



# Review of gaseous emissions of methane, nitrous oxide and ammonia, and nitrate leaching to water, from farm dairy effluent storage and application to land

## Final Report

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## Plain-language summary

Agricultural activity causes greenhouse gas (GHG) emissions in a number of ways. In New Zealand (NZ), methane exhaled by farm animals is the greatest source of agricultural GHG; it has proven very difficult to find technologies to reduce these emissions. Urine and dung deposited on the ground by farm animals are also major sources of GHG emissions (nitrous oxide and methane). Where these excreta are deposited at farm structures such as milking platforms, holding yards, feed pads and stand-off pads, they must be collected regularly. Usually, they are returned to the land to make use of the nutrients they contain. The timing of this land application is generally guided by the aim to minimise the leaching of nitrate into waterways. During the storage and treatment before application, GHG emissions will occur. Understanding the processes driving these emissions allows the design of storage and treatment practices that minimise them.

Collection of excreta is usually done by washing them off a hard surface with large amounts of water, creating a dilute manure liquid known as “effluent”. This effluent is either collected into a sump and, within less than a day, directly irrigated onto land, or it is stored in a larger pond for deferred land irrigation sometime later, when weather and paddock conditions are most suitable. The alternative practice of effluent treatment in a pond and subsequent discharge into waterways is increasingly discouraged and practiced less and less on NZ dairy farms. In NZ’s GHG inventory, emissions from farm dairy effluent appear as a minor contribution, constituting the largest proportion of the farm manure management category. Manure management accounts for only 2% of the total GHG emissions from agriculture. However, recent work has raised concerns that methane emissions from effluent stored in ponds are substantially underestimated. Moreover, with increasing intensification of dairy farming, increasing volumes of effluent are handled, so emissions from effluent are likely to increase at a faster rate than overall dairy farming emissions. It is thus important to better understand the magnitude and special characteristics of emissions from effluent management, not just to improve the accuracy of NZ’s GHG inventory, but also to assess the overall potential of practical mitigation technologies for these emissions and their cost-effectiveness.

Firstly, this report briefly summarises the principal processes leading to GHG emission from manure. Next, it describes past and present practices in NZ for the collection, treatment, storage and land-application of farm dairy effluent. Data on effluent pond usage, collected from all regional councils, are then presented. The number of farms with effluent ponds has been rapidly increasing in recent years, surpassing 9,700 in 2011 (80.5% of dairy farms), following promotion of pond use by industry and regional councils as a measure to reduce nitrogen leaching following land application. The average pond volume is about 1700 m<sup>3</sup>, representing about 3 months of effluent storage capacity. The primary sources of dairy effluent are the milking platforms and holding yards, but increasingly, these ponds receive waste from feed pads, stand-off pads, and winter shelters or housing. Along with the growth trend in pond usage goes increased usage of pre-treatment practices, in particular solids separation, but there are little data to accurately quantify the latter trend.

In the central sections, the national and international literature is reviewed, on gaseous emissions of methane, nitrous oxide and ammonia, along the treatment chain for effluent. (Ammonia is not a GHG, but a precursor of subsequent additional, “indirect”, nitrous oxide emissions, and as such included in GHG inventories.) There is no doubt that the methane emissions from ponds are the largest contributor to the total GHG emissions from effluent. This is because the solid matter at the bottom of such ponds is an anaerobic (oxygen-free) environment, in which methane-producing microorganisms thrive. Nitrous oxide emissions

from anaerobic ponds are negligible, while ammonia emissions vary widely between different studies, probably because these emissions depend strongly on pH (acidity) and manure composition. The second-largest contribution to GHG emissions from effluent are the nitrous oxide emissions from land application, but these occur from all animal manure, regardless of whether the manure is collected and treated as effluent or deposited directly by excreting animals. Ammonia emissions from land application of effluent were found to be less than those reported overseas from the application of slurries, which are less dilute. Methane emissions from land application are negligible. No data were found to document GHG emissions at the sources of excreta deposition (milking sheds, yards, pads, winter housing). There are also no relevant data at present to estimate methane emissions from effluent pre-treatment systems in use in NZ.

The literature on nitrate leaching is also briefly reviewed. Leaching occurs only from the last step of the effluent treatment chain, the land application. While there are numerous studies investigating nitrogen leaching from fertilisers, only three studies were found specifically for effluent in NZ. These report the leaching losses as 1.6 to 4.7% of the applied nitrogen.

The findings are then discussed with a critical view into the accounting methodology used in NZ's GHG inventory. The current inventory equation for methane emissions from ponds is flawed, and a corrected equation is suggested in this report. Both the flawed and the corrected equation are based on the amount of manure handled in ponds. Since the fraction of manure entering effluent ponds is not very accurately known, it is recommended that data are collected to obtain more accurate estimates. Ideally, this should include estimates of the amount of manure that originates from other sources than the milking shed and yards (i.e. feed pads, stand-off pads and winter housing), as these sources are currently not included in the inventory. The likely result of correcting the inventory equation and including all manure sources would be 3- to 5-times higher methane emissions from dairy effluent storage than the figure currently reported. This means that the emissions from collected manure (to which dairy farming contributes the largest proportion, ahead of pig and poultry farming) are more likely to represent about 4 to 7%, rather than 2%, of the total GHG emissions from agriculture.

Neither past nor current IPCC guidelines for national GHG inventories require inclusion of emissions caused directly at the source of manure deposition, or by pre-treatment technologies. Past guidelines also did not consider ammonia volatilisation from effluent ponds, but these will need to be included in the future. Thus, it is suggested that research be undertaken to understand these emission processes and provide the necessary input data, so that emissions from these treatment steps can be accounted for in the future. Data are required in particular to quantify emissions of methane, nitrous oxide and ammonia from solid separator systems and to quantify ammonia emissions from milking parlours and ponds.

Given that effluent pond emissions are currently underestimated, and are more amenable to reduction measures than methane emissions directly from animals, it is concluded that the potential for actual reduction of agricultural GHG emissions in NZ by applying suitable measures at effluent ponds is larger than previously assumed. Such measures include the covering of effluent ponds, with or without energy production from burning the captured "biogas", the use of methane-consuming bio-filters, and certain measures to keep stored effluent aerobic. These measures are easy to implement in principle. It should be researched which of them are also economical, and policy should be developed to promote their uptake nationwide.

# Executive summary

## Background

Atmospheric emissions arising from the management of dairy farm manure are generally poorly understood and quantified. New Zealand's greenhouse gas (GHG) inventory lists emissions from farm dairy effluent, predominantly in the form of methane, as a minor contribution to total dairy farm emissions. However, recent work indicates that for some emission pathways, like methane from effluent stored in ponds, emissions are substantially underestimated. Moreover, with increasing intensification of dairy farming and the handling of larger volumes of manure, dairy farm manure emissions are likely to increase at a disproportionately larger rate, compared with overall dairy farming emissions. It is therefore imperative to better understand the magnitude and special characteristics of emissions from dairy farm manure management. Furthermore, a number of practical mitigations technologies for these emissions exist; hence, a better understanding of these emissions could enable some moderate, but cost-effective GHG emission reductions in the dairy sector.

## Methods and structure of this report

In this report, after a brief overview of the principal processes causing GHG emissions from manure, the past and present practice of manure management on dairy farms in New Zealand (NZ) are described. This includes the collection, treatment, storage and land-application of farm dairy effluent. Data collected from all regional councils are presented, on the numbers of effluent ponds in existence, their storage capacity, and expected future trends in effluent storage practice.

Then, the national and international literature is reviewed, on gaseous emissions of methane ( $\text{CH}_4$ ), nitrous oxide ( $\text{N}_2\text{O}$ ) and ammonia ( $\text{NH}_3$ ) from managed dairy farm manure, as well as on nitrate leaching from manure application to pasture. In the NZ context, the review focuses strictly on studies that deal with emissions from farm dairy effluent as it is treated, stored and applied to land. Internationally, storage and irrigation of dilute effluent is not a common practice, so most international studies are concerned with more concentrated forms of manure (solids and slurries). Key overseas findings from such studies are included in this report but care is taken to assess their relevance for NZ farming practices.

Next, the findings are discussed with a view to suggesting potential improvements to the accounting methodology for NZ's GHG inventory. This includes scenario calculations to illustrate the likely effects of the current trend of increasing manure volumes handled in effluent treatment systems. Potential approaches for emissions mitigation are also briefly considered. Finally, current knowledge gaps regarding the emissions from farm dairy effluent in NZ are identified, and certain areas for future research are recommended.

## Results – manure management practice in NZ

The primary sources of dairy effluent are the milking platforms and holding yards, where the manure produced by the cows is washed off with water and collected, either for short-term storage in sumps and rapid land application, or in larger ponds for anaerobic treatment and deferred land application. The number of farms with ponds surpassed 9,700 in 2011 (80.5% of dairy farms), following promotion of storage pond use by industry and regional councils as a means to reduce nitrogen leaching from land application of the effluent. There are considerable differences between regions with respect to the proportion of farms that use ponds; however, if the growth trend for pond usage continues, these regional differences will shrink. The average pond volume is about  $1700 \text{ m}^3$ , representing about 3 months of effluent

storage capacity. In addition to the effluent from milking platforms, these ponds increasingly receive waste from feed pads, stand-off pads, and winter shelters or housing. Also increasing are pre-treatment practices, in particular solids separation and weeping walls, but statistics to quantify this trend are lacking at present.

## Results – literature review

For a complete accounting of GHG emissions from manure management, these would have to be quantified for the whole treatment chain, i.e. emissions at source (milking sheds, yards, pads, winter housing), emissions during pre-treatment, emissions from ponds, and emissions from land application.

No data documenting GHG emissions at source were found. There are also no relevant data at present to estimate CH<sub>4</sub> emissions from effluent pre-treatment systems in use in NZ.

Emissions of N<sub>2</sub>O from pre-treatment manure solids were found to range from 0.1 to 4.8% of total N, across 12 studies including one from NZ. Three studies were found that reported emissions of NH<sub>3</sub> from pre-treatment manure solids, ranging from 12 to 21% of total ammoniacal nitrogen.

Based on three year-long NZ studies, CH<sub>4</sub> emissions from anaerobic effluent ponds are about 0.20 m<sup>3</sup> CH<sub>4</sub> kg<sup>-1</sup> VS (where VS = volatile solids). The reviewed data suggest that N<sub>2</sub>O emissions from liquids and slurries with low dry matter (DM) contents are 0 to 0.1% of total nitrogen (eight relevant studies including one from NZ). The emissions of NH<sub>3</sub> from liquids and slurries reported in the literature are substantial and highly variable, at 1 to 65% of total ammoniacal nitrogen (six studies).

No direct measurements of CH<sub>4</sub> emissions exist from the land application of effluent on NZ farms. Overseas studies on the effects of slurry spreading indicate that such emissions are negligible. For N<sub>2</sub>O, the emission factors found in four NZ studies span from 0.01 to 4.9% of applied nitrogen. This range includes that reported overseas for N<sub>2</sub>O emissions from land-applied slurries, of < 0.1 to 3%. The NH<sub>3</sub> emissions from land application of effluent were found to be 0.2 to 1.2% of total ammoniacal nitrogen in three NZ studies. This is less than the 1.5 to 40% reported internationally for emissions from slurry spreading.

Nitrate leaching occurs only from the last step of the effluent treatment chain, the land application. In three NZ studies, this was quantified as 1.6 to 4.7% of the applied nitrogen.

## Conclusions and recommendations

The CH<sub>4</sub> emissions from ponds are the largest contributor to the total GHG emissions from effluent management in NZ. Crucially, the current inventory equation for these CH<sub>4</sub> emissions is flawed, and it is likely that as a consequence emissions are underestimated. We recommend replacing this equation with a modified version of the IPCC 2006 Tier 2 methodology, based on the amount of manure handled in ponds, on a monthly time basis.

The fraction of manure entering effluent ponds is not accurately known, as it relies on estimates of the time fraction that the cows spend at the milking shed and yards. We recommend that a method be developed to collect data that will improve the accuracy of these estimates. With the equation suggested by us and the assumption that, NZ-wide, 11% of all dairy manure produced during a 9-month long milking season is collected in FDE ponds, CH<sub>4</sub> emissions from ponds are obtained that are about twice as large as currently reported in the inventory.



At present, the amount of manure collected in ponds that originates from other sources than the milking shed and yards (i.e. feed pads, stand-off pads and winter housing) is not included in the inventory. The adapted IPCC Tier 2 equation suggested by us would allow for inclusion of these additional manure sources. The likely result of correcting the inventory equation and including all manure sources would be 3- to 5-times higher CH<sub>4</sub> emissions from manure management systems than currently reported. This means that manure management would account for about 11 to 17%, rather than 4%, of the total CH<sub>4</sub> emissions from dairy farms. Consequently, the manure management category would represent 4 to 7%, rather than 2%, of the total GHG emissions from the agricultural sector.

Given that pond emissions are currently underestimated, and are much more amenable to mitigation than enteric emissions, the potential for actual reduction of agricultural emissions in NZ by applying suitable measures at effluent ponds is larger than previously assumed. These mitigation options include the covering of effluent ponds (which allows for “biogas” collection and subsequent combustion, with or without energy recovery), the use of CH<sub>4</sub>-consuming bio-filters, and certain measures to keep stored effluent aerobic. These measures are easy to implement and thus possibly more cost-effective than approaches to reduce CH<sub>4</sub> eructated by animals or GHG emissions from excreta on pasture. It should be researched which mitigation options are economical, and policy should be developed to promote their uptake nationwide.

In compliance with the IPCC 1996 guidelines, the current GHG inventory does not accurately reflect the field situation in that it does not include any emissions caused directly at source or by pre-treatment technologies; nor does it account for NH<sub>3</sub> volatilisation losses from effluent ponds. The latter are included in the IPCC 2006 guidelines, which are to be used from 2015 onwards, so research should be undertaken to specify the activity data and emission factors required to estimate NH<sub>3</sub> emissions from effluent ponds. Going beyond inventory requirements, research is recommended to quantify emissions of CH<sub>4</sub>, N<sub>2</sub>O and NH<sub>3</sub> from solid separator systems (where “weeping walls” are the most popular type) and to quantify NH<sub>3</sub> emissions from milking parlours. There is also a need to collect data on how widespread the use of pre-treatment technologies is.

To allow for a complete yet simple accounting of indirect N<sub>2</sub>O emissions, it is recommended that a sub-inventory for NH<sub>3</sub> emissions is developed from all emission pathways combined, and to apply the conversion factor to N<sub>2</sub>O (1%) only once, to the total of the NH<sub>3</sub> emissions, rather than separately for each pathway.

Finally, in order to be able to design strategies that minimise emissions following the application of effluent to land, the soil and weather factors and processes controlling the emissions of N<sub>2</sub>O and NH<sub>3</sub> must be better understood than they are at present.

# 1 Introduction

New Zealand's dairy industry has grown rapidly over the past three decades, with the number of lactating dairy cows increasing from 2.03 million in the 1980/81 season to 4.78 million in 2012/13 (DairyNZ 2013). Today, it is the country's single biggest earner of export revenue. In 2012, the production of  $1.664 \times 10^9$  kg of milk solids generated  $13.9 \times 10^9$  dollars of export revenue (Austin et al. 2012). The intensification and expansion of NZ's dairy sector have been accompanied by increasing environmental side effects on the country's water, soil and atmospheric resources (Bisley 2010; MfE 2012). Some side effects, such as dairy farming impacts on ground and surface water resources, are being actively combated through voluntary measures, such as the Clean Streams Accord (Hobbs et al. 2003), as well as more stringent regulation such as prescribed effluent management and storage practices (IPENZ 2013) and land use restrictions imposed by various regional councils (Clarke et al. 2003; Horizons Regional Council 2010).

In comparison, dairy farming effects on atmospheric resources, such as emission of greenhouse gases (GHG), in particular methane ( $\text{CH}_4$ ) and nitrous oxide ( $\text{N}_2\text{O}$ ), emissions of ammonia ( $\text{NH}_3$ ) and odour emissions, have all increased with little mitigation action taken. This is mainly the result of two factors:

- The elusive, variable, diffuse or multi-point nature of these emissions, making them hard to quantify.
- The lack of practical and cost-effective abatement methods and technologies (in particular for GHG emissions), that can be applied at large scale in the NZ dairy farming context.

New Zealand actively pursues research into agricultural and other GHG emissions (MfE 2013; PGGRC 2013), including enteric  $\text{CH}_4$  emissions from ruminants, which is the single biggest contribution to NZ's overall GHG emission profile (~32.3% of total GHG, ~68.4% of agricultural GHG) according to the national inventory (MfE 2013). Despite the excellent work being carried out, field-scale reductions of enteric emissions may still be years or decades away (Eckard et al. 2010). In the absence of near-term reductions to enteric  $\text{CH}_4$  emissions, better understanding and the promotion of abatement technologies for the smaller emission categories from dairy farming become all the more important. These smaller emission categories include emissions from dairy farming soils, fossil fuel use in dairy farming (reported in the energy sector of the inventory) and, the topic of this report, emissions from manure management on dairy farms.

It is now a decade ago that Saggar et al. (2004) reviewed atmospheric emissions of  $\text{CH}_4$ ,  $\text{NH}_3$  and  $\text{N}_2\text{O}$  from agricultural manure management, including those from excreta deposited directly on pasture as well as those from actively-managed manure. These authors put strong emphasis on describing the processes controlling these emissions. Despite reasonable qualitative understanding of these processes, the emissions remain, by and large, poorly quantified. This is particularly true for the emissions from the collection and storage of farm dairy effluent (FDE) in ponds. New Zealand's GHG inventory estimates these emissions (predominantly occurring as  $\text{CH}_4$ ) as a minor contribution to GHG emissions from dairy farming, of order 4%:  $\text{CH}_4$  emissions from FDE ponds are reported as 419 Gg  $\text{CO}_2\text{-e}$  for 2011, while enteric  $\text{CH}_4$  emissions from the national dairy herd are reported as 10,382 Gg  $\text{CO}_2\text{-e}$  (MfE 2013). However, recent work (Craggs et al. 2008; Pratt et al. 2012; Chung et al. 2013) indicates that the  $\text{CH}_4$  emissions from ponds are probably severely underestimated. This is of concern not only because of the resulting inaccuracy of the

inventory, but also because, as a consequence, CH<sub>4</sub> emissions from ponds may be given too low priority when policies aiming at emissions mitigation are developed. With increasing intensification of dairy farming and the handling of larger volumes of manure, the emissions from manure are likely to increase at a disproportionately larger rate, compared to overall dairy farming emissions (Chung et al. 2013). It is thus imperative to better understand their magnitude and special characteristics. Since a range of practical mitigations technologies for dairy-farm manure management emissions exist (Shilton et al. 2009; Heubeck and Craggs 2010), such better understanding could enable some moderate, but cost-effective atmospheric emission reductions in the dairy sector.

