annually, or $\text{c.}5\%$ of the $\text{€}1.2$ billion direct farm payment budget (Department of Agriculture, Food and the Marine, 2021c). While this concept is consistent with CAP reforms that support climate and biodiversity objectives (to be implemented in 2023, pending final agreement between Member States) (European Commission, 2022b), the degree to which such a system might overcome inertia factors, incentivise farmer participation and progressively transition Irish agriculture to more environmentally friendly fertiliser practices requires validation.

4.2.3 Securing routes for the valorisation of slurry-derived nutrient products

Transitioning to sustainable slurry management practices is likely to require public subventions to be offered as incentives to remunerate transition costs and overcome inertia. It was hypothesised that the impact on public finances could be minimised if energy and nutrient products are recovered from slurry feedstocks and sold, generating revenues to offset processing costs. Sales of slurry-derived energy and nutrient products are dependent on productising the process outputs to meet consumer and market expectations, and developing viable routes to market and meeting relevant regulatory requirements. The new FPR (European Union, 2019) established specifications for using the CE mark across different PFCs (organic fertiliser, inorganic fertiliser, organo-mineral blends, liming material, soil improver, growth medium, etc.). As well as specifying labelling requirements, regulations also specify limits on levels of contaminants and pathogens (disease-causing microorganisms), minimum nutrient content and other relevant characteristics (depending on the PFC). They establish acceptable CMCs from which fertilisers can be sourced, which include “derived products” that achieve an “end of process” designation pursuant to the Animal By-products Regulation (European Union, 2009), and govern the use of materials such as excreta-sourced derivatives. Finalisation of “end of process” criteria for ABP CMCs under the new FPR (European Union, 2019) may enable CE marking for slurry-derived nutrient products, as described below.

Lime recovery

Increasing soil alkalinity is known to enhance the availability of plant nutrients (Teagasc, 2022f), and the application of lime together with P generates a measurable increase in grass production (Teagasc, 2022f). Nägele and Conrad (1990) report a significant decrease in NO and N₂O emissions from soils where the pH is increased from acidic to neutral, suggesting that the regular liming of soils also reduces GHG emissions. Lime applied for NH₃ stripping settles out partially as calcium carbonate and partially as hydrated lime, which offers a greater neutralisation value than the dolomite or quarried lime normally applied to farm fields. Agricultural application thus offers a potential route for the valorisation of recovered lime. Under new fertiliser regulations, clarification will be required as to whether lime recovered from NH₃ stripping is considered an ABP or derivative, as it is sourced from slurry; if it is considered an ABP, finalisation of the criteria for incorporation of ABPs into fertiliser products will be required before it can qualify as a CE-marked product.

Ca(OH)₂ is costly (supplier discussions indicated a 2021 cost of c.€150 t⁻¹, increasing to >€225 t⁻¹ as of August 2022), which at scale gives rise to a significant input cost, affecting commercialisation prospects. Methods to mitigate this cost were considered, including lime regeneration via calcination, which is common in process industries such as pulping or sugar refining (Sudhir Kumar *et al.*, 2007). Calcinating CaCO₃ is highly endothermic, requiring c.0.88 megawatts thermal (MWt) of heat to produce a tonne of CaO, which can then be hydrated to Ca(OH)₂. In 2021, natural gas costs ranged from €40 to €80 MWt⁻¹, but these had doubled by August 2022; however, significant cost savings relative to the procurement of hydrated lime may still be possible via this route. It may also be possible to reduce the carbon footprint of lime regeneration. Valmet (2019) advertises a biomass-fuelled lime kiln, while Yang *et al.* (2020) report that, by using heated CO₂ as the carrier gas, the calcination process can be modified to capture a “clean” CO₂ stream, avoiding the parasitic energy inputs otherwise required to separate CO₂ from conventional combustion flue gases. CO₂ capture may facilitate valorisation options, such as synthesis of sustainable urea (which would effectively demonstrate use of sustainably sourced NH₃ to mitigate CO₂ emissions). While these concepts are attractive from

an environmental standpoint, process development, optimisation and validation are required to determine their prospects for commercialisation.

Filter media recovery

Ash from the combustion of ABPs, such as poultry litter and meat and bonemeal, is high in P and K. During combustion, P and K are mineralised into inorganic forms. Mineralised K is more soluble than P, which makes it almost 100% plant available; there are numerous forms of mineralised phosphates in ash, but the plant availability is dependent on the forms present and their relative concentrations in the ash (Wilhelm and Steingrobe, 2018). The combustion of meat and bonemeal, for example, generates an ash comprising insoluble calcium apatites, in which P is not readily plant available. The low solubility of different forms of calcium phosphates means that the P is released over time, based on soil moisture, pH and other factors. A transition to slow-release P fertilisers may reduce the run-off losses currently associated with highly soluble P fertilisers, although the environmental impact must be balanced against near-term crop requirements. Optimising the value of P in ash fertilisers may require new techniques to improve near-term plant availability or possibly a change in application protocols that might include blending more soluble forms of plant-available P with slow-release forms.

The co-combustion of dried slurry solids with poultry litter or meat and bonemeal (currently used as solid fuels) may generate a P- and K-rich ash suitable for use as a filter aid. The recovery of spent filter aid may facilitate valorisation as an agricultural fertiliser, subject to compliance with specifications in the new FPR (European Union, 2019). Commercialisation of this concept is dependent on several factors, including the concentration of P and K in the ash, the forms of P and K in the ash, which determine their relative plant availability, the market value of the respective nutrients and farmers' willingness to accept alternative forms of plant nutrition. The value of the P and K in various feedstocks is estimated in Table 4.2.

The values of P and K in pig slurry, cattle slurry and poultry litter were reported by Komiyam *et al.* (2013), Tran *et al.* (2017) and Lynch (2013), respectively. The estimated values were calculated based on 80% of the relative price of P (€0.83 kg⁻¹ as determined by current spot price of triple superphosphate at

Table 4.2. Estimate of P and K values in ash from proposed combustion feedstocks and potential value

	Pig slurry	Cattle slurry	Poultry litter	Meat and bonemeal
P (kg t ⁻¹ DM)	112	42	110	113
K (kg t ⁻¹ DM)	87	155	170	15
Estimated mineral value t ⁻¹ ash (€)	179.08	188.31	259.60	108.64

US\$575 t⁻¹ (YCharts, 2022a) using an exchange rate of US\$1.16 to €1) and 100% of the relative price of K (€0.99 kg⁻¹ as determined by the current spot price of muriate of potash at US\$221 t⁻¹ (YCharts, 2022b) using an exchange rate of US\$1.16 to €1). Lynch (2013) uses a similar method to estimate value, and Reiter (2011) draws similar conclusions in a study on productising poultry litter ash as a fertiliser. Reiter also notes that nutrient value may be optimised if ash can be granularised to facilitate easy spreading. Moisture assists granularisation, and so incorporation of moist colloidal solids in the spent filtrate may be beneficial, although this must be validated.