This report collates current knowledge of GHG emissions from dairy-farm manure management, namely the emissions of CH<sub>4</sub>, N<sub>2</sub>O, as well as those of NH<sub>3</sub> (as the most important N<sub>2</sub>O precursor upon re-deposition). To achieve this, the report includes information on manure management practices held by regional councils, and it reviews the national and international literature on dairy farm manure management emissions. Key overseas findings are interpreted in the NZ context, which affects the relevance of reported data. The two key practices of manure management are effluent storage and effluent application to land, which are described in separate sections. The sections on land application include a review of the literature on nitrate (NO<sub>3</sub><sup>-</sup>) leaching, in order to arrive at recommendations that consider environmental impacts more generally, not just GHG emissions. Based on the review's findings, current knowledge gaps are identified, and potential improvements to the accounting methodology for NZ's GHG inventory are considered.

## 2 Processes causing emissions from effluent

### 2.1 CH<sub>4</sub>-generating processes

Methane is produced via the decomposition of organic matter under anaerobic conditions. Organic compounds are degraded by acid-producing bacteria to simpler organic compounds such as volatile fatty acids. After this initial degradation the resulting compounds are utilised by CH<sub>4</sub>-producing archaea (methanogens) under anaerobic conditions (Hellmann et al. 1997). At least four functional groups of microbes are required to perform the necessary chain of chemical reactions from the original long-chained bio-molecules in the substrate matter to CH<sub>4</sub> as the final, volatile, product (Le Mer and Roger 2001), and various anaerobic “ecosystems” containing such groups of microbes have evolved. Some of these live in animal dung, and their composition varies between animal species and dung characteristics, with the consequence that manures from different animals can differ in their CH<sub>4</sub> production capacity. Furthermore, the rate of CH<sub>4</sub> production increases with increasing temperature, and numerous other physical, chemical and biological factors will influence overall methane productivity and rate of formation. The main factor, though, is the amount of anaerobically biodegradable organic matter in the substrate (Saggar et al. 2004), commonly quantified as the biochemical oxygen demand (BOD). Where oxygen is available within the organic substrate, CH<sub>4</sub> emissions will be suppressed. The sites of CH<sub>4</sub> production are therefore generally not near the interface of the substrate with the atmosphere. In order to get into the atmosphere, the CH<sub>4</sub> gas needs to travel through the liquid or solid porous material in which it is produced, either by molecular diffusion or by ebullition, which is observable as the release of bubbles at the upper surface of the material. In the first study of CH<sub>4</sub> emissions from an effluent pond in NZ, McGrath and Mason (2004) recorded such bubbles in order to quantify the emissions but ignored the fraction emitted via molecular diffusion. Other studies since have shown that both fractions are significant, e.g. Park et al. (2010) for a manure storage tank and DelSontro et al. (2010) for a hydro lake (where the CH<sub>4</sub> is generated in the lake sediments, due to decomposition of the inundated biomass).

### 2.2 NH<sub>3</sub>-generating processes

Volatilisation is the evaporation of a volatile substance, in this case NH<sub>3</sub>. Both urine and dung excretion give rise to NH<sub>3</sub> volatilisation, by a combination of chemical and physical processes (no microbial activity is required). The essential steps involved are as follows. Firstly, there must be a source of ammoniacal nitrogen (N), i.e. either NH<sub>3</sub> itself, or ammonium (NH<sub>4</sub><sup>+</sup>). Next, there needs to be an elevation of pH in the liquid phase, which shifts the equilibrium between NH<sub>4</sub><sup>+</sup> and NH<sub>3</sub> to favour the latter (Emerson et al. 1975), and finally a volatilisation surface is required, where NH<sub>3</sub> in aqueous solution will be exchanged with NH<sub>3</sub> in the gas phase, as stated by Henry’s Law (Hales and Drewes 1979). Both equilibrium processes depend exponentially on temperature, in such a way that an increase in temperature strongly increases the volatilisation rate. Where the volatilisation surface is not in direct contact with the atmosphere (i.e. is not the top surface of liquid body), a fourth step required for emission to the atmosphere is the diffusion of the gaseous NH<sub>3</sub> through a porous medium (e.g. soil or dung).

In the case of urine, the elevation in pH and the formation of NH<sub>4</sub><sup>+</sup> is a direct consequence of the hydrolysis of the urea contained in the urine (Sherlock and Goh 1985). At summer temperatures, hydrolysis tends to be near-complete within a few hours, leading to rapid pH rise and high volatilisation rate in the first couple of days. The same is true for surface-applied slurry (Spirig et al. 2010), where the mixing of urea in the urine with urease enzyme

contained in faecal materials leads to rapid urea hydrolysis (Monteny and Erisman 1998) – so rapid that the urea hydrolysis is often already completed at the time of application. As volatilisation proceeds, a subsequent reduction in surface pH occurs as a consequence of the chemical transformation of  $\text{NH}_4^+$  to  $\text{NH}_3$  with the accompanying release of a proton into the soil solution. This re-acidifies the soil surface and after some time a surface soil pH arises that is inadequate to sustain further  $\text{NH}_3$  volatilisation (Sherlock and Goh 1985). In the case of dung pats, the chemistry leading to  $\text{NH}_3$  volatilisation is different (Laubach et al. 2013b), since dung does not contain any urea. However, where dung enters a manure management system it gets inevitably mixed with urine, which brings the necessary ingredients for urea hydrolysis into contact with each other.

The amount and rate of  $\text{NH}_3$  volatilisation from manure is thus determined by the total ammoniacal nitrogen (TAN) content (i.e.  $\text{NH}_4^+$  plus  $\text{NH}_3$ ), temperature, moisture content, and the pH of the excreta as well as the exposed excreta surface area and air movement across the source surface (Hartung and Phillips 1994; Sommer and Hutchings 1995). Volatilisation from soils is determined by the same factors, as well as by soil texture and cation exchange capacity.

## 2.3 $\text{N}_2\text{O}$ -generating processes

In soils, or other substrates with microbial populations, such as manure,  $\text{N}_2\text{O}$  is produced by either nitrification or denitrification.

Nitrification is the microbially mediated conversion of  $\text{NH}_4^+$  to  $\text{NO}_3^-$  that occurs under aerobic conditions. Although not the main product of this process,  $\text{N}_2\text{O}$  can be produced during nitrification.  $\text{N}_2\text{O}$  production during nitrification is dependent upon the biota present and environmental conditions. In soils, the ratio of  $\text{N}_2\text{O}/\text{NO}_3^-$  produced during nitrification increases with increased soil moisture but decreases under higher temperatures (Sitaula and Bakken 1993). Soil pH is also thought to influence the production of  $\text{N}_2\text{O}$  during nitrification (Parton et al. 1996; Bakken et al. 2012). The  $\text{NH}_4^+$  required to begin the nitrification chain may be of biological origin, as a waste product of microbial and animal metabolisms, with the major source for pasture soils being the urine of grazing animals (see previous section). Alternatively, it may have been added by fertilisation, either anthropogenically (as agricultural practice) or naturally, via deposition of atmospheric  $\text{NH}_3$ , which was previously volatilised elsewhere. Because of this process of re-deposition, emissions of  $\text{NH}_3$  (which is not a greenhouse gas) can indirectly lead to the emission of  $\text{N}_2\text{O}$  and must therefore be included in GHG inventories.

Denitrification is the microbially mediated conversion of  $\text{NO}_3^-$  to  $\text{N}_2$ . It is an anoxic process that produces  $\text{N}_2\text{O}$  in an intermediate step. The ratio of  $\text{N}_2\text{O}/\text{N}_2$  produced during denitrification is affected by the amount of  $\text{NO}_3^-$  present, the level of anoxia, pH, temperature, organic matter availability and microbial populations (Blackmer and Bremner 1978; Firestone et al. 1980; Firestone and Davidson 1989; Weier et al. 1993; van Cleemput 1998; Bakken et al. 2012; Peterson et al. 2013; Wang et al. 2013).

Saggar et al. (2004) reviewed experimental results on the  $\text{N}_2\text{O}$  emissions from excretal deposition as well as the spreading of effluent and slurry. Both the large number of influencing factors and the existence of two opposing reaction chains make the prediction of  $\text{N}_2\text{O}$  emissions a very challenging problem. Not only can  $\text{N}_2\text{O}$  be produced by microbial populations, it can also be consumed by three different processes (Chapuis-Lardy et al. 2007), which adds further complexity.

## 2.4 $\text{NO}_3^-$ -generating processes

Nitrate is the final product of the nitrification chain of reactions. Generally, the more aerobic the substrate, the more will the balance between nitrifying and denitrifying reactions be in favour of the former. The more nitrogen is made available in a substrate (e.g. soil or manure, the more  $\text{NO}_3^-$  will be produced in aerobic conditions. The  $\text{NO}_3^-$  ion is highly soluble and thus readily transported with water, which means it can be leached out of the upper soil layers into groundwater and surface runoff, from where it flows into downstream water bodies and accumulates there, leading to eutrophication. Nitrate leaching has therefore become a major undesirable side effect of the intensification of agriculture. It is considered in this review firstly because of the intimate link between the generation of  $\text{N}_2\text{O}$  and  $\text{NO}_3^-$ , and secondly because practices to reduce  $\text{NO}_3^-$  leaching, which are well-covered by best-practice guidelines in the agricultural industry already (Dairy Insight et al. 2007a, b), may sometimes be incompatible with the goal of minimising GHG emissions.

### 3 Effluent management practices in dairy farming

New Zealand's dairy industry has grown rapidly over the past two decades, with the number of dairy cows increasing from  $3.4 \times 10^6$  head in 1990 to  $5.9 \times 10^6$  head in 2010 (MfE 2013). This increase has led to increased volumes of cattle dung and urine. In a typical NZ grazing system, the majority of the excreta from dairy cows is deposited onto pastures. Only the smaller part is actively managed, most commonly the part deposited in and around the milking shed (Chung et al. 2013; MfE 2013). Feed pads and stand-off pads, increasingly used in NZ as dairying intensifies, are also areas where deposited excreta are actively managed (Chung et al. 2013). Actively managed excreta are generally washed from the respective areas of deposition. The use of wash water during this process, as well as the high annual rainfall received onto the generally uncovered areas, creates farm dairy effluent (FDE), a rather dilute liquid (in the UK also known as "dirty water"). Typical dry matter (DM) concentrations in FDE, expressed either as total solids (TS) or as volatile solids (VS), are between 0.5% and 1.0% (Craggs et al. 2003; Pratt et al. 2012). Houlbrooke et al. (2011) define excreta-containing substrates with less than 5% DM as FDE, with 5 to 15% DM as slurry, and with more than 15% DM as solid manure. The following subsections describe NZ's common effluent management systems in more detail.

#### 3.1 Types of effluent management systems

Effluent management on dairy farms in NZ has undergone several changes in the last few decades. The systems are discussed below in the order in which they were used as a "best practice" technology. Some systems have been modified to remain compliant as regulations have become more stringent. Currently effluent management systems are becoming more diverse and region-specific. This will complicate future industry-wide assessments of environmental impacts such as the national GHG inventory, and country-wide corrective action or improvements will be more difficult to achieve.

##### 3.1.1 Two-pond systems for discharge to water

During the 1970's, the standard effluent management system on a NZ dairy farm consisted of a moderately deep (2 to 3 m) primary pond and a shallower (~1 m depth) secondary pond (Figure 1). In the first pond, organic matter in the effluent is digested anaerobically, while larger particles fall out of suspension forming sludge at the bottom of the pond. The second pond is shallower and has a larger surface area compared with the first pond, allowing an aerobic upper layer (thus often called "oxidation pond") and an anaerobic lower layer to co-exist. This treatment system is efficient at removing biological oxygen demand (BOD), but high concentrations of nutrients are still present after treatment (Longhurst et al. 2000). The oxidation ponds were sized with an areal organic loading rate of 84 kg BOD<sub>5</sub>/ha/day. Final discharge from the oxidation pond was to a waterway. These systems had low treatment performance and consistency (Sukias et al. 2001), particularly for removal of ammoniacal nitrogen, leading to eutrophication of water bodies and loss of fertiliser nutrient resources (Houlbrooke et al. 2004). To reduce this harmful impact on the environment, regulations now require FDE to be applied to land rather than discharged into waterways. In compliance with this, two-pond discharge systems have been upgraded in most regions of NZ to irrigate treated FDE directly to land (sometimes only for part of the year), or to use some of the pond volume, generally the volume of the secondary "oxidation" pond, for effluent storage. On some farms, the traditional treatment pond systems have been extended such that the effluent passes through two or more ponds for removal of pathogens, organics and nutrients before being irrigated to pasture.



Figure 1: Farm in the Manawatu with a classic two-pond treatment system. (Photo: C. Pratt)

### 3.1.2 Direct land irrigation

Due to the shortcomings of the two-pond systems with discharge to water, from the mid-1990's onwards new FDE management systems, or major upgrades of existing ones, were generally based on direct land irrigation using travelling irrigators supplied from a small pump sump located at the cow shed. In such systems, daily irrigation is necessary, since they do not include any buffer storage volume, other than the sump itself with a maximum capacity of 1 to 2 days' storage. Effluent irrigation enables recycling of nutrients back to the land and mostly prevents the generation of CH<sub>4</sub> and odour emissions, since the effluent is usually kept aerobic. However, these systems are limited at times of wet weather when soils are saturated, or by failure of irrigation equipment. Either can result in run-off of raw FDE from the pasture into waterways. Furthermore, daily irrigation can lead to leaching of effluent nutrients out of the pasture root zone and into groundwater.

### 3.1.3 Deferred effluent irrigation including storage

To overcome the shortcomings of the direct land-irrigation approach, deferred effluent irrigation systems are now promoted by most regional councils and industry organisations (Dairy Insight et al. 2007a, b; IPENZ 2013). Such systems consist of a storage pond providing storage capacity depending on local climate, soil and farm conditions, and subsequent land irrigation of FDE, at a time when soil and weather conditions are suitable. On some farms, previously existing two-pond systems have been modified to provide storage capacity for deferred effluent irrigation. Either the shallow first pond, or a barrier ditch, contains the bulk of the solid effluent fraction, while the liquid fraction is discharged into a secondary pond allowing for extra storage capacity (Figure 2). Deferred irrigation systems successfully mitigate many of the adverse effects of FDE on surface water resources and increase flexibility of farm management, but have led to other problems. These include:

- Effluent storage ponds are new point sources of odour and CH<sub>4</sub> emissions where they replace direct land irrigation systems.
- Management costs of solids accumulating in storage ponds are high, particular for plastic-lined, shallow ponds.



- Low-rate application irrigation equipment is increasingly promoted to provide more even application, reduce effluent storage requirements and better use effluent nutrients; however such equipment often blocks unless the stored effluent is pre-treated using a solids separation device.

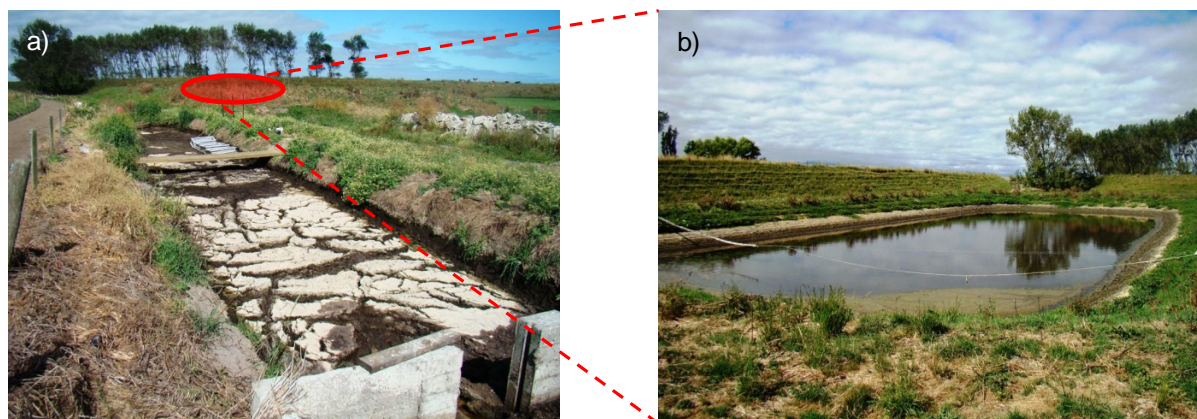


Figure 2: Farm in the Manawatu with a) barrier ditch pond, where effluent from the milking shed enters through the grey wall in the foreground and can be pumped to a secondary pond (red oval); b) the main effluent storage pond. (Photos: C. Pratt)

### 3.1.4 Deferred effluent irrigation including storage and pre-treatment

In order to overcome the problems listed in the previous section, effluent storage pre-treatment is becoming more and more common on NZ's dairy farms. The most relevant pre-treatment technologies include mechanical solids separators, weeping walls, and covered anaerobic ponds (Figure 3).

Various types of solids separators, such as screw press or static screen run-down separators, are used in NZ. The solids removal rate generally varies between 20% and 40% TS, depending on cow diet, season and technology. The heaps of separated solids are usually dryer (>25% DM) for screw-press-type separators than for static run-down screens, but solid heaps from all separator systems become biologically active quickly and start a (partial) composting process. The biological processes in solids separator heaps are generally uncontrolled (no aeration, no watering, continuous addition of fresh substrate), indicating that aerobic, anoxic and anaerobic decomposition processes are likely to occur in solids separator heaps simultaneously as well as successively. This leads to solids separator heaps being able to emit CH<sub>4</sub> (from anaerobic decomposition), N<sub>2</sub>O (from alternating decomposition conditions), and NH<sub>3</sub>, all at widely varying rates (regarding timing and volumes).

Solids from separator heaps are generally land-applied with compost spreaders, a task often carried out by agricultural contractors. The time that the solids are stored can therefore vary greatly from farm to farm, depending on storage capacity, the annual pasture or crop growth cycle, contractor availability and weather. In general solids are applied to land 2 to 3 times a year, indicating a storage time of no more than 4 to 6 months. Solids separator heaps are not known to be major sources of odour.



Figure 3: Typical effluent pre-treatment systems in NZ, employed prior to deferred-irrigation storage ponds : a) mechanical screw press separator, b) weeping-wall system, c) covered anaerobic pond. (Photos a, c): S. Heubeck; b): John Scandrett, Dairy Green.)