Both poultry litter and meat and bonemeal are available in significant quantities in Ireland, suggesting that there are prospects for developing a viable supply chain for these input materials. Some companies currently use ABP ash in commercial fertiliser products. FibroPhos, for example, is a UK company advertising the incorporation of poultry litter ash into blended fertiliser products, with different blends of P/K nutrients oriented to different crop varieties. Marketability and value will, in part, be determined by the supply of nutrients relative to specific crop requirements. Risks could potentially arise from concentrations of metals in ash that exceed the maximum thresholds specified in the new fertiliser regulations. Römer and Steingrobe (2018) review techniques for reducing heavy metal concentrations, including through the use of a rotary kiln to reheat materials to c.800–1000°C together with a chlorine donor (CaCl₂ or MgCl₂) to generate metal chlorides that, at temperature, become volatile and can be separated in a gas stream. This results in the generation of decontaminated ash with a P concentration of c.6–11%, which would qualify for use as a P fertiliser or as a raw material for the production of mineral P fertilisers. If required, it may be possible to adapt these techniques to modify heavy metal concentrations in excreta ash to facilitate compliance with the EU fertiliser regulations. Further processing of the ash may also facilitate product enhancement. Leng

et al. (2019) demonstrate acid-based techniques to recover P from meat and bonemeal ash and methods to convert the P from less plant-available forms to more useful plant-available forms, while Fiameni *et al.* (2021) test simultaneous P and silicon recovery using a staged alkaline and acid process to achieve the same. These findings suggest that methods are available to secure the P fertiliser value in the event that it cannot be monetised in its original form.

To better understand the ease or difficulty with which the P value in ash might be recovered, a bench-scale scoping exercise was undertaken to determine how the ash reacts when dissolved in acid. Ash samples were dissolved in concentrated nitric acid followed by filtration via filter paper to separate the soluble fractions from the residual solids. Lime was added to the filtrate solution, resulting in precipitation of calcium phosphates. The scoping tests established that the P in the ash was soluble and could be recovered via application of nitric acid, although, as this was only a scoping exercise, further work is required to validate and quantify the results in a rigorous scientific investigation to determine whether or not this process could be useful for adding value on a larger scale.

Vaporous ammonia recovery

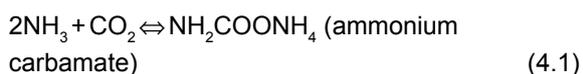
Vaporous NH₃ can potentially be recovered and productised in the form of NH₃-based fertilisers (e.g. CAN, MAP, DAP) with established N/P/K values and established routes to market. The synthesis of ammonium sulfate fertiliser from stripped NH₃ is a commercially proven process, as discussed above. Synthesis of other types of fertilisers from recovered NH₃ requires further technology development.

MAP and/or DAP can potentially be synthesised by diffusing NH₃ vapour through highly concentrated solutions of phosphoric acid (H₃PO₄ in 85% solution). Phosphoric acid concentrations must be maintained at high levels or precipitation is not triggered, as the NH₃ stays in solution. A bench-scale scoping exercise was undertaken to evaluate the feasibility with which this

reaction might be incorporated into a nutrient recovery system. Gaseous NH_3 was diffused through 50 mL of 85% phosphoric acid solution, generating a crystalline precipitate at ambient temperature and pressure. X-ray diffraction analysis of the precipitate confirmed the process output as MAP. The scoping exercise indicated that, if a concentrated NH_3 stream can be sourced from ammoniacal liquors, it may be feasible to valorise the NH_3 recovered. Additional investigation is required to further evaluate the prospects for MAP/DAP synthesis within the context of a system of ammoniacal nutrient recovery.

CAN (Cheremisinoff, 1995) is a widely used fertiliser, accounting for c.4% of N fertiliser application worldwide, that contains roughly 8% calcium and 21–27% N. CAN may be synthesised via a double salt reaction in which first nitric acid is combined with NH_3 to generate ammonium nitrate in an aqueous solution, and then powdered calcium carbonate is added to facilitate crystallisation and recovery. As noted above, lime and NH_3 were constituent components recovered from slurry processing, but investigation is required to determine if any process inhibition arises from the use of recovered lime first used as an alkaline agent for NH_3 stripping or if any other inhibitory issues or engineering considerations must be addressed. While CAN synthesis is a known process, the ammonium nitrate intermediate is potentially explosive. Specialist equipment and facilities and health, safety and reporting protocols will be required before this recovery route is further investigated, which precluded further development work being undertaken within the scope of this project.

Urea is normally synthesised from NH_3 sourced from the Haber–Bosch process and CO_2 . It is a carbon-intensive production process, generating 0.91 kg $\text{CO}_{2\text{-eq}}$ kg^{-1} of urea produced (Hoxha and Christensen, 2019). NH_3 and CO_2 are combined at high pressure and temperature, forming urea in a two-step reaction:



The urea is in equilibrium with unreacted NH_3 , CO_2 and ammonium carbamate. As pressure is reduced and heat applied, $\text{NH}_2\text{COONH}_4$ decomposes to NH_3

and CO_2 . The unreacted NH_3 and CO_2 are recycled. The reactants in solution are then concentrated to give 99.6% w/w molten urea, and subsequently granulated for use as fertiliser and chemical feedstock. Barzagli *et al.* (2016) describe an alternative approach that may reduce the carbon footprint of urea synthesis whereby CO_2 and NH_3 are combined in a highly concentrated ethanol solvent at low temperature to generate mixtures of ammonium carbamate and bicarbonate. The ammonium carbamate is converted into urea by heating to 428 K for 60–90 minutes, generating yields of up to 53% on a molar basis. The research indicates that excess amounts of water in the process tend to drive the initial reaction towards bicarbonate rather than the carbamate required for urea synthesis, but that maintaining this reaction in a highly concentrated ethanol solvent at a moderately low temperature drives carbamate formation. In a short scoping exercise to determine the relative ease or difficulty with which this reaction might be leveraged for nutrient recovery, the project attempted to replicate the synthesis of ammonium carbamate by adapting the reported protocols (e.g. relying on pressure release of pressurised NH_3 and CO_2 to generate the cooling effect in a slightly pressurised reactor at 2 bar and using a methanol rather than ethanol solvent). It was observed that combining NH_3 and CO_2 in methanol at 2 bar was slightly exothermic, and that the resultant heat made it difficult to maintain the required reaction temperature without using energy for cooling. The initial tests failed to generate ammonium carbamate, and time and budgetary constraints precluded further development work on this task.