Weeping-wall systems are becoming an increasingly popular FDE pre-treatment step, particularly in the South Island. Designs of weeping-wall systems vary greatly throughout NZ, but generally they consist of an earthen pit (lined or unlined) with a permeable (perforated) timber or plastic wall forming one embankment side. Raw effluent builds up behind the permeable wall, and while coarse solids settle or are filtered out behind the wall, the bulk of the effluent liquid “weeps” through the wall and is then either pumped or drained into the effluent storage pond. Solids removal rates of around 50% are often achieved. Solids accumulating behind the barrier are generally removed once or twice per year and applied to pasture or crops. Removal and handling of solids from weeping-wall systems is equally diverse, with pumps, slurry tankers, diggers (buckets) and compost spreaders all being used depending how wet or dry the accumulated solids are. The physical and biological conditions, and as a consequence the level of solids degradation and emissions, from effluent solids accumulating within a weeping-wall system can vary widely. Some lightly loaded systems may behave like anaerobic ponds (with a crust), while conditions in other weeping-wall systems may be more comparable to those in overseas slurry storage tanks. Rates and types of solids degrading in weeping-wall systems and volumes and types of emissions will vary accordingly. The release of  $\text{CH}_4$  (indicating anaerobic conditions) and  $\text{N}_2\text{O}$  (indicating alternating conditions) as well as  $\text{NH}_3$  are all possible. It is unknown to what degree weeping-wall systems cause odour emissions, but highly loaded systems with an acidic pH level could potentially be odour sources.

Covered anaerobic ponds offer a further FDE pre-treatment option that is as yet not widely adopted in NZ. Covered anaerobic ponds can remove more than 70% of raw effluent solids (Heubeck and Craggs 2010), and are therefore a more efficient solids removal technology than both weeping-wall and mechanical solids separator systems. Both dissolved and settleable effluent solids are anaerobically degraded in covered anaerobic ponds, which requires a minimum hydraulic residence time of 20 – 30 days. As such these systems require a larger treatment volume and footprint than a weeping wall. However, the intense anaerobic degradation occurring in covered anaerobic ponds means that only a small volume of inert solids needs to be removed every 1 – 3 years, which is usually conducted with slurry tankers. The  $\text{CH}_4$  emissions produced during anaerobic degradation are captured with an impermeable plastic cover and combusted, either in a flare or for energy recovery. Therefore no  $\text{CH}_4$  is emitted to the atmosphere. The system also does not produce any  $\text{N}_2\text{O}$  emissions, since covered anaerobic ponds are completely oxygen-free environments. The impermeable plastic

cover also minimises  $\text{NH}_3$  emissions; however residual amounts of  $\text{NH}_3$  gas contained in the biogas are combusted to  $\text{NO}_x$ , if they are not filtered out. The impermeable plastic cover also prevents the release of odour emissions from a covered anaerobic pond. One advantage of covered anaerobic ponds compared with solids separators and weeping-wall systems is that they reduce both the coarse and the dissolved solids content of FDE. This means that covered anaerobic ponds eliminate the odour and GHG impact of both the solids that are removed within the pond and the effluent that is subsequently stored in the storage pond. Covered anaerobic ponds can therefore be seen as a comprehensive odour and GHG mitigation tool.

To conclude, the type of pond and treatment system used on a farm will control which fractions of the manure will be stored in which conditions and for which length of time, which will greatly affect the rate and extent of GHG emissions.

### 3.2 National and regional statistics of effluent management systems

Information on the type of ponds used on NZ dairy farms, as well as average pond volumes and storage capacities was sought from regional councils, DairyNZ, the Fertiliser Association and Fonterra. The results are summarised in Table 1. It should be noted that there is a considerable degree of uncertainty regarding the data in Table 1, as many councils and industry organisations do not keep detailed records on effluent ponds. For example, staff at Environment Bay of Plenty noted: *“All of the farms in the attached spreadsheet should have at least a single pond (or possibly a barrier ditch), however I cannot guarantee it”*. DairyNZ staff in Southland reported *“We have approx. half of the systems in Southland that have ponds and the other half that will be required to put them in over the next 2-3 years.”* By contrast, some councils keep very detailed records on pond use, such as Taranaki, where the exact number of multiple-pond treatment systems are recorded.

For regions where data on pond type were available, approximately 37% of farms have two or more ponds, most likely as some form of treatment system or following conversion of an existing traditional two-pond system. However, it should be noted that data on pond type were lacking for several major dairying regions including Hawkes Bay, Manawatu-Wanganui, Wellington, Southland, Canterbury and Tasman. Nonetheless, this figure of 37% of farms having two or more ponds matches well with a survey (covering about 10% of NZ's dairy farms) by Kira et al. (2008) which found that 41% of dairy farms had a two-pond system.

Our review of data showed that the sizes of effluent ponds on dairy farms are extremely variable, with some ponds being small barrier ditch systems with volumes of only about 100  $\text{m}^3$  while other storage and treatment ponds had recorded volumes of >5000  $\text{m}^3$ . The average FDE pond volume across NZ was 1745  $\text{m}^3$ , not weighted, or 2015  $\text{m}^3$  if each region with data in Table 1 is weighted equally. The storage capacity of FDE ponds was also found to be variable, as capacity is closely linked with volume. Storage capacity ranged from just a few weeks in Canterbury to over 3 months in the Bay of Plenty. The average storage capacity for FDE ponds was over 86 days, not weighted, and over 70 days if weighted by region.

Table 1: Details of FDE ponds by region across NZ. Data from regional councils and DairyNZ (2011), partly updated from Pratt et al. (2012). Numbers in red are precise values, other numbers are estimates from available information. NA = information not available

Region	No. of dairy herds	Average herd size	No. of farms with more than one pond	No. of farms with any ponds	Average pond volume (m <sup>3</sup> )	Average storage capacity for ponds (days)
Northland	930	296	700	930	1680	>90
Auckland	446	248	59	231	NA	NA
Waikato	4200	318	450	4200	1100	>90
Bay of Plenty	700	320	243	700	1967	111
Gisborne	4	640	1	3	1825	56
Hawkes Bay	86	657	NA	74	1987	NA
Manawatu-Wanganui	950	475	NA	558	3067	NA
Taranaki	1795	279	1074	1074	2724	NA
Wellington	177	355	NA	67	NA	NA
Southland	872	555	NA	872	2714	82
Marlborough	61	277	17	61	1075	NA
Canterbury	922	750	NA	248	NA	26
Tasman	180	352	NA	180	NA	NA
Nelson	1	352	0	0	NA	NA
West Coast	372	390	220	372	NA	>30
Otago	381	559	152	152	NA	NA
NZ total	12077	384	> 2916	9722	1745	> 86

### 3.2.1 Trends in effluent pond usage, capacity and operation

On dairy farms without effluent ponds, the effluent from the milking shed is held in a small sump (typically < 50 m<sup>3</sup>) and from there irrigated directly to pasture, usually daily. On the basis of information obtained during this project, it appears that milking shed effluent from 23% of NZ's dairy cows must be irrigated directly to pasture from a sump, compared with 27% reported in 2012 (Pratt et al. 2012). This decrease in direct land irrigation likely reflects tighter regulation on FDE storage by regional councils across the country. The preferred practice is to have some form of FDE storage as contingency for periods when direct application of effluent to land is unsuitable (i.e. wet periods when soils are beyond their water-holding capacity). This is noted in a best-management guidelines document published by Dairy Insight et al. (2007a): *“Pumping directly from a sump on a daily basis is not recommended practice”*. *“An alternative to direct land application is to pump effluent from a storage or treatment pond when conditions suit. This is called deferred irrigation, and is the recommended option for Waikato dairy farmers.”*, and by Dairy Insight et al. (2007b): *“However, daily land application is a high-maintenance and labour-intensive option, prone to system failure and with little flexibility. Direct application also provides the least opportunity for the control of any harmful pathogens. As such it is not a recommended option without back-up storage capacity.”* In light of these considerations, there is a trend towards the installation of deferred irrigation storage ponds on NZ dairy farms.

This trend for increased installations of effluent ponds on farms is likely to approach complete coverage in the coming years, via promotion and eventually regulation: *“In recent years the dairy industry has been working with farmers on a national scale in order to get farms to have adequate storage so that effluent is only applied to land at optimum times. This being when effluent can be applied at an application rate less than the soil moisture deficit at the given time. By doing so there is much less chance of causing ponding, run-off or leaching through the soil to groundwater. To aid this movement a tool called ‘the pond calculator’ was developed by Massey University and Environment Waikato. This is now being used by the industry to provide the farmer with an indication of the pond size needed for them to achieve these best practices objectives. I bring this to your attention as this is the way ALL farms in the country will be (in the first instance) encouraged to take, and later forced through regulation of the industry. This meaning all farms will have storage ponds.”* (Tasman Regional Council, pers. comm., 2012).

The presence of an effluent pond system on a farm does not necessarily imply that deferred irrigation is practised by the farmer. It is still possible to apply the effluent frequently and keep the ponds relatively empty; for the objective of minimising CH<sub>4</sub> emissions this would be desirable. However, for the reasons given in the citations highlighted above (Dairy Insight et al. 2007a,b) such practice is not recommended. Generally it can be expected that increasing effluent storage capacity will lead to larger amounts of effluent being stored for some time, under anaerobic conditions, before being applied to land. Here, pond operation is a critical source of uncertainty; although councils may provide design and operational guidelines, it is unclear how farmers indeed operate their ponds.

The storage capacity recommended for “best practice” is designed to hold effluent from prolonged wet periods in unusually wet years (1 year in 30). In most years it will be possible to keep storage times much shorter and spread the FDE onto land without risking ponding, run-off or leaching. The actual storage times will vary widely, dependent on weather conditions and other factors that may influence farm management decisions. It is not possible to estimate average storage times on the basis of the data obtained for the present report. Data on actual FDE pond management would probably require the undertaking of targeted surveys. However, it should be noted that storage time is equal to neither hydraulic retention time (HRT) nor solids retention time (SRT), which are critical parameters in estimating CH<sub>4</sub> emissions during wastewater management. Once a storage pond is partially filled, it is possible to remove some of its contents daily for irrigation; however, settled solids may be retained in the pond and therefore have a longer SRT.

### 3.2.2 Other trends in effluent management

It is not only manure from milking sheds that enters effluent ponds on dairy farms. Many farms in NZ use feed pads where cows can feed prior to milking. From many of these structures, the deposited manure is washed down into an effluent pond (DairyNZ, pers. comm., 2011). Luo et al. (2013) reported that 27% of NZ dairy farms had a feed pad. This number has been rapidly increasing. For example, in 2010 only 33% of farms in Northland had a feed pad (DairyNZ, pers. comm., 2011); in 2013, information from Northland Regional Council (pers. comm., 2013) indicated that nearly every farm in the region had a feed pad. Similarly, in Bay of Plenty, 17% of farms had feed pads in 2010 (DairyNZ, pers. comm., 2011); in 2013 the number had risen to approximately 90% (Bay of Plenty Regional Council, pers. comm., 2013). Given that cows spend approximately the same amount of time on feed pads as in the milking shed (Chung et al. 2013; Bay of Plenty Regional Council, pers. comm., 2013), effluent deposition into ponds across NZ will be much higher with this increased feed



pad use. By how much will depend on the fraction of feed pads from which the manure is washed down into ponds. Alternatives are to use an absorbent material such as bark or sawdust, or to manage the waste as a solid or slurry, in particular where feed pads are roofed.

Further, many farms across NZ employ stand-off pads to keep cows off the pastures for prolonged periods when soils are saturated. Luo et al. (2013) reported that 22% of NZ's dairy farms had a stand-off pad. Another 2% provided winter shelters or housing for their animals. Chung et al. (2013) estimated that the effluent from 40% of these farms would be discharged into effluent ponds. As with feed pads, the uptake of stand-off pads across the country is reportedly on the rise (Chung et al. 2013). These data rely on estimates by the dairy industry (Fonterra) or regional councils, with considerable uncertainty. Yet, all dairy farms have milking sheds, and often yards, which farmers can use like stand-off pads to some extent. Consequently, the proportion of farms having a stand-off pad (22%) can be considered as a lower limit for the number of farms applying stand-off practice.

The trend towards greater use of feed and stand-off pads across the country will, in all likelihood, result in increased volumes of manure captured by FDE management systems. Increased capture and storage will reduce N<sub>2</sub>O emissions following irrigation, so any adjustment in the inventory for manure management should be accompanied by a corresponding adjustment in the inventory for agricultural soils.

Once manure enters a farm's waste management system, there are many factors that will affect the extent of GHG emissions. One such factor is the practice of solids separation. No firm numbers are available on the prevalence of this practice but it seems more farms are adopting solids separation due to higher effluent loading rates into ponds caused by increased use of feed and stand-off pads. DairyNZ (pers. comm., 2013) noted that the practice of solids separation *"...is on the rise and quite common in Southland and Canterbury due to larger herd sizes but we are seeing more being installed in the Waikato."* The effect of solids separation on CH<sub>4</sub> emissions from ponds is difficult to ascertain. If done efficiently, CH<sub>4</sub> emissions should decrease because the manure solids are removed from the anaerobic pond environment. However, in practice, complete fractionation of solids from liquid in dairy effluent is difficult to achieve, and solids-separation pits may simply just become concentrated anaerobic environments. Moreover, the effect of solids separation on N<sub>2</sub>O emissions needs consideration.

Another parameter that may affect GHG emissions is the degree to which the FDE in a pond is mixed. In fully-mixed systems the period of time that the liquid fraction of effluent remains in the pond (hydraulic retention time, HRT) will be the same as the time that the solid fraction remains in the pond (solids retention time, SRT). DairyNZ (pers. comm., 2013) indicated that many farmers are moving towards using agitators in ponds to keep solids in suspension in order to avoid damaging the pond lining, which would otherwise occur if the pond needed to be de-sludged with a digger. In these cases, the SRT of the pond would be relatively short and the ultimate CH<sub>4</sub> emissions potential of the manure may not be reached before the solids are discharged to pasture, where anaerobic degradation would be minimal. Yet, while the effect of mixing (shorter SRT) may theoretically reduce CH<sub>4</sub> emissions, there are no studies to date showing whether they do so in existing FDE ponds. In addition, mixing requires energy, and CO<sub>2</sub> emissions generated to produce this energy should be included in a GHG budget of FDE ponds with agitators.



Figure 4: Effluent storage pond in Southland with a well-developed crust. Crusts can form when the effluent loading rates into ponds are high and when there is little mixing of the effluent in the pond. (Photo: C. Pratt)

Pond crusting is another factor that can affect CH<sub>4</sub> emissions. Some effluent ponds develop extensive crusts on their surface and it has been reported that these crusts can mitigate CH<sub>4</sub> emissions from ponds through the presence of CH<sub>4</sub>-oxidising bacteria (methanotrophs). These crusts are more common in European effluent management systems where manure is stored as slurry rather than in a pond, yet on some NZ farms crusts do form on the surface of effluent ponds (Figure 4). Sommer et al. (2000) looked at oxidation rates in slurry crusts stored in large tanks, yet their results were inconclusive. During more than 10 sampling episodes spanning a year there were four occasions where CH<sub>4</sub> emissions from a crusted slurry were lower than from an uncrusted slurry, yet on another three occasions the opposite was true (for the other sampling events there was no clear difference between the crusted and uncrusted slurry). Based on rates reported in incubation tests on dairy manure reported by Petersen et al. (2005), it appears that a typical crust of 10 cm thickness can achieve only about a 4% decrease in CH<sub>4</sub> emissions from ponds. Overall, the effectiveness of crusts as a CH<sub>4</sub> mitigation strategy can be considered negligible based on the above results as well as the fact that farmers generally try to avoid crusting on ponds as these can hamper effluent treatment and irrigation.

### 3.3 Land application of effluent

The environmental impacts of FDE application to land are not well understood. The nutrients contained within FDE can still be mobilised and enter surface water bodies or be metabolised within the soil, leading to GHG production and emission. When either slurries or manures are applied to land it is widely acknowledged that their DM content largely affects both of these processes. Therefore, the nutrient and DM content of FDE is likely to influence the environmental impacts of land-applied FDE. These characteristics vary seasonally due to lactation, but also vary over longer periods due to changing farming practices (Longhurst et al. 2000). Recently, a trend of increasing DM and nutrient content in FDE has been identified due to the intensification of dairying in NZ (Longhurst et al. 2000). Greater amounts of excreta are being washed off milking pads into storage ponds with no or little increase in water volumes used for washing down these concrete surfaces (Longhurst et al. 2000),

leading to an increase in the nutrient and solid content of FDE. The mean N content of FDE doubled from approximately 200 to 400 mg L<sup>-1</sup> between 1977 and 1997 while during the same period DM content rose from 0.72 to 0.92% (Longhurst et al. 2000). Overall, Longhurst et al. (2000) reported that FDE solid content averaged 0.9% but ranged between 0.04 and 5.2% over 63 sites. N content varied in terms of total N loading and the various fractions that contribute to the total N. Total N content averaged 269 mg L<sup>-1</sup>, ranging from 181 to 506 mg L<sup>-1</sup> (Longhurst et al. 2000). The majority (80%) of the total N is in the organic N form, with ammonium-N accounting for between 10 and 20% and nitrate plus nitrite accounting for less than 3%. Phosphorus (P) concentrations ranged between 40 to 80 mg L<sup>-1</sup> and averaged close to 70 mg L<sup>-1</sup>.

These nutrient concentrations suggest there are both benefits and concerns regarding disposing of FDE by land application. When applied to land, either in their raw state or after treatment in the two pond system, FDE increases pasture yields by providing nutrients (Longhurst et al. 2000) and therefore reducing farm fertiliser costs (Jacobs et al. 2008; Schröder et al. 2007). However, land application of FDE may result in the degradation of water quality if FDE is leached from, or washed over soils into waterways. Surface applications of FDE can generate run-off that is high in total P and total N (McDowell et al. 2005), and N from land-applied FDE may also leach through soils (Di et al. 1998, 1999; Silva et al. 1999; Monaghan and Smith 2004) reaching ground water and potentially entering surface waterways. Also, land application of FDE may result in the emission of N<sub>2</sub>O, NH<sub>3</sub> and CH<sub>4</sub>. In the following, the available scientific literature on these emissions and the leaching of N after application of FDE to soils is reviewed. Because of the limited number of studies found for FDE, this review will also draw on literature relating to the land application of slurries, which can be regarded as a concentrated form of FDE (Misselbrook et al. 2000).

### 3.4 Differences between NZ and overseas practices

In order to evaluate the relevance of overseas studies of atmospheric emissions from dairy farm manure management, it is important to understand how NZ systems differ from overseas practices. Most available international studies have been carried out on intensive, fully housed dairy farms in climates colder than NZ's. This means, firstly, that the evaluated manure management systems are handling 100% of the manure and urine excreted by dairy cows, not just a smaller fraction of the total excreta as in NZ. Further, only small volumes of dilution water (wash water) are added to the manure, which generally has a DM concentration (TS or VS) between 5% and 10%. FDE in NZ has typically a 10 times lower solids concentration. Varying volumes of bedding material, such as straw, can also be present in the manure from housed animals, increasing DM concentrations further.

Most of the reviewed literature was from countries where dairy cow breeds are generally heavier than the widespread Friesian-Jersey cross of NZ (such as Canada, Austria, Switzerland, Scandinavia). The resulting higher feed intake of these larger animals, plus the common use overseas of concentrated feeds, indicates that figures expressed on a per cow basis (such as daily VS excretion etc.) are generally higher than the respective factors in NZ. The extensive use of concentrated feeds overseas should lead to a manure with a higher CH<sub>4</sub> potential (B<sub>0</sub>) per kg DM or VS. However, this is somewhat counterbalanced by the higher feed digestibility of these concentrates, and according to some literature, the differences in dairy cattle manure B<sub>0</sub> resulting from different dietary composition are generally minor (IPCC 2006).



On North American and Western or Northern European dairy farms, manure is often stored in round concrete or timber tank-type structures of considerable depth (3 – 8 m). When emptied, tank-type structures retain a very small volume of residual manure. By comparison, effluent ponds in NZ have a larger surface to volume ratio, are generally not emptied as completely as tank-type structures, and only rarely develop a substantial and long lasting surface crust.

In colder or more continental climates than NZ's, the main manure storage period is during winter, and part of the stored manure is then often frozen. Both lower temperatures and frozen layers generally reduce gaseous emissions, and/or delay their occurrence until springtime. Manure pH levels during storage, particularly in cold climates, are usually acidic, with  $\text{pH} < 6.5$  (Park et al. 2006; Massé et al. 2008; Rodhe et al. 2009; VanderZaag et al. 2011), while FDE in earthen storage ponds in NZ generally retains a neutral to alkaline pH level. A lower pH suppresses emissions of both  $\text{CH}_4$  and  $\text{NH}_3$ .

Finally, overseas dairy farm manure is generally handled and land-applied as slurry with tanker trailers, rather than irrigated through pipelines. The main concern of much research into slurry spreading has been the large amounts (and also large variability) of rapidly volatilised  $\text{NH}_3$ , recently reviewed by Sintermann et al. (2012). The use of pre-treatment technology such as solids separators is not common, other than for specialist applications.

## 4 Greenhouse gas emissions from effluent storage and treatment

### 4.1 CH<sub>4</sub> emissions

#### 4.1.1 Cow shed and yard emission factors

The studied literature on manure management on dairy farms provided no information about CH<sub>4</sub> emissions from milking parlours and holding yards. Under typical NZ farming conditions, manure deposited in or around the milking parlour is washed into the storage pond within hours of deposition. Methane emissions from these areas should therefore be very minor and could probably be ignored, despite the deposited manure being biologically active and inoculated with CH<sub>4</sub>-producing microorganisms.

#### 4.1.2 Pre-treatment emission factors

Fangueiro et al. (2008) reported that separated dairy manure solids emitted CH<sub>4</sub> during storage, albeit in small volumes. Over a 48-day storage period, emissions of 0.004 m<sup>3</sup> CH<sub>4</sub>/kg TS, or 0.55% of total carbon, were observed. These figures are in line with values from Hansen et al. (2006), who observed between 0.17% and 1.3% of initial carbon being emitted as CH<sub>4</sub> during a 120-day storage experiment with solids-separator solids from pig manure. Furthermore, Chadwick (2005) reported CH<sub>4</sub> emissions from the storage of solid cattle farmyard manure, a substrate rather similar to solids-separator solids, ranging between 0.4% and 9.7% of initial carbon. These figures suggest that solids separation is not a major source of CH<sub>4</sub> emissions in the NZ situation, relative to the other steps involved in manure management on dairy farms.

No literature information about CH<sub>4</sub> emissions from weeping-wall systems was found. Considering the wide range of operating conditions weeping-wall systems may have, emission factors and rates may be highly variable. There is an urgent need to get NZ-specific CH<sub>4</sub> emission data for weeping walls, since weeping-wall systems are increasingly being used on NZ dairy farms.

#### 4.1.3 FDE pond emission factors

Studies that evaluated CH<sub>4</sub> emissions from the storage of dairy farm manure, as well as other livestock wastes such as pig manure, are summarised in Table 2. However, most of these studies had shortcomings that make it difficult to interpret the results in the NZ context. Most important in this regard is that almost all studies were undertaken as batch experiments using high-DM manure without an adapted anaerobic inoculum. It is therefore not surprising that many of these batch storage experiments reported low to negligible CH<sub>4</sub> emissions, e.g. Zeeman et al. (1988); Külling et al. (2002, 2003); Massé et al. (2003). These are not included in Table 2 because the experimental conditions in these studies are not representative of actual field conditions in NZ, where some adapted methanogenic archaea will exist in the low-DM effluent discharged into FDE ponds.

The authors of several manure storage and anaerobic digestion experiments conducted at ambient temperatures (Zeeman et al. 1988; Sommer et al. 2007; Massé et al. 2008) concluded that the presence of a temperature- and substrate-appropriate microbial inoculum is important for efficient CH<sub>4</sub> production. For example, Zeeman et al. (1988) found that inoculation could initiate CH<sub>4</sub> generation from cow manure at temperatures as low as 5°C.

Table 2: Summary of CH<sub>4</sub> emission factors for manure from the literature

Source	Stated factor	Factor converted (approx.) to m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS	Explanation	Relevance for NZ situation
<b>Solids separator solids / solid manure / pre-treatment emissions</b>				
Fangueiro et al. (2008)	0.55% of total carbon emitted as CH <sub>4</sub> from solids-separator solids over 48 d storage period	0.004	Directly comparable to NZ situation	high
van der Weerden et al. (2014b)	0.27% of initial carbon emitted as CH <sub>4</sub> from weeping-wall solids over 112 d; this increased to 5.1% when stored for 197 d	0.002 to 0.038	NZ-specific data, field incubation	high
Hansen et al. (2006)	0.17% to 1.3% of initial carbon being emitted as CH <sub>4</sub> from solids-separator heaps over 120 d	0.0013 to 0.0097	Experiment with pig manure. Directly comparable to NZ situation	medium
<b>Liquid manure / slurry / pond emissions</b>				
Wood et al. (2012)	5.9 kg CH <sub>4</sub> m <sup>-2</sup> at 1.1% VS	0.33 at 1.1% VS	Batch experiment with no inoculum, lag time. Cattle manure stored at 10 to 20°C for 180 d VS concentrations from 0.2 to 6.8% VS, include solids concentrations (i.e. 1.1% VS) relevant for NZ	high
Craggs et al. (2008)	0.211 m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS	0.211	NZ field measurement	high
Umetsu et al. (2005)	0.19 m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS over 150 d manure storage at 20°C	0.19 over 150 d manure storage at 20°C	Batch experiment with high solids content manure (9.0% TS, 7.7% VS). However pH remained neutral to alkaline in all experiments – i.e. no pH inhibition	medium
Park et al. (2006)	23% of B <sub>0</sub> (0.298 m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS) for dilute pig manure 0.6 to 3% VS stored in cold climate Canada (average manure temperature ca. 12°C)	0.07	Filed flux measurement of pig manure storage Only partially relevant since storage time not specified Most importantly manure partially froze in winter and manure temperature dropped below 5°C, indicating hardly any winter activity	low

Source	Stated factor	Factor converted (approx.) to m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS	Explanation	Relevance for NZ situation
Sommer et al. (2007)	Peak CH <sub>4</sub> emission of 0.08 g C h <sup>-1</sup> kg <sup>-1</sup> VS from cattle slurry	Up to 0.14 at 20°C	Sequential batch fill experiments indicate importance of an active microbial inoculum for substantial CH <sub>4</sub> production. No pH data given, but results indicate pH inhibition or activity reduction in all of these experiments Manure more concentrated (ca. 8% VS) than NZ effluent	low
Massé et al. (2008)	For 5.9% VS manure:  0.08 m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS at 20°C  0.04 m <sup>3</sup> CH <sub>4</sub> kg <sup>-1</sup> VS at 10°C	For 5.9% VS manure:  0.08 at 20°C  0.04 at 10°C	Manure more concentrated than NZ effluent Batch experiment Author stresses the importance of active microbial inoculum – hard to simulate in batch experiment, but present in the field pH inhibition in lower CH <sub>4</sub> producing batches (pH < 6.5) clearly apparent.	low
Minato et al. (2013)	1.42% (g CH <sub>4</sub> g <sup>-1</sup> VS)	0.021	High-strength waste (5 to 6% VS) under field conditions, low storage-vessel pH (6.4 to 6.5) and consequently low CH <sub>4</sub> emissions	very low

The importance of an inoculum for the formation of CH<sub>4</sub> during manure storage experiments appears to be two-fold. As well as providing a substrate- and temperature-adapted microbial culture, the addition of inoculum appears to be important in maintaining pH levels that support anaerobic digestion, particularly for high-strength wastes, which tend to acidify and stop digesting if the initial and faster acid digestion phase is not countered by methanogenesis. Moreover, effluent pH is more likely to be maintained at near-neutral levels when it is buffered by dilution with water. Since FDE in NZ is typically 10 times more dilute than manures overseas, it is likely to have a more neutral pH and therefore to digest more completely throughout the storage period.

The relationship between dilution of dairy farm manure and resultant CH<sub>4</sub> emissions during storage has been analysed by Wood et al. (2012) who derived a mathematical regression model. They concluded that in the samples with a VS concentration of less than 2% all available carbon was digested and converted to CH<sub>4</sub> (biogas). Wood et al. (2012) did not report the manure pH values during or at the end of the storage experiment; the limiting effect of pH on CH<sub>4</sub> formation as a consequence of high solids concentrations is therefore only inferred by this study.

Working with manures of a higher solids concentrations (4.2 to 11.3% TS), Massé et al. (2003) concluded that a lower solids concentration leads to higher relative and absolute CH<sub>4</sub> emissions during storage. Umetsu et al. (2005) conducted CH<sub>4</sub> emission experiments with 7.7% VS cow manure at 20°C. For unexplained reasons the pH in all batches remained neutral or alkaline, and as a consequence an unusually high CH<sub>4</sub> production rate of 0.19 m<sup>3</sup> CH<sub>4</sub>/kg VS for concentrated manures was observed.

During an experiment with manure from two dairy farms stored at 10°C and 20°C, Massé et al. (2008) observed that the sample with a higher DM content (8.8% VS) produced much less CH<sub>4</sub> than the more diluted sample (5.9% VS), both at higher and lower storage temperature over a 370-day storage period. While the pH in the lower-concentration sample remained slightly alkaline (~7.5), a change in pH in the more concentrated sample from 6.5 to neutral conditions halfway through the experiment was accompanied by a corresponding and dramatic increase in CH<sub>4</sub> production from this sample. The cause and effect of acidic pH levels (at high solids concentrations) and low CH<sub>4</sub> production during storage was also confirmed by Chadwick et al. (2011), who report that experiments aimed at reducing NH<sub>3</sub> emissions during storage through acidification also brought about reduced CH<sub>4</sub> emissions.

For effluent ponds in NZ, however, comparatively high emission factors have been reported. A field study by Craggs et al. (2008) yielded a value of 0.21 m<sup>3</sup> CH<sub>4</sub>/kg VS, higher than the emission factor of 0.17 m<sup>3</sup> CH<sub>4</sub>/kg VS recommended by the IPCC for dairy manure lagoons in the Oceania region (IPCC 2006), calculated as the maximum potential (0.24 m<sup>3</sup> CH<sub>4</sub>/kg VS) times a climate-dependent conversion factor of 0.71. These high emission factors can be understood by the chemical and physical differences between FDE in NZ and manures that are typical overseas. The latter have a high solids concentration and acidic pH; the former have a low solids concentration and neutral or alkaline pH. Therefore, the low CH<sub>4</sub> emission factors of overseas manures are not relevant for the NZ situation. Lack of dilution, pH buffering and inoculation of adapted anaerobic archaea likely explain the large differences between these overseas literature values and those recently found in NZ (Craggs et al. 2008; Pratt et al. 2012). While these were observed for treatment ponds, they are most likely also relevant for deferred-irrigation storage ponds, which accumulate anaerobic solids on the pond bottom.

## 4.2 N<sub>2</sub>O emissions

Interpretation of the reviewed literature regarding N<sub>2</sub>O emissions was found to be complex and complicated, not only due to the difference in manure characteristics between NZ and many overseas situations, but also because of the multi-point and multi-pathway nature of N<sub>2</sub>O emissions. Figure 5 and Figure 6 (see Section 4.3) provide an overview of the potential *direct* and *indirect* N<sub>2</sub>O emission pathways for manure management on NZ dairy farms, respectively. This includes some that are not considered by the IPCC guidelines for GHG inventories (IPCC 1996; 2006), namely those for emissions from pre-treatments and indirect emissions from milking parlour and holding yards. To achieve a complete budget, N<sub>2</sub>O emissions from dairy-farm manure management in NZ should be accounted for as the sum of 8 to 12 (depending on pre-treatment) individual emission pathways (Figures 5 and 6).

In comparing this multitude of possible N<sub>2</sub>O emission pathways with the current inventory methodology (see Figure 8, Section 6.2) it can be concluded that simply providing more applicable direct and indirect N<sub>2</sub>O emission factors for the current methodology may be insufficient to provide a comprehensive and accurate picture of the current NZ situation, since a number of potential emission pathways are not adequately covered by the current

inventory methodology (e.g. N<sub>2</sub>O emissions from pre-treatment). To overcome these shortcomings, it is therefore recommended that the mass balance methodology suggested by Dämmgen and Hutchings (2008) is adopted to account for both N<sub>2</sub>O and NH<sub>3</sub> emissions from dairy farming manure management. Use of this methodology would help to address the omissions in the current inventory methodology, and be flexible enough to incorporate future changes and newly emerging technologies in future inventories.

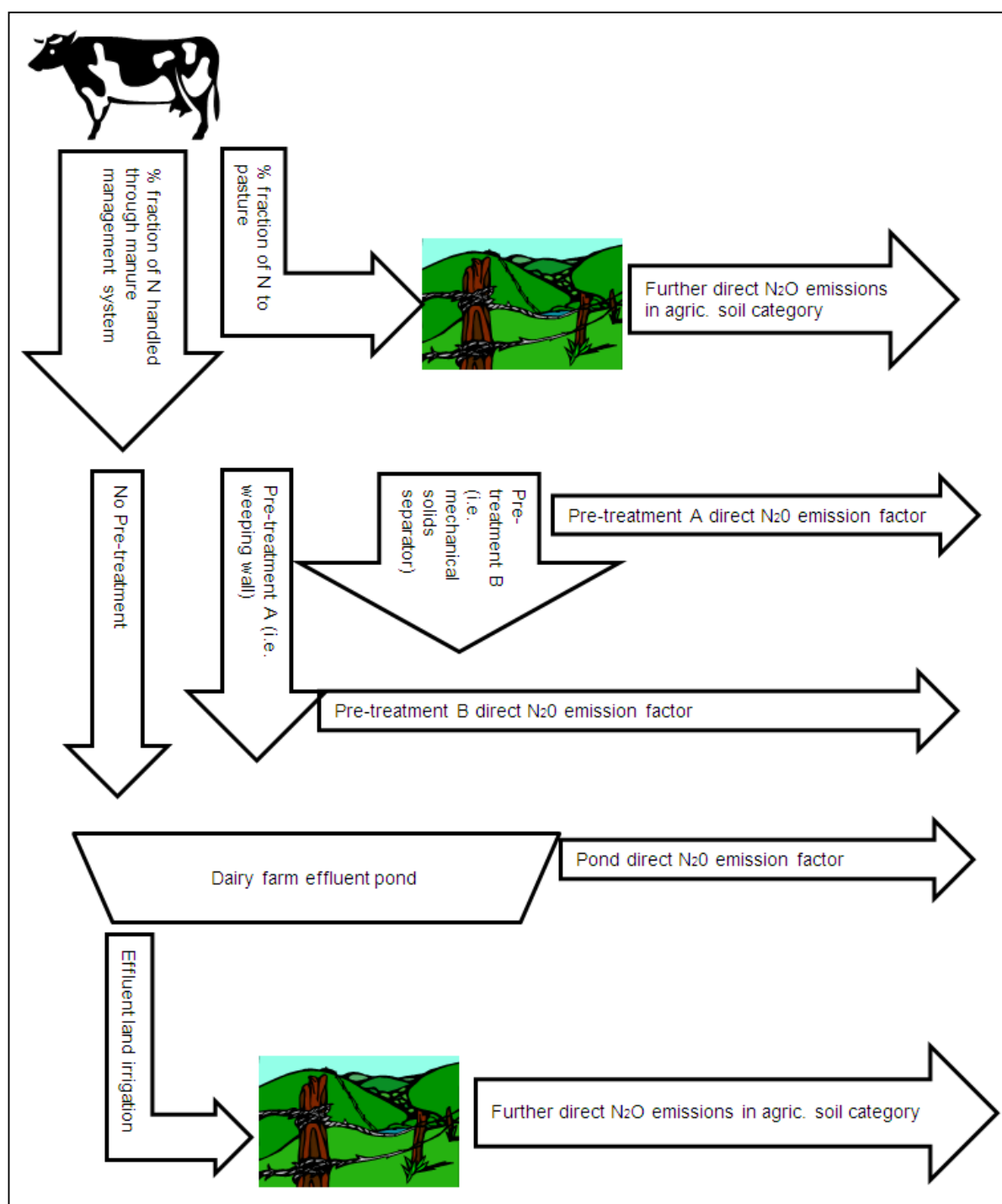


Figure 5: Schematic diagram of possible direct N<sub>2</sub>O emission pathways in NZ dairy farming systems.

Direct N<sub>2</sub>O emission pathways (Figure 5) and emission factors are reviewed in the following subsections, and summarised in Table 3. Indirect N<sub>2</sub>O emissions are those that occur as a consequence of N volatilisation losses in other forms (predominantly as NH<sub>3</sub>) and redeposition of this airborne N elsewhere; they are thus reviewed in the section on NH<sub>3</sub> emissions. (Indirect emissions also occur where NO<sub>3</sub><sup>-</sup> is transported off-site or leaches from FDE after land application, these are not separately considered here.)

#### 4.2.1 Cow shed and yard emission factors

No direct N<sub>2</sub>O emission factors for milking sheds and holding yards were found in the literature. Since manure is present in these areas for only very short periods, it is unlikely that direct N<sub>2</sub>O emissions from this source are significant.

#### 4.2.2 Pre-treatment emission factors

Pre-treatment systems such as mechanical solids separators can be a source of direct N<sub>2</sub>O emissions (Hansen et al. 2006; Fangueiro et al. 2008; Chadwick et al. 2011).