Struvite recovery

Struvite reagent costs can range from €400 to €650 t^{-1} , depending primarily on the price of P. Struvite is a hydrate that increases the mass of the end product relative to the stoichiometric mass of the non- H_2O components. It has an NPK value of 6-29-0 and can be applied on its own or used as a component of compound blends. Römer and Steingrobe's review (2018) references test results indicating that the average P uptake values from three struvite samples were 91%, 100% and 71% relative to P sourced from triple superphosphate, suggesting that the plant availability of struvite P is only slightly less than that of soluble triple superphosphates. While soil structures may influence plant uptake of P from mineralised

sources, the research concluded that struvite is a suitable plant fertiliser. Szyman *et al.* (2019) report that the slow-release multi-nutrient mechanism allows the released P to be more efficiently used by crops than P from fast-release artificial fertilisers and that struvite may limit the soil P losses that typically occur under treatment with fast-release artificial fertilisers. However, the low solubility of struvite may result in an insufficient supply of P to crops, particularly during the early phase of growth, which may reduce its fertiliser value. The report references studies that confirm that the solubility and uptake by plants of P from struvite is comparable to that from triple superphosphate or potassium phosphate (KH_2PO_4). It also notes that incomplete dissolution in the first year after application may result in a residual soil P value in succeeding years, which may offer benefits to grasslands and forests, where fertilisers are applied once every several years (Szyman *et al.*, 2019). Given that the value of struvite is primarily driven by the level of plant-available P, the value should reasonably offset input costs, provided that slurry-derived struvite complies with specifications in the EU fertiliser regulations.

Recovery of bulk slurry/digestate solids

The use of slurry or digestate solids as fertiliser products is also envisaged in the new FPR (European Union, 2019) (PFC classifications have been established for both organic fertilisers and organo-mineral fertiliser blends). Given that most agricultural land in Ireland is grass pastureland (which is less depleted in organic matter than intensively tilled soils are), the demand for organic fertilisers may be limited. In markets where grasslands predominate, supplying nutrients in low-loss mineral forms that are easily applied is likely to be of greater value than supplying nutrients as organo-mineral blends. In mineral soils, such as those common in Mediterranean climates, organo-mineral blends may be of higher value. Retail growth media products may also provide a high-margin route to market (although on a limited scale). Excreta-derived growth media products, such as pelletised chicken manure and packaged farmyard or equine manures, are accepted in the market and available at retail prices ranging from €5.00 to €10.00 for a 5–10 kg package. Provided that the CMC criteria enable ABP derivatives to be incorporated into CE-marked fertiliser products and provided that fertiliser products are produced that meet the agreed PFC criteria, it is

likely that broader market outlets will be developed for slurry-derived nutrient products.

Recovery of residual potassium in effluents

The project evaluated the potential for use of RO to concentrate K and sodium (Na) in a retentate (c.20% of original solution by volume) followed by a multi-stage evaporation to concentrate the resultant KOH to 16% in solution by mass. An estimated price for KOH of c.€600 t⁻¹ suggests that the target concentration of 160 kg KOH t⁻¹ solution might be valued at c.€96 t⁻¹ solution. Achieving the target concentration in solution would require removal of c.100 t of water for every tonne of solution generated. The RO and the multi-stage evaporation process are energy intensive. Power requirements for RO range between 3 and 8 kilowatt hours of electricity (kWh_e) per tonne permeate removed, while estimates of thermal energy required to remove 1 t of water range between 0.45 and 1 MWt (depending on heat recovery and process configuration). While process economics are dependent on the cost of the energy inputs, given current energy prices it may be difficult to justify this step purely from an economic standpoint. It was concluded that, while the technology exists to recover the K in the liquor, a qualitative judgement is required as to whether or not this last stage is desirable, taking account of cost, environmental and other factors that are specific to the location/application.

4.2.4 Securing routes for the valorisation of slurry-derived biomethane

Recent updates to the EU's 2030 Climate and Energy Framework incorporate revisions to raise the binding target for the share of renewable energy sources in the EU energy mix to 40% by 2030, which will not meet the Paris Agreement commitment to reduce emissions of CO_{2-eq} by at least 40% by this year (European Commission, 2021b). Acceleration measures, directed at increasing the proportion of renewable energy sources used for heating and cooling and transport, are designed to close the gap in achieving the 40% emission reduction objective.

Biogas is widely viewed as a contributor to Ireland's renewable energy and environmental obligations; however, the higher cost of biogas than of natural gas is a barrier to wider deployment. IEA Bioenergy reports

that the cost of biogas ranges from c.€70 MWh⁻¹ to €120 MWh⁻¹ (Thr n *et al.*, 2014) and is influenced by factors including feedstock supply, biogas yield, plant capital costs and biogas upgrade requirements. The processing system for ammoniacal streams discussed herein can potentially make AD more economically feasible. Liming separated slurry solids increases biogas yield and assists air drying, which reduces transport burdens if solids are routed to off-site AD plants. At scale, the cost of digestate management is significant. Managing the digestate from a 1500-m³ digester processing 30 kt of feedstock can require more than 1200 vehicle movements. Depending on how the ABP provisions in the new fertiliser regulations are finalised, recovering digestate solids and nutrient fractions as fertiliser products may provide a revenue source to offset costs. Moreover, clarifying digestate liquor effluents to a standard suitable for local discharge with minimal environmental impact would avoid the need for transport.

With minimal conditioning, raw biogas (i.e. at c.65% methane, with moisture and H₂S removed) can fuel co-located combined heat and power or process heat applications. Processing slurry solids to a transportable, aesthetically acceptable standard enables the cost-efficient supply of slurry solids from multiple sources, supporting co-digestion in larger-scale AD plants that benefit from economies of scale and the location of AD plants at the point of energy use (as opposed to the point of feedstock supply), which may improve deployment prospects for this type of low-cost biogas application.

When biogas is upgraded to biomethane, it can be distributed via the gas grid as a natural gas replacement. For grid injection, biomethane must meet the Gas Grid Code (2017) standard, which can be challenging and costly. It also requires a connection to the gas distribution network, which can be capital intensive. In terms of pricing, grid-supplied biomethane competes with the (carbon-taxed) wholesale price of natural gas, which recently increased to c.€50–60 MWh⁻¹ from <€25 MWh⁻¹ (S&P Global, 2022). Despite this, significant market interventions may be required to enable biomethane to compete with grid-distributed natural gas. Alternatively, upgraded biomethane can be compressed to c.200 bar, containerised and delivered via road haulage. Delivery via the virtual grid facilitates retail price competition with liquefied petroleum gas or other distillates,

the prices of which are reported to have ranged between €100 and €160 MWh⁻¹ (VAT and carbon tax inclusive) during the last half of 2021 (Sustainable Energy Authority of Ireland, 2022a). Even considering the additional cost of compression (estimated at c.€20 m⁻³), this route to market may be attractive in rural areas, where distillates are a principal source of fuel.