Mechanical solids separators, in particular screw-press-type separators, produce solids heaps that are consistent in composition, irrespective of whether the original manure has been diluted with washwater or not. This means that emission factors measured overseas can be applicable to the NZ situation, dependent on DM composition. An ambient-temperature storage experiment using dairy cow manure solids from a screw-press separator recorded direct N<sub>2</sub>O emissions of 0.62% of total nitrogen (TN) and 3.2% of total ammoniacal nitrogen (TAN) from the solids heap over a 50-day storage period (Fangueiro et al. 2008). Hansen et al. (2006) found an N<sub>2</sub>O loss of 4.8% of TN over a 4-month storage period of solids separated from pig slurry, and concluded that liquid manure separation leads to an overall increase in N<sub>2</sub>O emissions compared with the management of the manure as a liquid. These direct N<sub>2</sub>O emission factors from separated-solids heaps are similar to emission factors for solid cattle or pig farmyard manure, which vary between 1% and 9.8% of TN (Chadwick et al. 2011; Webb et al. 2012). These values are in line with those used by Dämmgen and Hutchings (2008), who modelled direct N<sub>2</sub>O emissions of between 5.6% and 10.4% of TAN for cattle manure storage.

Direct N<sub>2</sub>O emissions from separator solids heaps end when the material is land-applied, which generally brings the substrate to a completely aerobic state. However, the TN content of the separator solids that are land-applied may contribute to further N<sub>2</sub>O emissions from soils, which are separately reported in the agricultural soil category of the national inventory.

No studies were found that evaluated direct N<sub>2</sub>O emissions from weeping-wall pre-treatment systems. Since these systems come in many designs and are operated quite differently from farm to farm and even during an individual dairy season on the same farm, it can be expected that direct N<sub>2</sub>O emissions from weeping walls will vary greatly. The lack of international and, more importantly, NZ-specific data on weeping-wall N<sub>2</sub>O emissions represents a substantial knowledge gap that should be addressed with urgency, firstly because such systems are becoming widely used (e.g. in Southland) and secondly because direct N<sub>2</sub>O emissions could be quite high, since a weeping-wall system represents a high-energy and high-nitrogen environment that may cycle between aerobic and anaerobic conditions (up to twice a day for the upper layer). An indication that direct N<sub>2</sub>O emissions will occur from weeping-wall systems is provided by studies from manure storage facilities that report the presence of a surface crust substantially increases direct N<sub>2</sub>O emissions (Chadwick et al. 2011). For example, Hansen et al. (2009) found that the presence of a crust, with conditions changing

between anaerobic and aerobic, increased N<sub>2</sub>O emissions from 4 g TN L<sup>-1</sup> pig manure by 100-fold (> 800 mg N<sub>2</sub>O m<sup>2</sup> d<sup>-1</sup> after 4 to 6 weeks) compared with crust-free, fully anaerobic conditions.

Just as for separator solids, direct N<sub>2</sub>O emissions from weeping-wall solids end with application to land, but the applied N may lead to N<sub>2</sub>O emissions from soils, which are reported in the agricultural soil category of the inventory.

Table 3: Summary of N<sub>2</sub>O emission factors for manure from the literature

Source	Stated factor	Factor converted (approx.) to % TN	Explanation	Relevance for NZ situation
<b>Solids separator solids / solid manure / pre-treatment emissions</b>				
Hansen et al. (2006)	4.8% of initial N lost as N <sub>2</sub> O from a solids separator heap of pig manure during 4 months of storage	4.8	Directly comparable to NZ situation	high
Chadwick et al. (2011)	Emissions of N <sub>2</sub> O are typically from < 0.1% to 4.3% of TN in stored cattle and pig farmyard manure heaps. Emissions as high as 9.8% of TN have been reported	0.1 to 4.3 typically (9.8 max.)	Review of data from 8 studies (2 pig, 5 cattle, 1 chicken manure). Directly comparable to NZ situation	high
van der Weerden et al. (2014b)	0.09% of initial N lost from weeping-wall solids over 112 d; increasing to 0.20% when stored for 197 d	0.20	NZ-specific data, field incubation	high
Fangueiro et al. (2008)	N <sub>2</sub> O emissions from solids separator solids over 50 d of storage 0.62% of TN (3.6% of NH <sub>4</sub> -N)	0.62	Batch experiment with pig manure over 50 d, temperature < 13°C, cooler than in NZ	medium
Moral et al. (2012)	1.0% of TN emitted as N <sub>2</sub> O from 20% DM solids heaps during 52 d storage	1.0	Directly comparable to NZ situation	medium
<b>Liquid manure / slurry / pond emissions</b>				
Hansen et al. (2009)	Liquid manure without crust: 1 mg N <sub>2</sub> O m <sup>-2</sup> d <sup>-1</sup>	N <sub>2</sub> O emissions from liquid manure stores without crust (typical for NZ) are negligible:	Batch experiment with pig slurry, 1.4% VS	high
	Liquid manure with crust: 800 mg N <sub>2</sub> O m <sup>-2</sup> d <sup>-1</sup>	0	Presence of a dry crust on the manure storage facility can reduce NH <sub>3</sub> emissions but increase direct N <sub>2</sub> O emissions up to 100-fold compared with a no-crust situation	



Source	Stated factor	Factor converted (approx.) to % TN	Explanation	Relevance for NZ situation
van der Weerden et al. (2014a)	<0.01% of initial N lost from slurry over 197 d	<0.01	NZ-specific data, field incubation	high
Park et al. (2006)	Under field conditions N <sub>2</sub> O emissions from liquid storage are negligible	0	Field flux measurement of pig manure Very low temperature	high
Sommer et al. (2000)	N <sub>2</sub> O emission from slurry or liquid manure with no surface cover is negligible	0	Directly comparable to NZ situation	high
Wood et al. (2012)	Total emissions of 2 g N <sub>2</sub> O m <sup>-2</sup> at 1.1% VS for manure stored for 180 d	0.08	Batch experiment Direct N <sub>2</sub> O loss approximated to about 2% of NH <sub>3</sub> loss during storage for a wide range of examined TS/VS concentrations	high
Fangueiro et al. (2008)	N <sub>2</sub> O emissions between 0.01 and 0.02% of TN, and 0.03% of NH <sub>4</sub> -N, for the liquid fraction from solid separators and for raw liquid slurry	0.01 to 0.02	Batch experiment with pig manure over 50 d, temperature < 13°C, cooler than NZ. No inoculum DM content of liquid substrates 4.5 to 6.3%	medium
Külling et al. (2002)	0.86 to 0.96 mg kg <sup>-1</sup> undiluted manure per day from manure storage for 14 weeks	0.03	Batch experiment Much less water dilution than in NZ indicates that initial N concentration was higher	low
Külling et al. (2003)	<0.01% total N loss during 5 weeks of storage at 20°C	< 0.01	Batch experiment Manure far more concentrated than in NZ	low

#### 4.2.3 FDE pond emission factors

The reviewed literature concluded that direct N<sub>2</sub>O emissions from slurry or liquid manure storage facilities (ponds, lagoons and tanks) without crusts are negligible (Park et al. 2006; Chadwick et al. 2011). The potential impact of a surface crust on direct N<sub>2</sub>O emissions has been discussed in the previous subsection (Sommer et al. 2000; Hansen et al. 2009; VanderZaag et al. 2009; Chadwick et al. 2011). Dairy farm manure storage and treatment ponds in NZ generally do not tend to develop massive and long-lasting surface crusts, due to dilution, particularly in situations where solids removal is used. It may therefore be appropriate for inventory purposes to assume a zero emission factor for direct N<sub>2</sub>O emissions for properly managed manure treatment and storage ponds on NZ dairy farms.

### 4.3 NH<sub>3</sub> emissions

The review of international literature identified many possible indirect N<sub>2</sub>O emission pathways for manure management on dairy farms that are of relevance to the NZ situation. These are discussed below and summarised in Figure 6. The principal route for indirect N<sub>2</sub>O emissions is through volatilisation of NH<sub>3</sub> gas and subsequent conversion of a fraction (currently a constant 1% assumed in the inventory) of this volatilised N in the wider environment. However, the focus on NH<sub>3</sub> may be too simplistic, and in fact the IPPC guidelines (IPCC 2006) are based on the combined N volatilisation losses of NH<sub>3</sub> and NO<sub>x</sub>. For manure management NH<sub>3</sub> volatilisation is the dominant form of N volatilisation loss to the atmosphere, since most handling and treatment is dominated by anaerobic processes (see Section 3.1). Solids-separator heaps may, however, be a more substantial source of NO<sub>x</sub> emission due to their cycling between anoxic and aerobic conditions (Chadwick et al. 2011; Dinuccio et al. 2012).

On the other hand, Dämmgen and Hutchings (2008) strongly suggest that a higher degree of rigour is required, which should not only account for the very small NO losses from liquid manure storage (~0.1% of TAN), but also for losses of N<sub>2</sub> gas, albeit primarily as a means to account for the reduction in substrate TN and TAN concentrations during each step of the manure management chain, and thereby enable more accurate mass balance models. The mass balance approach of Dämmgen and Hutchings (2008) should be considered as a potential method to improve future GHG emission accounting for dairy-farm manure management in NZ. The following review of national and international literature was conducted focusing on NH<sub>3</sub> volatilisation as the basis for indirect N<sub>2</sub>O emissions; it is summarised in Table 4.

#### 4.3.1 Cow shed and yard emission factors

Emissions of NH<sub>3</sub> from sheds and yards are currently ignored in NZ's GHG inventory (MfE 2013) because they are not a reporting requirement under the 1996 IPCC Guidelines. Dämmgen and Hutchings (2008) give accounting examples for housed animals where they assume that 14.6% of TN excreted is deposited in the milking parlour and holding yard, of which 52.3% is TAN (available for volatilisation). Some 21.5% of TAN deposited in the milking parlour and holding yard is assumed to be volatilised, which represents 1.64% of the TN excreted by the dairy cows (Dämmgen and Hutchings 2008). These model values are relevant for the NZ situation since the time dairy cows spend in the milking parlour and holding yard does not substantially change between a grazing or fully-housed dairy farming system. The 21.5% TAN volatilisation factor for manure deposited in the milking parlour and holding yard is compatible with another modelled factor, namely 30% of TAN volatilisation, assumed by Webb and Misselbrook (2004) for dairy-cow housing systems in the UK. The higher figure of Webb and Misselbrook (2004) would reflect a somewhat longer residence time of the manure in the milking parlour and holding yard, before it is moved into the manure management system, which is less realistic for NZ dairy farms.

#### 4.3.2 Pre-treatment emission factors

Pre-treatment N volatilisation losses and the resulting indirect N<sub>2</sub>O emissions are currently not required to be reported in GHG inventories. Attempts to estimate these would need to distinguish between different technologies (mechanical solids separator, weeping wall, etc.). Some data regarding N volatilisation from solids-separator heaps are available from international literature, which appear to be relevant to the NZ situation. Fangueiro et al. (2008) found 0.42% of TAN volatilised from solids-separator heaps during a 50-day storage period, while Dinuccio et al. (2012) recorded TAN losses of between 11.7% (summer) and

15.7% (winter) during an experiment with pig manure separator solids. In an experiment with cattle-farmyard manure that contained 20% DM, a substrate rather similar to solids-separator solids, 11% of TAN was volatilised, largely during the first week of storage (Moral et al. 2012).  $\text{NH}_3$  emissions from solid manure during storage have been found to vary greatly (0.3 to 34% of the initial N content) with temperature and weather conditions (Petersen et al. 1998; Chadwick 2005; Hansen et al. 2006; Sommer et al. 2007; Moral et al. 2012).

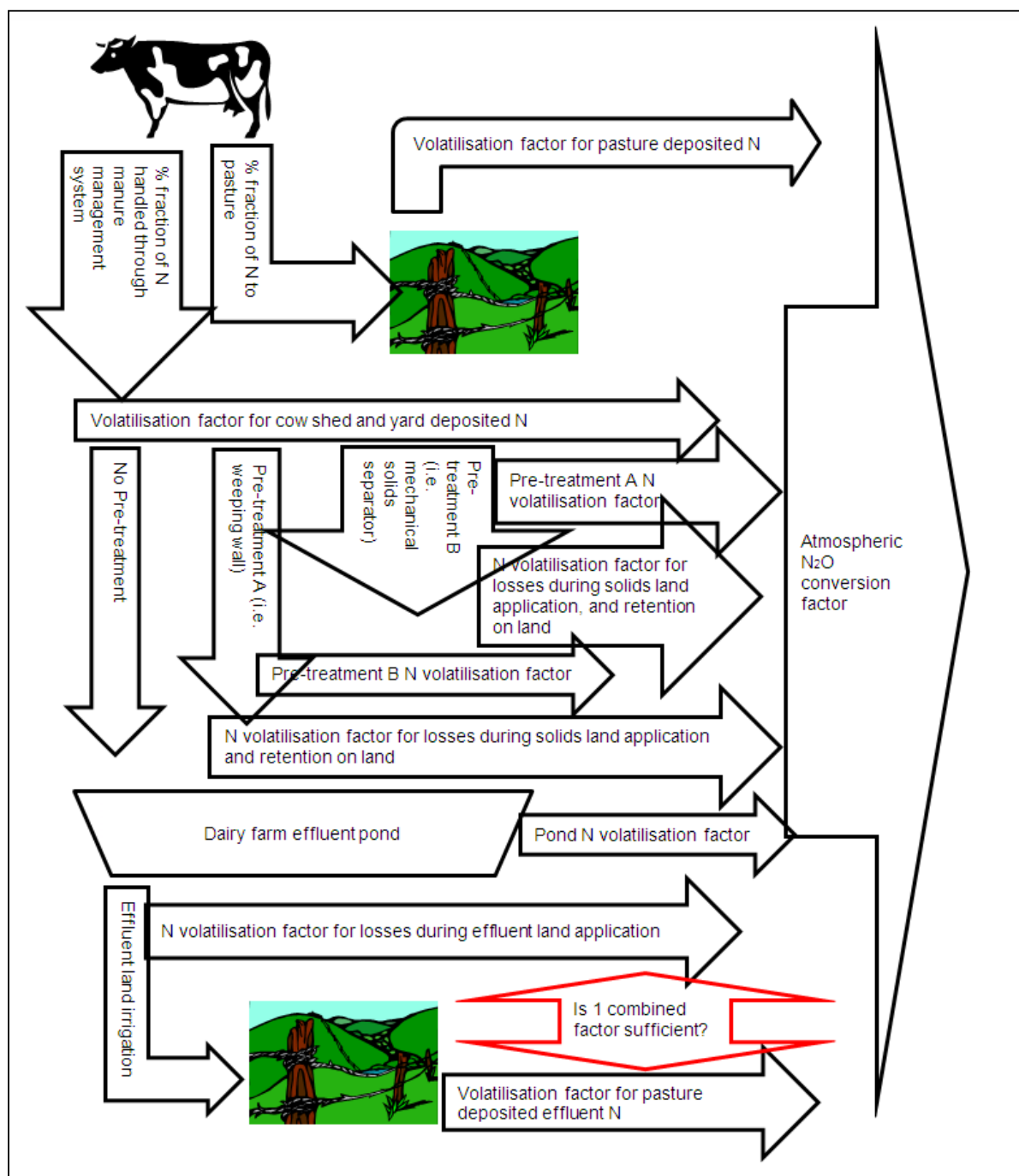


Figure 6: Schematic diagram of possible indirect  $\text{N}_2\text{O}$  emission pathways in NZ dairy farming systems.

**Table 4: Summary of NH<sub>3</sub> emission factors for manure from the literature**

Source	Stated factor	Factor converted (approx.) to % TAN	Explanation	Relevance for NZ situation
<b>Milking parlour and holding yard emissions</b>				
Dämmgen and Hutchings (2008)	14.6% of TN excreted deposited in milking parlour and yard (52.3% of TN being urine), 21.5% of TAN volatilise as NH <sub>3</sub> from yard.	21.5	Despite being data from an overseas housed situation, volatilisation from milking parlour and holding yard is relevant for NZ situation. Note: factor is modelled!	high
Webb and Misselbrook (2004)	30% of TAN may be lost from the milking parlour and housing system as NH <sub>3</sub>	30	Data from overseas fully housed operation. Moderately relevant for NZ. Note: factor is modelled!	low
<b>Solids separator solids / solid manure / pre-treatment emissions</b>				
Dinuccio et al. (2012)	Solids heaps lost 12 – 16% of TAN during storage	12 to 16	Experiment with pig manure over 30 d. Directly comparable to NZ situation	high
van der Weerden et al. (2014b)	5.67% of initial N lost from weeping-wall solids over 112 d; no further NH <sub>3</sub> emissions after that; similar result for bunker manure	20.7	NZ-specific data, field incubation	high
Moral et al. (2012)	1.5% of TN lost during storage of 20% DM solids in heaps for 52 d	ca. 15 (based on final TAN in heap)	Experiment with cattle farm yard manure relatively similar to solids separator solids	medium
<b>Liquid manure / slurry / pond emissions</b>				
Wood et al. (2012)	100 g NH <sub>3</sub> m <sup>-2</sup> storage vessel surface during 180 d of storage for manures with both 0.2 and 1.1% VS	ca. 10	Batch experiment NH <sub>3</sub> losses not proportional to TN content for concentrations as low as 0.2 and 1.1% VS	high
Dinuccio et al. (2012)	NH <sub>3</sub> storage loss of 1.9 to 5.5% of TAN for raw pig slurry of 3.6% TS NH <sub>3</sub> loss of 2.4 to 6.2% of TAN for liquid pig waste of 1.6% TS, separated during storage	1.9 to 6.2	Experiment with pig manure over 30 d Directly comparable to NZ situation	medium
Fangueiro et al. (2008)	1.5 to 2.6% NH <sub>3</sub> loss (based on initial NH <sub>4</sub> -N) over 50 d	1.5 to 2.6	Batch experiment 50 d, temperature < 13°C	medium

Source	Stated factor	Factor converted (approx.) to % TAN	Explanation	Relevance for NZ situation
Külling et al. (2002)	30 to 32% N loss in liquid samples over 14 weeks storage at 20°C	60 to 65	Batch experiment Much less water dilution than in NZ indicates that initial N concentration was higher NH <sub>3</sub> loss found to decrease strongly (per day) with increasing storage time and also with reduced initial N concentration	low
Külling et al. (2003)	14 to 28% total N loss during 5 weeks of storage at 20°C	30 to 60	Batch experiment Manure far more concentrated than in NZ	low
Minato et al. (2013)	0.43% (g NH <sub>3</sub> -N g <sup>-1</sup> total N)	ca. 1	Field experiment with high-strength slurry and acidic pH Daily NH <sub>3</sub> emissions varied widely (0.1 to 1.7 g m <sup>-2</sup> d <sup>-1</sup> )	low

Solids-separator solids contribute to further NH<sub>3</sub> emissions during land application, which also do not need to be accounted for in GHG inventories. International literature generally reports very high N volatilisation losses for land application of solid manure, such as 57 to 96% of TAN for pig manure solids from a separator (Dinuccio et al. 2012) and 90% of TAN for solid cattle manure (Dämmgen and Hutchings 2008). Webb and Misselbrook (2004) calculated that 81% of TAN is volatilised from cattle farmyard manure during land application.

There are few data on NH<sub>3</sub> volatilisation losses from weeping-wall systems and no NZ-specific information. Webb and Misselbrook (2004) set the NH<sub>3</sub> emission factor for weeping-wall systems treating cattle manure to 10.8% of TAN, assuming it to be equal to that for crusted slurry storage tanks. Since the layout and operating regime of weeping-wall systems in NZ differ so widely it is important to collect more NZ field data on weeping-wall N volatilisation factors.

No data on the N volatilisation losses from land application of weeping-wall solids were found, but it can be assumed that such emissions are likely to be between the factors reported for solids-separator solids and dry farmyard manure (of order 57 to 96% of TAN, see above) and those reported for slurry and liquid effluent (of order 1 to 40% of TAN, see Section 5.3).

#### 4.3.3 FDE pond emission factors

Fangueiro et al. (2008) observed NH<sub>3</sub> losses of 1.5 to 2.6% of TAN during a 50-day storage experiment with cattle manure containing 4.5 to 6.3% DM, at cold temperatures (< 13°C). Dinuccio et al. (2012) found losses of 1.9 to 6.2% of TAN from the storage of either raw slurry or its liquid fraction after solid separation; losses in their winter experiment (average storage temperature of 6°C) were less than in summer (average 18°C). In line with these experiments, Dämmgen and Hutchings (2008) assumed that 5% of TAN was lost as NH<sub>3</sub> during the storage of liquid cattle manure, while simultaneously 0.1%, 0.1% and 0.3% of TAN were lost as NO, N<sub>2</sub>O and N<sub>2</sub> respectively as well. Much higher N losses of between 30% and 65% TAN, or 14% and 32% total N, were observed with similar material stored for

between 5 and 14 weeks (Külling et al. 2002, 2003). Even though the losses in these studies may have included substantial  $N_2$  losses, which were not analysed separately, they indicate that  $NH_3$  losses may strongly depend on storage conditions.

The most relevant data for NZ on effluent pond  $NH_3$  emissions are from Wood et al. (2012), who found surface fluxes of  $100 \text{ g } NH_3 \text{ m}^{-2}$  storage vessel surface, equating to ~5% TN loss during the storage of cattle manure with 0.3% and 1.3% TS contents, over 180 days.

The volatilisation of  $NH_3$  from an effluent pond is influenced by manure TAN concentration, temperature, pH, agitation and wind. It appears that some of these factors offset each other, e.g. where dilute, low-TAN effluent may have a relatively high pH compared with higher-strength waste. Field measurements of  $NH_3$  emissions from FDE ponds in NZ would be required to quantify volatilisation losses during storage. For these, the floating-cover method, as applied by Craggs et al. (2008) and Pratt et al. (2012) for  $CH_4$ , is unsuitable for  $NH_3$ . This is because the cover would inhibit wind mixing,  $NH_3$  would build up in the headspace under the cover until the phase-exchange equilibrium between the gaseous and aqueous  $NH_3$  concentrations was reached, after which further  $NH_3$  emission would be suppressed. For meaningful measurements, micrometeorological approaches appear most suitable because they employ the natural wind flow, rather than inhibiting it. These could be realised with  $NH_3$  collection samplers as employed by Laubach et al. (2013b), or with in situ open-path concentration measurements (e.g. Flesch et al. 2007).

#### 4.4 Nitrogen leaching

Recent industry guidelines (IPENZ 2013) state that all parts of FDE handling systems need to be constructed from either tightly sealing synthetic materials (concrete, plastic), or from a clay or earth seal that has been compacted to reduce permeability to less than  $1 \times 10^{-9} \text{ m/s}$ . All effluent handling and storage structures that fulfil these prerequisites will therefore not contribute to any physical leaching losses of effluent.

## 5 Emissions from effluent application to land

### 5.1 CH<sub>4</sub> emissions

To the authors' knowledge, CH<sub>4</sub> emissions resulting from FDE application have not been reported in the literature. Furthermore, data concerning the emission of CH<sub>4</sub> following land application of slurries are also limited (Webb et al. 2012). Therefore, this review will briefly investigate the available literature on CH<sub>4</sub> emissions from slurry application.

Methane emissions resulting from land application of slurries are generally small, occurring within 24 h of application. Chadwick et al. (2000) showed that > 90% of CH<sub>4</sub> emitted over a two-month period took place within 24 h of slurry application. Both pig and dairy cow slurries were applied during this study with the resulting emissions ranging from < 0.1 to 23 mg CH<sub>4</sub> L<sup>-1</sup> of applied slurry. When compared with storing slurries, CH<sub>4</sub> emissions following land application are low. This is due to two factors: firstly, methanogens are sensitive to O<sub>2</sub> and the diffusion of O<sub>2</sub> into slurries is greater after application to land than when compared with during storage, which inhibits CH<sub>4</sub> emissions (Chadwick et al. 2011). Secondly, it is thought that most of the CH<sub>4</sub> emitted from land-applied slurries is produced before the land application and, while the manure is stored, is dissolved within its liquid fraction (Sommer et al. 1996; Rodhe et al. 2006). Soils applied with slurries may even become sinks for, rather than sources of, CH<sub>4</sub>. Rodhe et al. (2006) reports that over a 7-week period following the application of cattle slurry to a grassland soil, the soil consumed 362 g C ha<sup>-1</sup>. However, due to the chemical difference between dilute slurries and FDE, factors that affect CH<sub>4</sub> emission from slurries should be regarded with caution when applied to FDE applications. There is a need for targeted studies to confirm if CH<sub>4</sub> emissions are indeed insignificant.

### 5.2 N<sub>2</sub>O emissions

Specific investigations of N<sub>2</sub>O production resulting from the land application of FDE are limited to four NZ studies (Barton and Schipper 2001; Bhandral et al. 2007; Luo et al. 2008; Li et al. 2014b). The N<sub>2</sub>O emission factors reported in these studies range from 0.01 to 4.9% (Table 5). This range encompasses N<sub>2</sub>O emission factors reported for land-applied slurries, which range from < 0.1 to 3% (Chadwick et al. 2011), suggesting that the differences in the characteristics of FDE and slurries may not be the dominant factor in determining N<sub>2</sub>O emissions, rather other variables such as soil conditions may influence N<sub>2</sub>O emissions and associated emission factors. This part of the review will focus on the four NZ studies where N<sub>2</sub>O emissions from land-applied FDE were observed.

The application of FDE onto soils increases N<sub>2</sub>O production in two separate ways: firstly, by adding N and labile carbon (C), and secondly by increasing soil moisture, enhancing anaerobic conditions within the soil. Barton and Schipper (2001) attempted to separate these effects, by measuring N<sub>2</sub>O emission from non-irrigated plots, water-irrigated plots, irrigated plots with mineral-N addition, and FDE-irrigated plots, yet obtained inconclusive results. Bhandral et al. (2007) found that water irrigation alone increased N<sub>2</sub>O emission above the non-irrigated treatment by 0.014 kg N ha<sup>-1</sup> in autumn and 0.029 kg N ha<sup>-1</sup> in winter, which showed that the anaerobic conditions created by irrigation alone enhance N<sub>2</sub>O production. The application of un-treated FDE by Bhandral et al. (2007) increased N<sub>2</sub>O emission above that of the water-only application by 0.24 kg N ha<sup>-1</sup> in autumn and 0.052 kg N ha<sup>-1</sup> in winter. Thus, Bhandral et al. (2007) illustrated that FDE irrigation increases N<sub>2</sub>O emission by both increasing soil moisture and providing a source of C and N to the soil microbial communities.

Table 5: N<sub>2</sub>O emission factors resulting from land application of FDE

Soil type	Soil moisture (% WFPS)	Total N applied (kg ha <sup>-1</sup> )	Duration of monitoring after application (d)	Emission factor (%)	Study
Peat	63 – 90	50	3	1.2 <sup>a) b)</sup>	Barton and Schipper (2001)
	35 – 66	50	3	0.15 <sup>a)</sup>	
Loamy silt	79 – 100	50	3	0.4 <sup>a)</sup>	
	44 – 85	50	3	0.25 <sup>a)</sup>	
Sandy loam	35 – 58	22	102	0.9 <sup>c)</sup>	Bhandral et al. (2007)
	35 – 58	61	102	0.4 <sup>d)</sup>	
	ca 47 – 86	13	56	0.2 <sup>c)</sup>	
	ca 47 – 86	49	56	0.2 <sup>d)</sup>	
Silt loam	30 – 53	50	10	0.03	Luo et al. (2008)
	26 – 38	50	4	0.01	
Silt loam	61 – 90	24	17	1.97	
	69 – 94	25	14	4.93	
	65 – 77	18	9	2.49	
Silt loam	ca 78 – 91	100	172	0.14 <sup>e)</sup>	Li et al. (2014b)
	ca 78 – 94	100	172	0.03 <sup>f)</sup>	

<sup>a)</sup> Not given by authors but estimated from their Figure 1, with estimated reading error of 30%.

<sup>b)</sup> Emissions trajectory had not fully returned to baseline after 3 d.

<sup>c)</sup> Treated FDE

<sup>d)</sup> Untreated FDE

<sup>e)</sup> Fresh FDE

<sup>f)</sup> Stored (4 month old) FDE

Seasonality appears to influence N<sub>2</sub>O emissions from land-applied FDE. Bhandral et al. (2007) applied FDE in both autumn and winter, and found the highest emission for treated and untreated FDE occurred in autumn, 2.0 and 0.7 kg N ha<sup>-1</sup>, respectively. Part of this seasonal variation was due to changes in soil moisture, known to be an important regulator of N<sub>2</sub>O emissions from soils (Luo et al. 2008). Emissions are highest when soils are saturated creating anaerobic conditions within the soil while soil temperatures are high enough to allow high rates of microbial respiration. This combination creates conditions that are conducive for denitrification and the production of N<sub>2</sub>O. The importance of soil moisture was also highlighted when rainfall events resulted in higher WFPS levels which increased N<sub>2</sub>O emissions (Bhandral et al. 2007; Luo et al. 2008), with Bhandral et al. (2007) noting a significant correlation between WFPS and N<sub>2</sub>O emission during both winter and autumn. Barton and Schipper (2001) also observed higher N<sub>2</sub>O emissions when the soil was wetter, both for an organic soil and a mineral soil (Table 5).



Dissolved organic carbon (DOC) content of the soil and N<sub>2</sub>O emissions were affected by the application rate of FDE with a range of C contents (Bhandral et al. 2007). Effluent application increased soil DOC concentration and caused significant increases in N<sub>2</sub>O emissions. However, application of untreated (fresh) FDE, which had the highest total C content, did not result in the highest N<sub>2</sub>O emissions. Bhandral et al. (2007) suggest that variation in C/N ratio and the availability of the C and N within the effluent affected both the denitrification rate and the N<sub>2</sub>O : (N<sub>2</sub>O + N<sub>2</sub>) ratio produced via denitrification following FDE application. Increased rates of denitrification do not necessarily equate to increased N<sub>2</sub>O emissions. The ratio of N<sub>2</sub>O:(N<sub>2</sub>O+N<sub>2</sub>) produced during denitrification is affected by many factors including the presence of NO<sub>3</sub><sup>-</sup>, the level of anoxia, pH, temperature, organic matter availability and microbial populations (Blackmer and Bremner 1978; Firestone et al. 1980; Firestone and Davidson 1989; Weier et al. 1993; van Cleemput 1998). Recently, Li et al. (2014b) calculated a significantly greater N<sub>2</sub>O emission factor for fresh FDE compared to stored FDE (4 months old), following application to a Waikato pasture in late winter (0.14 vs. 0.03%, respectively). Both FDE materials contained the same amount of total C; however, the fresh FDE had a higher NH<sub>4</sub><sup>+</sup> content and a greater pH (8.8 compared to 7.0 for the stored FDE). Consequently, these workers suggest the difference in the emission factor was due to greater nitrification rates in soils receiving the fresh FDE. Further research into how the availability of C influences denitrification rates is required.

### 5.3 NH<sub>3</sub> emissions

Three NZ studies have documented the production of NH<sub>3</sub> resulting from FDE application (Table 6). Di et al. (1998) measured NH<sub>3</sub> volatilisation for 2 weeks after FDE was applied to fine sandy loam. Only 0.05 to 0.3% of the total N applied in the FDE was lost as NH<sub>3</sub>, equivalent to 0.2 to 1.2% of TAN applied. Recently, Li et al. (2014a; b) measured NH<sub>3</sub> emissions from fresh and stored FDE applied to a Waikato pasture. Ammonia emissions ranged from 1.0 to 3.1% of total N applied as fresh FDE and 0.4 to 2.0% of total N applied as stored FDE. The observed difference in NH<sub>3</sub> emissions from fresh and stored FDE are considered to be due to the greater NH<sub>4</sub><sup>+</sup> content and higher pH of the fresh material (Li et al. 2014a; b). Furthermore, these authors note that NH<sub>3</sub> emissions were greater in the summer compared to spring and winter applications, suggesting the high temperatures decrease the solubility of NH<sub>3</sub>, resulting in greater soil-air exchange. Soil moisture content was also lower in summer, which produces a relatively high NH<sub>4</sub><sup>+</sup> concentration in the soil solution, leading to greater NH<sub>3</sub> volatilisation (Li et al. 2014a).

Owing to the limited available data on NH<sub>3</sub> emissions from FDE applied to land, we have included information on NH<sub>3</sub> emissions from land-applied slurries with a low DM content (Table 7). Misselbrook et al. (2000) suggest that dilute slurries would have lower NH<sub>3</sub> emissions than more concentrated slurries, due to their lower DM content which allows faster infiltration rates and lower TAN contents. The differences between FDE and dilute slurries were documented by Defra (2010) who reported the typical DM and N contents of dirty water (their term for FDE) as 0.5% and 0.5 kg N m<sup>-3</sup>, respectively, while cattle slurries contained 6% DM and 2.6 kg N m<sup>-3</sup>. However, no direct comparisons of NH<sub>3</sub> emissions from land-applied FDE and dilute slurries have been made. Linear relationships between slurry DM content and NH<sub>3</sub> emissions have been noted repeatedly (Figure 7).

**Table 6: NH<sub>3</sub> emission factors resulting from FDE applied to pasture surfaces.** TN = total N applied, TAN = total ammoniacal N.

FDE type	DM content (%)	pH	Duration after application (d)	TN (mg N L <sup>-1</sup> )	TAN (mg N L <sup>-1</sup> )	NH <sub>3</sub> emissions (% of TN)	Study
Fresh	not reported	7.9 – 8.8	14	138 – 662	37 – 162	0.05 – 0.3	Di et al. (1998)
Fresh	1.6	8.7	27	1100	450	1.0	Li et al. (2014b)
Store d	3.1	7.2	27	1100	120	0.4	
Fresh	1.4	8.7	27	1200	400	1.9	Li et al. (2014a)
Store d	2.0	7.0	27	700	100	0.6	
Fresh	1.8	7.4	21	700	400	3.1	
Store d	2.5	7.2	21	700	100	2.0	
Fresh	2.8	7.7	26	1100	300	2.5	
Store d	3.8	7.2	26	1000	100	2.0	

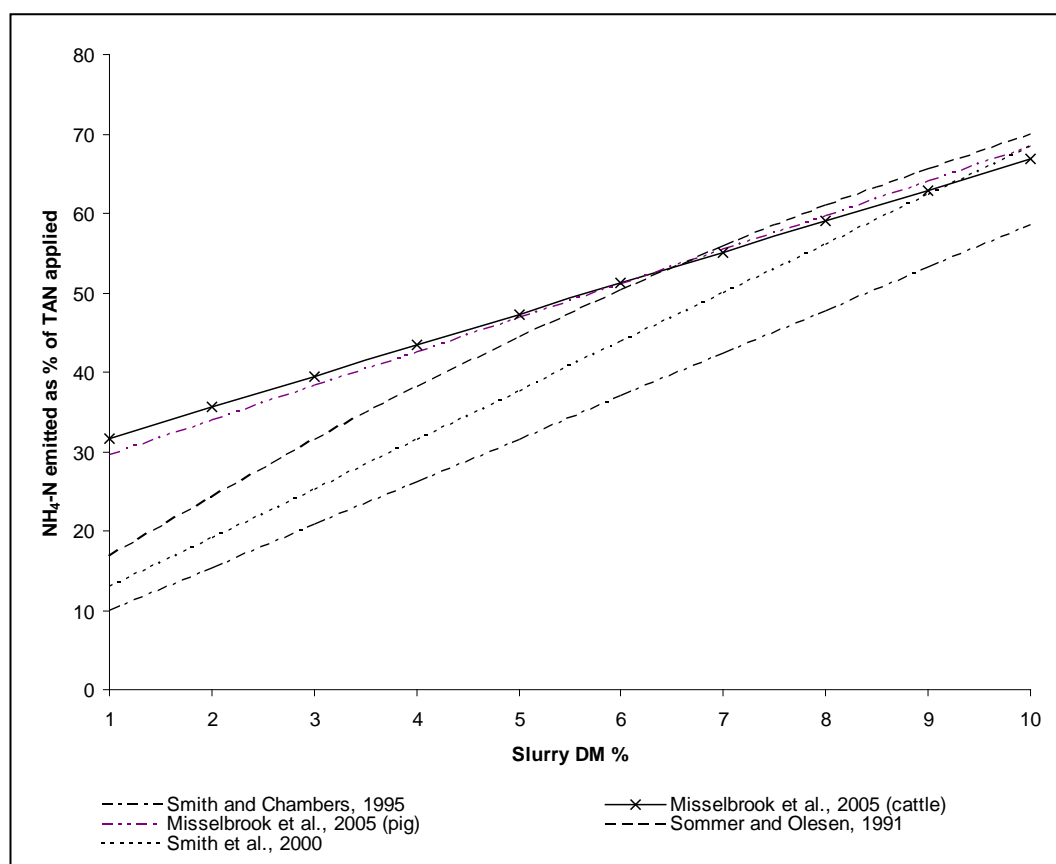


Figure 7: Comparison of simple models estimating the isolated effect of slurry DM content on  $\text{NH}_3$  emissions, following broadcast surface application (splash plate). See also Tables 6 and 7. (From: online report [www.clrtap-tfrn.org/webfm\\_send/197](http://www.clrtap-tfrn.org/webfm_send/197))

For slurries with higher DM contents, Sommer and Olesen (1991) reported that this linear relationship broke down 6 h after land application. This altered relationship between slurry DM content and  $\text{NH}_3$  emissions is most likely due to changes in pH (Sommer and Olesen 1991). Increasing DM content led to a declining pH, resulting in reduced  $\text{NH}_3$  emissions (expressed as % of TAN; Sommer and Olesen 1991). This inverse relationship between DM content and pH was shown by Misselbrook et al. (2005) for pig slurries (DM content ranging from 1.2 to 12.6%) applied to grassland and arable land. These authors found this relation was stronger for the more fibrous cattle slurries than for “gravelly” swine slurries, suggesting the former results in more soil pores being blocked, exposing TAN to the atmosphere for a longer duration. While FDE may have a fibrous solid fraction, the low solid content is likely to minimise any pH effect on its  $\text{NH}_3$  emissions.