One of the most promising potential applications of slurry-derived biomethane is its use as a renewable transport fuel for heavy goods vehicles. While electrification is preferred for improving private car sustainability, there are few viable options for improving the sustainability of heavy goods transport. The transition to natural gas for vehicles facilitates compliance with the increasingly strict emissions standards (Euro VI) governing heavy goods transport, as natural gas combustion generates significantly lower levels of particulates than diesel. Biomethane can qualify as a renewable fuel provided that it meets standards detailed in the revised Renewable Energy Directive (RED II) (European Union, 2018). Recent increases in oil prices and increases in taxation applied to fossil fuels used for transport have driven the price of petrol and diesel up to levels that could make biomethane a competitive alternative, provided that it continues to be excluded from excise and carbon taxation.

4.2.5 *Microeconomic illustrations*

Technology for the sustainable management of slurry can be configured differently, depending on operational circumstances. To demonstrate how this impacts on economic sustainability, microeconomic illustrations were prepared, comparing the projected results of operations under two different scenarios. In each scenario, operations are configured across five agri-spoke sites deploying plants to recover solids and nutrients in forms transportable to a central processing hub. Slurry liquors are processed at each agri-spoke site to facilitate local discharge in compliance with Nitrates Directive regulations. The agri-spoke sites are sites that have existing personnel and infrastructure that can be used to aggregate an on-site slurry supply from slurry collected from neighbouring farms. The illustrations assume a fee of €4.00 t⁻¹ for slurry collection and processing in return for an N removal certificate that can be redeemed for the purchase of

low-loss fertiliser. Transfer pricing for soil improvement products (limed solids, spent filter aid and spent stripping lime) is set at €125t⁻¹, while struvite is valued at €325t⁻¹ and NH₃ at €312t⁻¹. For the purposes of the illustrations, it is assumed that all recovered fractions are suitable for productising in compliance with the EU fertiliser regulations. Electrical power is sourced from the grid and thermal energy is sourced from the on-site combustion of P-rich biomass (e.g. poultry litter) collected at no gate fee. Ash from combustion is recovered for use as a filter aid. Biomass as a source of renewable heat attracts support from the Support Scheme for Renewable Heat subvention. Any benefits from lime regeneration are not considered.

Finance is incorporated as a combination of private investment, grant aid at 50% of projected capital equipment costs and low-cost government-supported loans.

The “LITE” scenario is based primarily on nutrient recovery, incorporating a protocol comprising bulk solid separation and conditioning for co-combustion, followed by colloidal solid filtration, NH₃ stripping and struvite precipitation. It excludes energy recovery via AD and liquor polishing via RO.

The “INTENSE” scenario is based on AD, co-digesting conditioned slurry solids with a supplemental feedstock and upgrading the biogas generated to biomethane, valued at €1.10t⁻¹. It incorporates the NH₃-stripping of filtered liquor prior to use of the conditioned liquor

as the AD process medium. It applies the multi-stage screening–filtration–stripping–struvite protocol to the AD digestate, conditioning digestate solids for co-combustion. Digestate liquors are polished via RO prior to discharge.

All costings were based on 2021 prices. Inflation during 2022 will have significantly increased processing costs, interest rates and other factors, but will also have influenced the value of the recovered products. Additional detail on the generation of costings can be found in the SLURRES pilot report (Sustainable Energy Authority of Ireland, 2022b).

The LITE scenario (Tables 4.3–4.5) shows that projected revenues from the valorisation of recovered nutrients would be sufficient to cover projected direct operating costs; however, they would not cover the depreciation and financing costs relating to investment in the plant and equipment. This is confirmed by the sources and uses of funds illustration, which shows that, over 10 years, the total funds from operations would not be sufficient to repay initial borrowings, even when 50% of the projected investment in the plant and equipment is assumed to be funded via grant aid. Coupled with a projected enterprise value of c.€11.2 million (based on 10 × year 5 EBITDA (earnings before interest, taxation and depreciation)), this scenario is unlikely to be able to attract the projected initial private investment of €8.5 million required and is unlikely to be economically sustainable.

Table 4.3. Comparative microeconomic illustration: extracts from projected operational parameters

Projected operational parameters (year 5 – projected year of maturity)	Five agri-spoke sites combined	
	Scenario: LITE	Scenario: INTENSE
Pig slurry processed (6% solids) (t)	100,000	100,000
Cattle slurry processed (9% solids) (t)	150,000	150,000
Supplemental co-digestion feedstock (t at 20% DM)	–	100,000
Supplemental co-combustion feedstock (t)	11,700	11,700
Recovered spent filter aid, including colloidal solids dried to 80% DM (t)	23,957	33,564
Recovered lime solids dried to 80% DM (t)	11,636	22,174
Recovered struvite dried to 80% DM (t)	1103	2170
Recovered NH ₃ in concentrated solution (t)	661	1337
Biomethane output from AD (m ³)	–	7,252,000
Struvite effluent discharge at 20 ppm NH ₃ , 1250 ppm K (m ³)	164,710	–
RO permeate discharge at 2 ppm NH ₃ , 50 ppm K (m ³)	–	259,350
RO retentate available as liquid fertiliser at 100 ppm NH ₃ , 6000 ppm K (m ³)	–	64,835

Table 4.4. Comparative microeconomic illustration: extracts from financial projections

Financial projections (year 5 – projected year of maturity)	Five agri-spoke sites combined	
	Scenario: LITE (€000s)	Scenario: INTENSE (€000s)
Cumulative capital investment		
Cumulative capital investment in plant and equipment	44,844	108,103
Revenues		
Processing fees (supported by sustainable N subvention)	1080	1080
Recovered soil improver, fertiliser and NH ₃ product sales	5445	8796
Renewable energy sales (including Support Scheme for Renewable Heat subventions)	329	8889
Total revenues	6854	18,765
Operating costs		
Operating costs (excluding depreciation and amortisation)	5731	13,068
EBITDA	1123	5697
Depreciation expense	4122	10,221
Interest and taxation expense	548	1403
Net income (loss)	3547	5927

Table 4.5. Comparative microeconomic illustration: projected sources and uses of funds over 10 years

Projected sources and uses of funds over 10 years (cumulative to year 10)	Five agri-spoke sites combined	
	Scenario: LITE (€000s)	Scenario: INTENSE (€000s)
Sources of funds		
Contribution from commercial participants	8530	13,950
Grant aid	22,422	53,513
Borrowings (state backed)	14,108	35,242
Funds from operations	9323	51,601
Total sources of funds	54,383	154,306
Uses of funds		
Capital costs: plant and equipment	44,844	108,103
Debt service: principal repayments	3800	35,242
Interest on borrowings (4%)	4601	8985
Working capital requirement	105	110
Total uses of funds	53,350	152,440
Funds balance		
	1033	1866