Infiltration rates of FDE or slurries into soil are also affected by soil conditions such as soil moisture and temperature. Beauchamp et al. (1982) stated that rainfall events reduced  $\text{NH}_3$  emission from land-applied dairy cow liquids by increasing the infiltration of the effluent into soil. However rain events were also accompanied by reductions in temperature, which is expected to reduce  $\text{NH}_3$  emission, making it difficult to separate their effects. Temperature has been noted by several authors to have a strong effect upon the emission of  $\text{NH}_3$  from land-applied slurries (Beauchamp et al. 1982; Sommer and Olesen 1991). However, when applying slurries (DM content ranging from 1.2 to 12.6%) to grasslands or arable land, Misselbrook et al. (2005) were unable to find a relationship between temperature and total  $\text{NH}_3$  emissions. Their measurements were made using a series of wind tunnels moved to 10 sites across the UK, where different times of the year were used to test a range of temperatures. It was suggested other factors such as soil moisture, relative humidity and solar radiation had a greater influence on  $\text{NH}_3$  emissions. A positive relationship between solar radiation and  $\text{NH}_3$  emission was found for a liquid cattle manure that was applied to a grassland (Braschkat et al. 1997). The effects of soil moisture, temperature or solar radiation on  $\text{NH}_3$  emissions from land-applied FDE have not been studied but are required to gain a better understanding of the consequences of FDE application to land.

In summary, the lower  $\text{NH}_3$  generated by FDE application when compared with applied slurries is the result of FDE’s lower N and TAN content in conjunction with its lower solid content. However, emission of  $\text{NH}_3$  from applied FDE will still be affected by its rate of infiltration into the soil, and therefore soil moisture, and by climatic conditions such as wind speed and air temperature.

**Table 7: NH<sub>3</sub> emission factors resulting from land application of dilute manure slurries with differing physical and chemical characteristics**

Manure type	DM content (%)	pH	Temperature (°C)	Duration after application (d)	Application method	Total N (mg N L <sup>-1</sup> )	TAN (mg N L <sup>-1</sup> )	NH <sub>3</sub> emissions (% of TAN)	Study
Cattle slurry	0.9	7.1	9.7	6	Surface	2900	2500	19	Sommer and Olesen (1991)
Cattle slurry	0.9				Surface	2900	2500	26.3	
Cattle slurry	2.8				Surface	3900	2700	39.9	
Dairy slurry	1.6	-	14.3	0.42	Surface	2200	1060	19.6 (morning appl.), 5.9 (evening appl.)	Gordon et al. (2001)
Dairy slurry	2.6	-	8.9	0.42	Surface	2800	1800	1.8 (morning appl.), 1.5 (evening appl.)	
Dairy slurry	3.4	7.3		1	Splash plate	1800	1000	28.8	Smith et al. (2000)
Dairy slurry	3.6	7.4		1	Splash plate	1700	1000	12.4	
Dairy slurry	2.5	7.4		1	Splash plate	1600	1000	6.9	
Dairy slurry	2.0	7.5		1	Splash plate	1500	1000	3	
Dairy slurry	2.0	7.2		1	Splash plate	1400	900	5.9	
Dairy slurry	1.9	6.7		1	Splash plate	1100	600	9.4	
Dairy slurry	2.1	7.2		1	Splash plate	1100	800	3.8	
Dairy slurry	2.4	7.6		1	Splash plate	1000	400	3.8	
Pig slurry	3.0	7.9	6	4	Hand-broadcast	3000	2300	11.4	Dinuccio et al. (2012)
Pig slurry	4.0	7.8	21	4	Hand-broadcast	3600	2600	28.0	
Pig slurry liquid after solids separation	1.1	8.1	6	4	Hand-broadcast	1800	1500	8.8	
Pig slurry liquid after solids separation	1.3	8.1	21	4	Hand-broadcast	2100	1900	23.8	

## 5.4 Nitrogen leaching

Leaching of N from soils after the application of FDE varies between different soil types and is largely related to their hydraulic properties. Soils that allow preferential flow paths, such as well-structured soils or soils with tile drains can leach large amounts of N after FDE application when compared with poorly drained soils. In a review of nitrate leaching losses from different agro-systems Di and Cameron (2002b) note that losses are generally lower in fine textured soils when compared with courser soil texture due to slower drainage and greater opportunity for denitrification to occur.

Soil moisture levels at the time of application, as well as rainfall events soon after application, have a large effect on N leaching from land applications of FDE. Many studies have noted larger leaching losses of nitrate when FDE is applied to wet soils (Cooke et al. 1979; Macgregor et al. 1979; Di et al. 1998; Houlbrooke et al. 2008) due to reduced interactions between the soil and the FDE which reduces N retention within the soil. A good example of how soil moisture condition affects N leaching losses was provided by Houlbrooke et al. (2008), who compared N losses from water only, non-deferred, and deferred irrigation. In the deferred irrigation treatment, the same volume of water or FDE was applied as in the other treatments, but only after the soils had reached a state of soil moisture deficiency, whereas in the other treatments irrigation was applied regardless of soil moisture conditions. Similar nitrate leaching losses of 31.4 and 31.1 kg N ha<sup>-1</sup> occurred from 'water-only' irrigated plots and deferred FDE applications, respectively. When these losses are compared to the 36.7 kg N ha<sup>-1</sup> found when non-deferred irrigation was used, the importance of antecedent soil moisture is highlighted.

It may be expected that N leaching increases with increasing rates of N applied as FDE. However, this is not always the case. For instance, Roach et al. (2001) applied FDE at different N loading rates of 0, 100, 200 and 400 kg N ha<sup>-1</sup> and found leaching losses of approximately 19, 18, 20 and 48 kg N ha<sup>-1</sup> yr<sup>-1</sup>, respectively, where losses at the highest application rate were significantly greater ( $P < 0.05$ ) than those from lower rates. When these data were corrected for losses measured from the control treatment, N leaching from the application of FDE at  $\leq 200$  kg N ha<sup>-1</sup> was virtually zero. In contrast, applying FDE at 400 kg N ha<sup>-1</sup> resulted in 7.3% of N being leached. Other authors (Silva et al. 1999; Di and Cameron 2002a) applied FDE at 200 and 400 kg N ha<sup>-1</sup> and did not observe significant differences in the relative N losses from leaching. Silva et al. (1999) applied FDE at 200 and 400 kg N ha<sup>-1</sup> to a free-draining fine sandy loam soil and measured N leaching losses of 1.6 and 1.7% of applied N, respectively, after correcting the data for losses measured from control plots. Di and Cameron (2002a) flood-applied FDE at N loadings of 200 and 400 kg N ha<sup>-1</sup> to a shallow soil over two years and observed N leaching losses of 3.8% and 4.7%, respectively, in the second year. Losses in the first year were much greater, at 27.5% and 19.6% of applied N, but these occurred only two months after pasture renovation. It is highly likely that the pasture renovation released mineral N which was then available for N leaching (Di and Cameron 2002a). The losses measured in the second year were similar to those found by Di et al. (1998), who applied FDE at a loading rate of 400 kg N ha<sup>-1</sup> with two applications of 200 kg N ha<sup>-1</sup> and reported N leaching losses of 2.5 to 3.7% of the N applied. Taken together, and excluding the special case of pasture renovation, the NZ studies suggest that leaching losses from FDE irrigation are most likely of order 1 to 5% of applied N.

## 6 Implications for NZ's greenhouse gas inventory

By international agreement, national GHG inventories submitted until 2014 must follow the guidelines of IPCC (1996). Inventories to be submitted from 2015 onwards are to apply the revised guidelines of IPCC (2006). For CH<sub>4</sub> emissions from FDE ponds, NZ applies a country-specific method different to the default methods in the IPCC guidelines; it is argued in this section that for the future the Tier 2 method of IPCC (2006) should be considered instead. For N<sub>2</sub>O emissions from manure management, the 1996 and 2006 IPCC guidelines differ in that the 1996 version requires only the reporting of direct N<sub>2</sub>O emissions, while the 2006 version considers these negligible and requires the reporting of indirect N<sub>2</sub>O emissions instead. NZ has so far followed the IPCC 1996 default method and should therefore be prepared to the impending change in guidelines.

### 6.1 Current inventory practice for CH<sub>4</sub>

#### 6.1.1 NZ's inventory equation for CH<sub>4</sub> emissions from FDE ponds

In NZ's GHG inventory, CH<sub>4</sub> emissions from FDE ponds are computed according to Equation 1. The terms on the right-hand side of the equation can be grouped as two products, with the following meanings. The first product ( $FDM \times MMS$ ) is the amount of the faecal dry matter anaerobically digested in ponds during the time period assessed. The second ( $W/1000/d \times Y_m$ ) is supposed to describe the yield of CH<sub>4</sub> emitted per unit of FDM, accounting for the volume of the ponds. These two products and their data sources are discussed in the following two subsections, respectively.

Equation 1: NZ's GHG inventory equation for CH<sub>4</sub> emissions from effluent ponds.

$$M = FDM \times MMS \times W/1000/d \times Y_m$$

where

$M$  = CH<sub>4</sub> emission from FDE ponds (kg CH<sub>4</sub>/year)

$FDM$  = faecal dry matter generated (kg FDM/year)

$MMS$  = proportion of faecal material deposited in ponds (kg FDM treated/kg FDM generated)

$W$  = water dilution rate (90 L wastewater treated/kg FDM)

$d$  = average depth of a pond (4.6 m)

$Y_m$  = mass of emitted CH<sub>4</sub> per anaerobic pond area per year (3.27 kg CH<sub>4</sub>/m<sup>2</sup>/year)

### 6.1.2 Amount of faecal dry matter anaerobically digested in FDE ponds

In Equation 1, the amount of faecal dry matter (FDM) anaerobically digested in FDE ponds is expressed as the product of the total amount of FDM generated during the period assessed (in kg DM/year) and the fraction of this FDM actually treated in anaerobic ponds (*MMS*, dimensionless). The total amount of FDM generated is calculated on the basis of dietary requirement, using a methodology validated for enteric emissions. The current inventory practice is documented as using the FDM produced by the total dairy cattle population (e.g.  $5.861 \times 10^6$  head in 2009) and assuming 5% of that to be treated in FDE ponds, i.e. *MMS* = 0.05). However, since only cattle in milk actually deposit manure at the milking shed, it would be more accurate to base the FDM estimation on the *lactating* cow population, for which data are available from Statistics NZ (MAF 2011, Appendix 6; e.g.  $4.348 \times 10^6$  head in 2009). The Ministry of Primary Industries (MPI) does so internally and uses *MMS* = 0.06 as the fraction treated in ponds (Chung et al. 2013). The two estimates (5% of total dairy herd's manure vs. 6% of manure from cows in milk only) are roughly consistent because the portion of dairy cows in milk is approximately 80% of the total dairy population (MAF 2011, Appendix 6). Both are based on an estimate of 8% for the manure fraction deposited at the milking shed over a milking season of 270 days (Ledgard and Brier 2004).

There are two potential causes for bias in these estimates. The first is that the 10-year-old study by Ledgard and Brier (2004) may not accurately represent current management practices any more. Larger herd sizes and practices of milking two or three times daily are likely to have led to an increase in the time fraction that the cows in milk spend on the milking platform, with a resulting increase in the manure fraction deposited there. Pratt et al. (2012) inferred from their year-long observations of FDE pond emissions of CH<sub>4</sub>, at two farms not using feed pads, that the annual VS inputs to the ponds would have been about 10% of the total VS produced by the herds. Saggart et al. (2004) already assumed that 10 to 20% of daily discharge occurs on the milking platform throughout a milking season of 270 days<sup>1</sup>. The upper end of this range would be realistic for very large herds that are milked twice daily. On an annual basis, these figures would convert to a range of 0.07 to 0.15 for *MMS*, bracketing the value of 0.10 estimated by Pratt et al. (2012).

The other cause for bias is the seasonal pattern of FDM generation. The FDM generation rate in the winter months is considerably less than throughout the milking season (Table 8). This is so because the energy demand (and thereby the DMI and FDM per cow) of dairy cows is higher during lactation than during the “dry” winter months. In Table 8, monthly population statistics are used to compute the monthly FDM generation of NZ's lactating cow population, and *MMS* = 0.06 is used to estimate the amount of FDM treated for each month, following current MPI practice. (Equation 1 is then applied on a monthly basis to estimate CH<sub>4</sub> emissions from ponds for the year 2009; the annual total represents the current inventory figure.) If, by contrast, one applied a collection rate of 8% for the 9 months of July to March

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<sup>1</sup> With these assumptions, Saggart et al. (2004) follow a report by Heatley, P. (2001): *Dairying and the environment: managing farm dairy effluent*. Palmerston North, New Zealand Dairy Research Institute.

The lower and upper values of 10 and 20% of daily discharge are realistic for once-daily and twice-daily milking schedules, respectively, with large herds (over 300 cows), provided that the time for which the cows queue up on the platform is of the order of 2 to 3 h per milking.

Luo et al. (2013), p. 86, employed a collection fraction of 15% in the additional scenario of their life-cycle assessments, on the basis of the draft version of the present report.

and of 0% for April to June, to correctly reflect the seasonality of the milking activity, then the total amount of FDM anaerobically digested would be obtained as 274,643,803 kg FDM. This is 17% higher than the current inventory estimate of 234,709,701 kg, which is based on the simplified assumption of 6% collection across the entire year.

Hence, adequate specification of the population basis for FDM inputs, the length of the milking season, and crucially, of the factor *MMS*, is important to arrive at accurate inventory estimates. As demonstrated, it is likely that the currently used value is too small, which potentially causes a significant underestimation of the amount of manure anaerobically treated in FDE ponds. A more accurate quantification should be prioritised.

Table 8: Population statistics for lactating cows in 2009, and calculations used in NZ's GHG inventory. Data from MPI, table adapted from Pratt et al. (2014a).

Month	<i>DMI</i> <sup>1)</sup> (kg DM head <sup>-1</sup> )	Population (head)	<i>DMD</i> <sup>2)</sup>	FDM anaerobically treated <sup>3)</sup> (kg DM)	<i>M</i> <sup>4)</sup> (Gg) <sup>5)</sup>
Jan	358	4,347,656	0.762	22,243,814	1.423
Feb	326	4,347,656	0.736	22,466,720	1.437
Mar	331	4,347,656	0.744	22,108,571	1.414
Apr	189	4,347,656	0.786	10,523,121	0.673
May	160	4,606,970	0.812	8,335,435	0.533
Jun	180	4,606,970	0.802	9,868,292	0.631
Jul	439	4,782,412	0.837	20,569,651	1.316
Aug	427	4,606,970	0.795	24,265,605	1.552
Sep	402	4,606,970	0.791	23,281,128	1.489
Oct	381	4,606,970	0.805	20,546,949	1.315
Nov	366	4,606,970	0.785	21,741,196	1.391
Dec	397	4,606,970	0.738	28,759,218	1.840
<b>2009 total</b>	3,956	4,347,656 <sup>6)</sup>		234,709,701 <sup>6)</sup>	<b>15.014</b>

<sup>1)</sup> Dry-matter intake, based on the metabolisable energy and energy concentration of feed (MAF 2011).

<sup>2)</sup> Dry-matter digestibility (MAF 2011, Appendix 3).

<sup>3)</sup> Amount of faecal dry matter anaerobically digested =  $FDM \times MMS$ , where  $FDM = DMI \times (1 - DMD)$  (MfE 2013, p. 162), and proportion of faecal material deposited in ponds  $MMS = 0.06$  (MPI, Chung et al. 2013).

<sup>4)</sup> Amount of methane released from pond according to Equation 1, where  $FDM \times MMS$  from <sup>3)</sup>, dilution rate  $W = 90 \text{ L (kg FDM)}^{-1}$ , average pond depth  $d = 4.6 \text{ m}$ , and per-area  $\text{CH}_4$  emission rate  $Y_m = 3.27 \text{ kg m}^2 \text{ year}^{-1}$ .

<sup>5)</sup> The units Gg are reported in the inventory, but as can be seen from the units of the contributing factors, the actual units would be  $\text{Gg} \times \text{m}^3 \text{ effluent (m}^3 \text{ pond} \times \text{year)}^{-1}$ .

<sup>6)</sup> In the inventory, it is assumed that the January population size is representative for the entire year, and consequently the average FDM output is reported as  $900 \text{ kg FDM head}^{-1}$  and the average amount of FDM entering anaerobic ponds as  $53.99 \text{ kg FDM head}^{-1}$  (in 2009).



### 6.1.3 Yield of methane emitted per unit of FDM anaerobically digested

In the current inventory equation for CH<sub>4</sub> emissions from manure treated in anaerobic FDE ponds, it is assumed that the pond manure emits at a rate of  $Y_m = 3.27 \text{ kg CH}_4 \text{ m}^{-2} \text{ year}^{-1}$ , following dilution with  $W = 90 \text{ L}$  of water per kg of FDM and collection in a pond of depth  $d = 4.6 \text{ m}$  (MfE 2013). The  $Y_m$  value was derived from a single study (McGrath and Mason 2004) taking observations over 6 months at a single anaerobic FDE pond. The suitability of this value for inventory purposes is questionable because: 1) it is a mean of daily emission values covering a 20-fold range and not weighted for seasonality or management effects, 2) it was obtained by a visual bubble-counting method where the conversion of bubble numbers to biogas volume was inferred but not calibrated, and 3) the farm monitored in this study (Massey Dairy Farm #4, Palmerton North) is not representative of all New Zealand dairy farms in terms of management practice and FDE treatment system, see Section 3 of this report. The values for  $W$  and  $d$  are also questionable, as is the assumption of a uniform pond depth; these issues are discussed in Pratt et al. (2014a). The inventory methodology must thus be considerably improved to provide the necessary level of refinement and accuracy.

In addition, Equation 1 is mathematically erroneous due to a mistake in the conversion of the surface methane flux ( $Y_m$  in  $\text{kg CH}_4 \text{ m}^{-2} \text{ year}^{-1}$ ) into a yield (in  $\text{kg CH}_4$  per kg FDM treated). As a result, CH<sub>4</sub> emissions are currently predicted in  $\text{kg CH}_4 \text{ m}^3 \text{ wastewater m}^{-3} \text{ pond year}^{-2}$ , which does not make sense. A correct conversion factor (instead of  $W/1000/d$ ) should have been expressed as a retention time (in  $\text{year m}^3 \text{ pond m}^{-3} \text{ wastewater treated}$ ), with correct consideration of pond geometry (surface-to-volume ratio).