The INTENSE scenario (Tables 4.3–4.5) illustration is slightly better, showing that projected revenues from both renewable energy and nutrient recovery would be sufficient to cover projected direct operating costs, yet insufficient to cover the depreciation and financing costs relating to the investment in the plant and equipment. The sources and use of funds illustration, however, suggests that, over 10 years, funds from operations would be sufficient to repay

initial borrowings. It estimates an enterprise value of c.€56.9 million (based on 10 × year 5 EBITDA), which may attract the projected initial private investment of €13.9 million. Economic sustainability, however, must be underpinned by capital grants covering 50% of the investment in the plant and equipment, as well as ongoing incentives for development of the slurry supply chain and for integration of renewable energy into the energy mix (including an excise tax derogation

for the use of biomethane as an alternative transport fuel, which supports a higher value than that from valorisation via combined heat and power generation).

While further technology development (e.g. lime regeneration and synthesis of NH_3 -based fertilisers) may improve the processing cost profile, transitioning to a sustainable slurry management protocol will be challenging and will require supportive regulatory and commercial frameworks that promote:

- cost-effective mobilisation/aggregation of slurry resources and displacement of high-loss slurry with low-loss mineral fertilisers, overcoming supply chain fragmentation and market inertia;
- timely access to suitable markets for slurry-derived fertiliser and energy products;
- timely access to planning, permits and licensing authorisations, enabling: multi-site plant deployments and inter-site transfers of materials recovered from slurries; co-incineration of P-rich agri-wastes for recovery of renewable energy, enabling use of ash (i) first as a filter aid and (ii) thereafter as a component of a fertiliser product; and calcination of recovered lime to regenerate slaked lime for re-use in NH_3 stripping;
- timely access to finance.

4.2.6 State aid considerations

Transitioning to sustainable slurry processing protocols will be capital intensive. The process technology is untested, and market outlets for the proposed energy and nutrient outputs are not yet mainstream. Commercial financial institutions struggle to finance immature technologies and it is likely that this market failure will initially have to be overcome with finance

provided from public sources, e.g. capital grants and low-cost, state-backed loans. Financing supports may have to co-exist with mechanisms to incentivise farmers to participate in developing the slurry supply chain and mechanisms to stimulate market uptake of renewable energy and nutrient product outputs. While individual measures would be targeted at different stakeholder groups (e.g. farmers, plant operators and renewable energy users), EU state aid regulations prohibit double funding for a given target application and provision of state assistance to any enterprise at levels that interfere with the normal working of commercial markets. Certain limited exemptions, however, are applicable in circumstances requiring transformation of social and commercial practices (e.g. energy decarbonisation and climate change). Ideally, separate support mechanisms would be structured to work together to incentivise participation of each separate stakeholder group. Where public funding commitments are required to progress deployment objectives, it may be useful to consider specialist financing mechanisms and stakeholder rights in respect of cash distribution to minimise the risks of falling foul of state aid constraints. Conceptually, it may be possible to structure public funding as mezzanine finance or low-cost loans that provide public funding bodies with the right to a return on capital and preclude the accounting of this type of finance as state aid. This concept requires testing and validation, and specific terms and conditions governing such distributions will have to be agreed on with the Department of Finance to strike the right balance between incentivising private capital contributions and compliance with state aid provisions for public funding contributions.

5 Socio-economic and Environmental Impacts

5.1 Consistency with State Policy Objectives

- **CAP objectives.** The sustainable slurry management objectives are directly aligned with recent agreements in respect of proposed reforms to the CAP, which incorporate a heavy emphasis on agricultural sustainability, contributions to rural socio-economic development and contributions to climate change and biodiversity objectives.
- **Renewable energy and climate change objectives.** The primary imperative behind development of the sustainable slurry management concept was the reduction in ammoniacal and GHG emissions to improve agricultural sustainability. In addition, the sustainable slurry management concept promotes the supply and use of advanced thermal technologies, combusting biomass for the generation of renewable heat, and the use of renewable wind power to generate the electricity needed for processing slurry and integrating AD. It promotes the supply and use of renewable biogas for process heat and power, and the use of upgraded biogas as a renewable transport fuel.
- **Circular economy objectives.** The recovery and recycling of plant nutrients is one of the goals stated in the EU's Circular Economy Action Plan.
- **Rural socio-economic development objectives.** Implementation of the sustainable slurry management concept will contribute to closing the urban–rural economic divide by attracting investment and creating sustainable employment in agricultural communities.
- **Knowledge-based economy objectives.** Consistent with the development of a knowledge-based economy, consolidating sustainable slurry management infrastructure into concentrated processing locations opens prospects for the development and application of biorefining or other industrial-scale improvements in processing protocols.
- **Prospects for efficient policy implementation.** The development of a centralised slurry management infrastructure under a defined management scheme may make policy

implementation more efficient via improved reporting, process monitoring, technology and implementation, which may otherwise be difficult in a fragmented agricultural sector.

5.2 Contributions to Farming Convenience, Cost Reduction and Agricultural Economic Sustainability

Implementation of a slurry management system may reduce environmentally derived pressures on stocking levels and agricultural production, as well as contribute to farming convenience, productivity and cost reduction:

- **Reduced pressure on slurry storage capacity and associated management costs.** Introducing a regular local slurry collection schedule and constructing an aggregated local slurry storage infrastructure may reduce the cost of constructing on-farm slurry storage facilities.
- **Reduced biohazard risk.** Recovering manure-derived nutrients in mineralised forms may minimise biohazard concerns associated with the application of unprocessed slurry to farmland.
- **Lower cost of and more convenient practices for fertiliser application.** The land spreading of slurry is a tedious, time-consuming exercise, requiring multiple tractor movements for a relatively low nutrient benefit. The application of fertiliser in mineralised forms requires much less time and lower-cost equipment than applying the same fertiliser load by slurry land spreading.
- **Improved productivity from the better matching of fertiliser supply with crop demand.** The convenience of applying fertiliser in granularised mineral form may make it much easier to apply fertiliser more often in smaller increments (possibly including the application via pumped umbilical systems of clarified slurry liquors that retain a K value). This may result in a better match of fertiliser supply with crop requirements, improving productivity and reducing loss to emissions and run-off. Avoiding the application of slurry in concentrated volumes may

reduce the short-term inhibition of grass growth or the “burning” that sometimes follows slurry application.

- **Reduced pressure to decrease stocking rates caused by environmental concerns.** Concerns over agricultural sustainability contribute to a public narrative that sometimes includes calls for herd reductions or environmentally driven constraints on the growth of the agricultural sector. The successful introduction of a sustainable slurry management system may offer a practical demonstration of how production can be maintained while agricultural sustainability is improved.
- **Economic benefits from stakeholder participation in a sustainable slurry management venture.** If appropriately structured, it may be possible for farm stakeholders to acquire an ownership interest in a community-based sustainable slurry management venture. Over time, it is possible that such an interest may become valuable, contributing to farm economic sustainability.

5.3 Contributions to Environmental Objectives and Agricultural Sustainability

- **Improved air quality from a reduction in ammoniacal emissions.** Slurry land spreading results in significant atmospheric NH_3 emissions. Replacing slurry land spreading with the application of low-loss mineral fertilisers, together with protocols for scheduled slurry collections and standardised storage and management practices, could significantly reduce the fugitive NH_3 emissions that result from current slurry management practices.
- **Improved water quality via a reduction in N run-off or leaching.** Slurry incorporates N in soluble NH_3 and NO_3^- forms. Land spreading practices include the intensive application of concentrated volumes over a short period. This results in an excess supply of soluble N, a portion of which is lost in run-off and leaching to groundwaters and surface waters, causing eutrophication and water quality issues. Low-loss mineral fertilisers, such as protected urea, are more convenient to apply and significantly reduce loss levels while maintaining crop productivity. Introducing a protocol that promotes the displacement of land spreading using high-loss slurry with the periodic application of (lower volumes of) low-loss mineral fertilisers will reduce water quality pressures.
- **Improved water quality via a reduction in P run-off or leaching.** When P fertilisers are applied at high concentrations in soluble forms (e.g. triple superphosphate), some of the soluble P is lost to run-off or leaching. The development of slow-release (multi-cycle) mineral P fertilisers may facilitate application protocols incorporating the application of lower concentrations of soluble P together with slow-release P fertilisers. In addition, the application of lime to adjust soil pH is known to increase the plant availability of P, and co-application of lime and P has been shown to result in increased grass growth. The integration of lime into a mineralised, slow-release P fertiliser, together with protocols that balance the application of a slow-release limed P fertiliser with lower concentrations of soluble P, may reduce run-off losses and agricultural pressures on water quality.
- **Reduced methane emissions via improved slurry management protocols.** The GHG effect of methane is 20 times more intensive than that of CO_2 . Standardising slurry management protocols may facilitate the introduction of methane management technology that will reduce the levels of methane emissions during storage and handling.
- **Reduced N_2O emissions via the avoidance of excess N.** The GHG effect of N_2O is 290 times more intensive than that of CO_2 . Minimising the presence of excess N in soils (via replacing land spreading using high-loss slurry with the use of low-loss mineral N fertiliser) will reduce the direct soil N_2O emissions resulting from the natural soil nitrification–denitrification cycle. Reducing ammoniacal volatilisation from soil and waterbodies will reduce the indirect N_2O emissions that arise as volatile N is oxidised.
- **Reduced CO_2 emissions from the production of agri-lime.** Fossilised CO_2 is released when quarried lime (applied to soil mainly as CaCO_3) dissolves and makes its way through the calcium cycle. The development of a centralised facility for the production of agricultural lime may justify incorporation of carbon capture technology that,

together with calcination technology, could allow the capture and storage of CO₂ from quarried lime. The integration of a centralised liming facility with the sustainable slurry management technology may enable the regeneration and re-use of lime in the liming step, facilitating the removal and capture of slurry-entrained CO₂ prior to use of the lime in the production of an integrated lime-mineralised slow-release P fertiliser.

- **Energy decarbonisation.** The liming of slurry solids enhances biogas yield when slurry solids are used as a co-digestion feedstock in AD. Incorporation of the sustainable slurry management system into AD operations will significantly reduce the cost of digestate management, which can be a rate-limiting factor in large-scale AD operations. Alternatively, drying slurry solids will facilitate their use as a co-combustion solid fuel to generate renewable heat. The integration of renewable energy to be used for process energy, with excess supplies of biogas or heat sold to energy users within

the stakeholder group, will increase the share of renewable energy in the energy mix and correspondingly reduce energy-related GHG emissions.

- **Agri-food transport decarbonisation.** Transitioning agri-food heavy goods vehicle transport fleets to the use of compressed gas fuels may facilitate the integration of compressed biomethane, which, as a qualifying renewable fuel, would result in a reduction in transport-related GHG emissions.
- **Biodiversity.** It may be possible to designate a land bank adjacent to processing sites, engineered to a standard similar to a constructed wetland, to receive conditioned process liquors and discharge them via transpiration. Given the low impact on the environment expected from conditioned liquors, the site could be maintained as a wildlife refuge, not only offering a method to biopolish process discharges, but also fulfilling a role in promoting biodiversity.

6 Conclusions and Recommendations

The sustainable slurry management concept developed in this project offers a potential route to improve the economic and environmental sustainability of agriculture. Validation of the potential benefits, however, requires further development of the process technology, which to date has been undertaken only with first pilot-scale prototypes. It requires integration of technology and business model concepts in a multi-cycle, end-to-end, commercial-scale demonstration to validate economic and environmental assumptions and to determine acceptance among stakeholder groups.

Additional **technology development** is required to finalise appropriate process protocols, optimise engineering and validate performance assumptions:

- **All technology components.** Development work is required to achieve robust technology readiness level 6/7 datasets and meet commercialisation standards, ensuring compliance with health and safety codes, electrical codes and other CE standards.
- **Filter-aided filtration unit.** Further development work is required to improve throughput while minimising leakage and downtime.
- **Struvite reactor design.** Nucleation and crystallisation reactors were initially designed as continuous stirred tank reactors; however, agitation of struvite-laden media would be more efficient if reactors were designed as upflow reactors, allowing gravity to assist with mixing.
- **NH₃ recovery and productisation.** COVID-19-related testing delays limited investigations into the NH₃ recovery and valorisation processes. Further work is required to test the recovery and valorisation options in the process protocols referenced in this report.
- **Integration of lime calcination and CO₂ recovery.** Developments in calcination designs for integrating CO₂ recovery have the potential to make significant contributions to reductions in agricultural GHG emissions. These processes need to be tested, validated and optimised in a manner that can be integrated into the sustainable slurry management concept.
- **Ash filter aid and lime valorisation.** The fertiliser value of recovered ash filter aid and lime is key to the economic sustainability of the concept. Further work is required to transform these materials into marketable products, which may require some chemical interventions to modify structures and to optimise plant availability, together with performance testing over multi-cycle growing periods.
- **Life cycle analysis.** This will be required once process protocols have been finalised to validate the contribution to environmental sustainability.

This proposed technology development work will inform a more comprehensive assessment of the economic viability of the concept than was possible during this project. The monetisation of sustainability is a challenging concept to understand, and is rarely self-supporting, as the underlying drivers are perceived as societal imperatives rather than commercial. Some form of support framework (either obligatory or incentivised, or both) that underpins technology development and deployment until the sustainability concepts become mainstream (e.g. waste recycling, wind power) is normally required. In any particular market application, there are usually multiple interdependent elements that must be developed coincidentally to underpin a coherent process flow. In some cases, this coincidental development has been publicly funded through large subventions, and developing the supply chain has been left up to stakeholders. Examples such as the Renewable Obligation Certificate scheme in Northern Ireland or the biofuel subsidies that were offered in the early years of biofuel development suggest that publicly funded subventions may be inefficient in terms of cost to the public exchequer. It may be less costly to adapt and leverage existing programmes to fulfil some of the support requirements or, if new support frameworks are required, to establish frameworks such that they can co-exist with existing ones, to create a coherent market structure, and are structured in a way that provides an incentive to move away over time from the public purse as the primary funding mechanism. Elements of the sustainable slurry management

concept that are likely to require **public support mechanisms** are as follows:

- **Feedstock mobilisation.** A mechanism is required that incentivises farmers to supply slurry for sustainable processing, and to replace the lost nutrients by applying low-loss fertilisers. The project team has outlined how the direct farm payment scheme might be adapted to support certified N removal when slurry is removed from farms for sustainable processing, with the redemption mechanism structured to promote the application of low-loss fertilisers.
- **Development of viable routes to market for the ash–lime fertiliser product.** The introduction of the EU's new FPR (European Union, 2019) may help to underpin a potential market for slurry-derived fertiliser. Introduction in Ireland should ensure that it does not inadvertently prevent marketing of the proposed sustainable slurry process outputs. In addition, introducing a redemption mechanism to the “certified N removal” concept that supports the use of process outputs would help to secure routes to market for the filter aid-lime fertiliser.
- **Development of routes to market for energy outputs.** Support grants payable to renewable energy users may be required to incentivise demand for renewable energy supplied under contract by community schemes. Adaptations to the existing Support Scheme for Renewable Heat to facilitate this may result in the increased uptake of renewable heat and/or biogas energy. The continuation of excise tax derogations for the use of compressed biomethane as a transport fuel would underpin a market for gaseous transport fuels, especially as the price of oil rises and carbon taxes are increased.
- **Access to finance.** The sustainable slurry management concept is capital intensive. It is not yet mainstream, nor market proven, and, as such, is unlikely to attract normal commercial finance. Finance will be required to adapt plants and deploy equipment, a portion of which should be sourced from community stakeholders, but the largest proportion is likely to have to come from the public exchequer. The microeconomic illustrations suggest that, provided that market outlets for the energy and nutrient products can be identified, it would be quite possible for the system

to generate an annual cash flow. To minimise the cost to the public purse, the structure requires a specialist financing mechanism that entitles the exchequer to recover its investment.

It was beyond the scope of this project to address the numerous possible **regulatory roadmap** scenarios pertaining to the eventual full-scale application of the system. For example, inputs from EPA regulatory experts suggest that, if a LITE or INTENSE process hub was to be located within the boundary of a licensed landfill site, whether or not the existing landfill site's licence could accommodate this would need to be established. If processing could not be accommodated, the licence would need to be reviewed, with time and cost implications. If the process hub was to be located elsewhere, then the authorisation requirements would most likely be set out in the First Schedule EPA Act 1992, as amended, or in the Third/Fourth Schedule of the Waste Management Act 1996, as amended. Furthermore, for any waste from the sustainable slurry management process to be granted a product status, the waste would need to achieve end-of-waste status. The mode for achieving end-of-waste status will, however, depend on whether the process is included under the scope of the EU FPR, which, at the time of writing, had not been finalised. If the products from the process are not included under the scope of the FPR, then end-of-waste status may be possible to achieve through an application to the EPA under the EU (Waste Directive) Regulations 2011–2020. Finally, wastewaters may be exempt from waste authorisation requirements under the Waste Management Act or may not be eligible for end-of-waste status, as per Regulation 26 of the EU (Waste Directive) Regulations 2011–2020.

A key outstanding element of the project relates to the EU FPR process and what it means with respect to organic fertilisers. The conclusions reached on definitions of “end of process” for a range of ABP materials in the FPR will be critical with respect to the authorisation pathway that will apply to the recovered fertilising compounds produced by the modular treatment system described and the ability to use the CE mark.

The development of the regulatory roadmap(s) will be critical to allow more comprehensive economic and impact assessments to be undertaken, and for

the progression of the proposed system towards full-scale application. We recommend that this work is undertaken when there is clarity on the CE authorisation criteria and it should be carried out alongside further scientific and engineering work to develop the modular treatment system, which will determine the final product output mix and

specifications. An EU-wide project to develop a full-scale demonstrator at a specific location would appear to offer the best opportunity to accelerate the development of the sustainable nutrient recovery concept outlined here, given the scale of activity and range of expertise required, and the potential EU-wide impact of a successful outcome.

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Abbreviations

ABP	Animal by-product
AD	Anaerobic digestion
BOD	Biochemical oxygen demand
CAN	Calcium ammonium nitrate
CAP	Common Agricultural Policy
CMC	Component material category
C:N ratio	Carbon-to-nitrogen ratio
CO_{2-eq}	Carbon dioxide equivalent
COD	Chemical oxygen demand
DAFM	Department of Agriculture, Food and the Marine
DAP	Diammonium phosphate
DM	Dry matter
EBITDA	Earnings before interest, taxation and depreciation
ELV	Emission limit value
FPR	Fertiliser Product Regulation
GHG	Greenhouse gas
HRT	Hydraulic residence time
LULUCF	Land use, land use change and forestry
MAP	Monoammonium phosphate
MWt	megawatts thermal
NAP	Nitrates action programme
N_{eq}	Nitrogen equivalent
PAO	Polyphosphate-accumulating organism
PFC	Product function category
PM	Particulate matter
RO	Reverse osmosis
TAN	Total ammoniacal nitrogen
VS	Volatile solid
WFD	Waste Framework Directive
WWT	Wastewater treatment

An Gníomhaireacht Um Chaomhnú Comhshaoil

Tá an GCC freagrach as an gcomhshaoil a chosaint agus a fheabhsú, mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaoil a chosaint ar thionchar díobhálach na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialáil: Rialáil agus córais chomhlíonta comhshaoil éifeachtacha a chur i bhfeidhm, chun dea-thorthaí comhshaoil a bhaint amach agus díriú orthu siúd nach mbíonn ag cloí leo.

Eolas: Sonraí, eolas agus measúnú ardchaighdeán, spriocdhírthe agus tráthúil a chur ar fáil i leith an chomhshaoil chun bonn eolais a chur faoin gcinnteoireacht.

Abhcóideacht: Ag obair le daoine eile ar son timpeallachta glaine, táirgiúla agus dea-chosanta agus ar son cleachtas inbhuanaithe i dtaobh an chomhshaoil.

I measc ár gcuid freagrachtaí tá:

Ceadúnú

- > Gníomhaíochtaí tionscail, dramhaíola agus stórála peitрил ar scála mór;
- > Sceitheadh fuíolluisce uirbhig;
- > Úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe;
- > Foinsí radaíochta ianúcháin;
- > Astaíochtaí gás ceaptha teasa ó thionscal agus ón eitlíocht trí Scéim an AE um Thrádáil Astaíochtaí.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

- > Iniúchadh agus cigireacht ar shaoráidí a bhfuil ceadúnas acu ón GCC;
- > Cur i bhfeidhm an dea-chleachtais a stiúradh i ngníomhaíochtaí agus i saoráidí rialáilte;
- > Maoirseacht a dhéanamh ar fhreagrachtaí an údaráis áitiúil as cosaint an chomhshaoil;
- > Caighdeán an uisce óil phoiblí a rialáil agus údaruithe um sceitheadh fuíolluisce uirbhig a fhorfheidhmiú
- > Caighdeán an uisce óil phoiblí agus phríobháidigh a mheasúnú agus tuairisciú air;
- > Comhordú a dhéanamh ar líonra d'eagraíochtaí seirbhíse poiblí chun tacú le gníomhú i gcoinne coireachta comhshaoil;
- > An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaoil.

Bainistíocht Dramhaíola agus Ceimiceáin sa Chomhshaoil

- > Rialacháin dramhaíola a chur i bhfeidhm agus a fhorfheidhmiú lena n-áirítear saincheisteanna forfheidhmithe náisiúnta;
- > Staitisticí dramhaíola náisiúnta a ullmhú agus a fhoilsiú chomh maith leis an bPlean Náisiúnta um Bainistíocht Dramhaíola Guaisí;
- > An Clár Náisiúnta um Chosc Dramhaíola a fhorbairt agus a chur i bhfeidhm;
- > Reachtaíocht ar rialú ceimiceán sa timpeallacht a chur i bhfeidhm agus tuairisciú ar an reachtaíocht sin.

Bainistíocht Uisce

- > Plé le struchtúir náisiúnta agus réigiúnacha rialachais agus oibriúcháin chun an Chreat-treoir Uisce a chur i bhfeidhm;
- > Monatóireacht, measúnú agus tuairisciú a dhéanamh ar chaighdeán aibhneacha, lochanna, uiscí idirchreasa agus cósta, uiscí snámha agus screamhuisce chomh maith le tomhas ar leibhéal uisce agus sreabhadh abhann.

Eolaíocht Aeráide & Athrú Aeráide

- > Fardail agus réamh-mheastacháin a fhoilsiú um astaíochtaí gás ceaptha teasa na hÉireann;
- > Rúnaíocht a chur ar fáil don Chomhairle Chomhairleach ar Athrú Aeráide agus tacaíocht a thabhairt don Idirphlé Náisiúnta ar Gníomhú ar son na hAeráide;

- > Tacú le gníomhaíochtaí forbartha Náisiúnta, AE agus NA um Eolaíocht agus Beartas Aeráide.

Monatóireacht & Measúnú ar an gComhshaoil

- > Córais náisiúnta um monatóireacht an chomhshaoil a cheapadh agus a chur i bhfeidhm: teicneolaíocht, bainistíocht sonraí, anailís agus réamhaisnéisiú;
- > Tuairiscí ar Staid Thimpeallacht na hÉireann agus ar Tháscairí a chur ar fáil;
- > Monatóireacht a dhéanamh ar chaighdeán an aeir agus Treoir an AE i leith Aeir Ghlain don Eoraip a chur i bhfeidhm chomh maith leis an gCoinbhinsiún ar Aerthruailliú Fadraoin Trasteorann, agus an Treoir i leith na Teorann Náisiúnta Astaíochtaí;
- > Maoirseacht a dhéanamh ar chur i bhfeidhm na Treorach i leith Torainn Timpeallachta;
- > Measúnú a dhéanamh ar thionchar pleananna agus clár beartaithe ar chomhshaoil na hÉireann.

Taighde agus Forbairt Comhshaoil

- > Comhordú a dhéanamh ar ghníomhaíochtaí taighde comhshaoil agus iad a mhaoiniú chun brú a aithint, bonn eolais a chur faoin mbeartas agus réitigh a chur ar fáil;
- > Comhoibriú le gníomhaíocht náisiúnta agus AE um thaighde comhshaoil.

Cosaint Raideolaíoch

- > Monatóireacht a dhéanamh ar leibhéal radaíochta agus nochtadh an phobail do radaíocht ianúcháin agus do réimsí leictreamaighnéadacha a mheas;
- > Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as tasmí núicléacha;
- > Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta;
- > Sainseirbhísí um chosaint ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Ardú Feasachta agus Faisnéis Inrochtana

- > Tuairisciú, comhairle agus treoir neamhspleách, fianaise-bhunaithe a chur ar fáil don Rialtas, don tionscal agus don phobal ar ábhair maidir le cosaint comhshaoil agus raideolaíoch;
- > An nasc idir sláinte agus folláine, an geilleagar agus timpeallacht ghlan a chur chun cinn;
- > Feasacht comhshaoil a chur chun cinn lena n-áirítear tacú le hiompraíocht um éifeachtúlacht acmhainní agus aistriú aeráide;
- > Tástáil radóin a chur chun cinn i dtithe agus in ionaid oibre agus feabhsúchán a mholadh áit is gá.

Comhpháirtíocht agus Líonrú

- > Oibriú le gníomhaireachtaí idirnáisiúnta agus náisiúnta, údaráis réigiúnacha agus áitiúla, eagraíochtaí neamhrialtais, comhlachtaí ionadaíochta agus ranna rialtais chun cosaint comhshaoil agus raideolaíoch a chur ar fáil, chomh maith le taighde, comhordú agus cinnteoireacht bunaithe ar an eolaíocht.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an GCC á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóir. Déantar an obair ar fud cúig cinn d'Oifigí:

1. An Oifig um Inbhuanaitheacht i leith Cúrsaí Comhshaoil
2. An Oifig Forfheidhmithe i leith Cúrsaí Comhshaoil
3. An Oifig um Fhianaise agus Measúnú
4. An Oifig um Chosaint ar Radaíocht agus Monatóireacht Comhshaoil
5. An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tugann coistí comhairleacha cabhair don Gníomhaireacht agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

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