### 6.1.4 Other considerations regarding the inventory of CH<sub>4</sub> emissions from FDE

In compliance with IPCC guidelines, NZ's GHG inventory does not include potentially significant contributions to the dairy-farm manure management system, such as from manure deposited onto feed and stand-off pads, waste milk, and feed wastage on feed pads. Chung et al. (2013) estimated that these additional contributions may result in total CH<sub>4</sub> emissions potentially being 1.5 to 2 times larger than from manure collected at the milking parlour alone, see Section 6.4.5.

Care has to be exercised that emissions from these additional sources are accounted for in a way that reflects actual management practice. For example, where feed pads are simply a source of additional manure that is washed into FDE ponds, it is appropriate to account for this simply via the additional FDM treated in these ponds. However, where these waste streams are stored or pre-treated in a more solid form, their methane conversion factors may be substantially different from those for FDE. Data will be required on which types of waste-stream management practices are in use on NZ farms, and how widespread they are.

Collecting a larger fraction of manure on the farm leads to less manure deposited directly on pasture and will thus cause a reduction in pasture-based GHG emissions. However, this reduction is far too small to offset the increase in manure-based emissions in the case of CH<sub>4</sub>. This is because for the same amount of anaerobically biodegradable material, far more CH<sub>4</sub> is produced in anaerobic ponds: anaerobically, typically 70 to 80% of the manure's biochemical methane potential (BMP) is reached, while under predominantly aerobic pasture conditions only about 1% of the manure's BMP is realised, as quantified by the pasture emission factor of  $0.98 \text{ g CH}_4/\text{kg TS}$  used in the inventory.

## 6.2 Current inventory practice for N<sub>2</sub>O and NH<sub>3</sub>

Nitrous oxide emissions from dairy farming manure management, and as a consequence the majority of NH<sub>3</sub> emissions that contribute to indirect N<sub>2</sub>O emissions, are recorded in NZ's GHG inventory at several levels (MfE 2013). The method of estimating these emissions is to account for the amount of excreted manure solids, and consequently excreted amounts of N, to partition these amounts into treatment pathways, and to apply emission factors (of the form amount of N lost per amount of N present) to each treatment step (Figure 8).

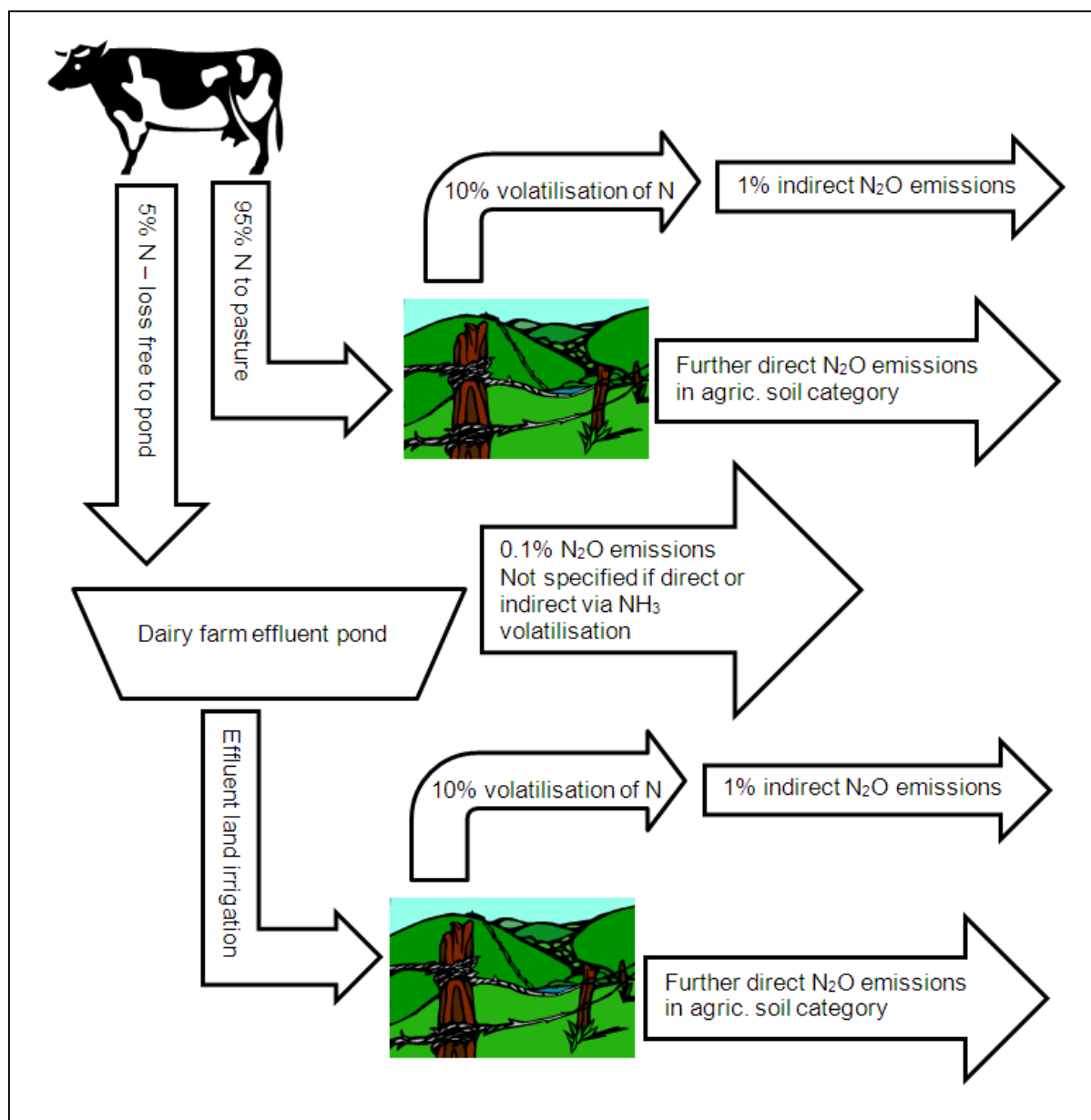


Figure 8: Schematic illustration of NH<sub>3</sub> and N<sub>2</sub>O loss pathways in NZ dairy manure management according to the current inventory methodology.

According to the inventory documentation (MfE 2013), it is assumed that 95% of the manure produced by all dairy cattle is deposited onto pasture and 5% of manure is treated in anaerobic ponds, before being land-applied by irrigation. As explained in Section 6.1.2, there is considerable uncertainty about these fractions (as well as variability at the farm level), and it is likely that in fact more than 5% of manure is handled in ponds. A more accurate assessment of these fractions would therefore improve the accuracy of the computation of  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions. Unlike in the case of  $\text{CH}_4$ , where a higher fraction of manure treated in ponds leads to higher emissions, this is not necessarily true in the case of  $\text{N}_2\text{O}$ : the net result will depend on the whole combination of partition fractions and emission factors applied throughout the treatment chain, including direct and indirect emissions.

In the current inventory, manure deposited onto pasture is assumed to volatilise 10% of the original nitrogen (as  $\text{NH}_3$  or  $\text{NO}_x$ ), which contributes to indirect  $\text{N}_2\text{O}$  emissions (atmospheric deposition) with a factor of 1% ( $\text{kg N}_2\text{O-N}$  per  $\text{kg}$  volatilised N). The remaining N deposited onto pasture is then assumed to contribute to further direct  $\text{N}_2\text{O}$  emissions, which are reported under the separate agricultural soils category (MfE 2013). This N also gives rise to leaching losses, currently assumed as 7% of applied N, of which 2.5% are assumed to account for additional indirect  $\text{N}_2\text{O}$  emissions (this pathway is omitted from Figure 8 for simplicity).

Currently, manure deposited in the milking yard is assumed to be transferred to the anaerobic pond without any  $\text{NH}_3$  volatilisation losses. The anaerobic pond is then assumed to contribute to  $\text{N}_2\text{O}$  emissions with a factor of 0.1% of the N entering the anaerobic pond. These emissions are thus defined as if they were *direct*  $\text{N}_2\text{O}$  emissions, while no indirect emissions are assumed to exist. This practice is in full compliance with the IPCC 1996 guidelines, which were to be applied to GHG inventories submitted until 2014. However, this practice appears inconsistent with the inventory treatment of FDE application (irrigation) to land, which is assumed to contribute to  $\text{NH}_3$  volatilisation losses of 10% and, from these, *indirect*  $\text{N}_2\text{O}$  emissions of 1%, similar to emission rates of manure directly deposited onto pasture (MfE 2013). Anaerobic-pond effluent applied to land is also assumed to contribute to direct  $\text{N}_2\text{O}$  emissions, which are again reported under the separate agricultural soils category.

It is clear from Figure 8 that as long as  $\text{NH}_3$  volatilisation losses occurring before land application are neglected (i.e. those from yards, pre-treatment systems and ponds), the only difference between the pasture pathway and the FDE pond pathway is the minor loss of 0.1% N as  $\text{N}_2\text{O}$  from ponds. In other words, 99.9% of all excreted N is assumed to be returned to pasture, where the emission factors for direct and indirect emissions are the same, regardless of whether the N originates from direct excretion by cattle or from FDE irrigation. This allows for a very simple estimation of the relative importance of the  $\text{N}_2\text{O}$  emissions from FDE, as currently reported: it is simply about 5% of the total  $\text{N}_2\text{O}$  emissions from dairy cattle excreta. However, by explicitly accounting for the different pathways, the inventory has already the potential to obtain improved estimates that would include  $\text{NH}_3$  and  $\text{N}_2\text{O}$  losses from milking platform, pre-treatment systems and ponds with individual emission factors. A requirement for implementing such an improved approach would be to quantify the amount of N present in each treatment step as the amount present in the previous treatment step minus the gaseous losses that occurred during that step, i.e. a full mass budget for N.

To assess the relative importance of FDE pond emissions of  $\text{N}_2\text{O}$ , compared with those of  $\text{CH}_4$ , consider the values provided in the 2010 inventory (MfE 2012), for a total dairy cattle herd of  $5.915 \times 10^6$  head. Assuming each cow excreted  $116.5 \text{ kg N year}^{-1}$ , the amount of N excreted totalled  $6.891 \times 10^8 \text{ kg N year}^{-1}$ . Of this,  $3.447 \times 10^7 \text{ kg N year}^{-1}$  (5%) was treated in ponds. Using the IPCC default emission factor of  $0.001 \text{ kg N}_2\text{O-N kg}^{-1} \text{ N}$  for ponds and

lagoons (IPCC 2006), the resulting direct N<sub>2</sub>O emissions from ponds were  $5.42 \times 10^4$  kg N<sub>2</sub>O year<sup>-1</sup>. Multiplying this with the global-warming-potential (GWP) factor<sup>2</sup> (310 kg CO<sub>2</sub>-e kg<sup>-1</sup> N<sub>2</sub>O) yields GHG emissions of 16.8 Gg CO<sub>2</sub>-e year<sup>-1</sup>. By comparison, the inventory reports 16.2 Gg CH<sub>4</sub> emissions from ponds for 2010. Using a GWP factor<sup>2</sup> of 21 kg CO<sub>2</sub>-e kg<sup>-1</sup> CH<sub>4</sub>, this converts to GHG emissions of 340 Gg CO<sub>2</sub>-e year<sup>-1</sup>. The relative contribution of N<sub>2</sub>O to the total FDE pond emissions is thus minor (of the order of 5% if the CH<sub>4</sub> values were accurate, and even less if CH<sub>4</sub> values were underestimated, as is argued in the following sections).

The contribution of NH<sub>3</sub> emissions from ponds, currently ignored in the inventory (following international practice), may well be of similar magnitude as that of the direct N<sub>2</sub>O emissions. The IPCC 2006 guidelines, to be used for future inventories, remove direct N<sub>2</sub>O emissions from manure management systems (considered negligible) but include indirect N<sub>2</sub>O emissions. As discussed in Section 4.3.3, there is at present a lack of data to specify the NH<sub>3</sub> emission factor for pond conditions in NZ, but the overseas data suggest it could quite possibly be of the order of 10% of the total N contents. The IPCC default factor for anaerobic lagoon is even higher, at 35%. If such high volatilisation losses were confirmed, then the indirect N<sub>2</sub>O emissions from ponds, taken as 1% of the volatilised NH<sub>3</sub>-N, would be comparable to or even in excess of the direct N<sub>2</sub>O emissions.

The relative importance of NH<sub>3</sub> may rise further if volatilisation at the milking platform is included (Section 4.3.1), while the relative importance of both N<sub>2</sub>O and NH<sub>3</sub> may rise if emissions from solid separators and weeping walls are included in the inventory (Sections 4.2.2 and 4.3.2).

### 6.3 Shortcomings of the current inventory values and methodology

The current methodologies in NZ's GHG inventory have a number of shortcomings that should be rectified to better reflect the actual farming situations, improve inventory accuracy and support the use of abatement measures to reduce manure-management GHG emissions. Both the CH<sub>4</sub> and N<sub>2</sub>O datasets are inaccurate, firstly, in their methodological rigour and appropriateness of emission factors:

- CH<sub>4</sub> emissions from ponds are currently estimated using a mathematically erroneous equation and without consideration to pond geometry.
- Few emission factors appear to be based on NZ studies, in particular for direct and indirect N<sub>2</sub>O emissions (the latter via NH<sub>3</sub> volatilisation).

Secondly, they do not reflect the field situation well in the following respects:

- The CH<sub>4</sub> methodology probably underestimates the fraction of manure that is actually handled by manure management systems, both from collection at the milking platform

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<sup>2</sup> The quoted GWP factors for N<sub>2</sub>O and CH<sub>4</sub> reflect current inventory practice. They are based on a 100-year timeline and originate from the IPCC's Second Assessment Report (1996). From 2015 onwards, revised 100-year GWP factors from the IPCC's Fourth Assessment Report (2007) are to be used in national inventories. For N<sub>2</sub>O, this revised factor, at 298 kg CO<sub>2</sub>-e kg<sup>-1</sup> N<sub>2</sub>O, is smaller than the current value. For CH<sub>4</sub>, the revised factor, at 25 kg CO<sub>2</sub>-e kg<sup>-1</sup> CH<sub>4</sub>, is larger than the current value. Using the revised GWP factors, direct N<sub>2</sub>O emissions from ponds accounted for 16.1 Gg CO<sub>2</sub>-e year<sup>-1</sup> in 2010, while CH<sub>4</sub> emissions from ponds accounted for 405 Gg CO<sub>2</sub>-e year<sup>-1</sup>.

and from increasing fractions of manure being collected at stand-off pads and feed pads (Chung et al. 2013).

- The CH<sub>4</sub> methodology is based on the assumption that a fixed fraction of the manure produced is treated in anaerobic ponds, not on actual pond size statistics. Consequently, emissions from increasing numbers of FDE storage ponds are not included.
- The impacts of different management practices, including pre-treatment technologies (which may alter C and N contents, pH or aeration of the handled FDE), on subsequent emissions of CH<sub>4</sub>, N<sub>2</sub>O and NH<sub>3</sub> are not considered. IPCC guidelines do not require the distinction of these practices, but actual emissions may strongly depend on these.

The agricultural GHG emissions reported in NZ's inventory include all emission pathways that are required under international practice (IPCC 1996). It should be noted that this practice ignores a number of potentially significant pathways, namely:

- The current methodology completely omits NH<sub>3</sub> volatilisation losses from the milking parlour and holding yard.
- Direct N<sub>2</sub>O and CH<sub>4</sub> emissions from pre-treatment (solids separator, weeping wall) are not accounted for as there is currently a lack of data to determine suitable emission factors for these systems.
- There are no specific data on non-manure wastes (e.g. milk and supplementary feed) to compute the contribution of these organics to GHG emissions from manure treatment systems.

The following two sections describe alternative methodologies and accounting systems to better record manure management emissions from NZ dairy farms.

## 6.4 Alternative inventory approaches for CH<sub>4</sub>

Pratt et al. (2012) reported CH<sub>4</sub> fluxes measured from the anaerobic effluent ponds on three dairy farms across varying climatic regions (Northland, Waikato and Southland). They also determined the number of farms in NZ with anaerobic ponds by information obtained from regional councils and DairyNZ. It was thus estimated that the milking shed effluent from approximately 73.5% of dairy cows was handled through effluent ponds, the other 26.5% being directly irrigated to pasture upon collection at the milking shed. Pratt et al. (2012) then estimated the nationwide CH<sub>4</sub> emissions from FDE ponds by combining the surface CH<sub>4</sub> flux data with the proportion of milking shed effluent discharged to ponds (effluent irrigated directly to pasture will produce negligible CH<sub>4</sub> emissions). The results revealed that these CH<sub>4</sub> emissions were approximately 1000 Gg CO<sub>2</sub>-e/year in 2010, almost 3 times higher than the estimate given by NZ's GHG inventory. Pratt et al. (2014a) show that the low-emission estimate in the inventory is partly due to mathematical errors, as explained in Section 6.1.3. Two important questions are therefore: Can the current methodology be mathematically corrected? Can it be improved to incorporate changes and variations in farm operation? In the next three subsections, options for improved inventory equations are discussed with regards to these questions. Then, the recommended option is applied to arrive at a new inventory estimate for CH<sub>4</sub> emissions from FDE ponds. In a final subsection, model calculations for two scenarios are presented to account for substrates treated in FDE ponds that do not originate from manure deposited at the milking shed.

#### 6.4.1 Correcting the current inventory equation for CH<sub>4</sub> emissions from FDE

In principle, it is straightforward to replace the erroneous conversion factor in Equation 1 ( $W/1000/d$ ) by a correctly-defined one, see Section 6.1.3. However, closer scrutiny of Equation 1 shows that the nationwide CH<sub>4</sub> yield from ponds is simply a function of cow numbers (as  $W$  is a function of  $FDM$  and  $FDM$  is a function of the nationwide herd size). A simpler alternative is therefore to determine emissions directly from a cow-population basis using the pond surface flux data. Still, such an approach would be based solely on the monitoring of a few farms regarded as typical for all NZ farms (e.g. all farms manage the same fractions of waste, all wastes have the same BMP, all treatment systems cause the same fraction of the waste BMP to be emitted). Hence, neither can it be considered as representative nor can it be used to account for temporal changes (e.g. the CH<sub>4</sub> yield per cow may change over time). To account for such changes, a methodology with dynamic inputs is required, such as the currently used FDM input based on cow weight, productivity and diet.

#### 6.4.2 IPCC Tier 2 equation for CH<sub>4</sub> emissions from manure management

The IPCC's Tier 2 methodology for CH<sub>4</sub> emissions from manure management is outlined on p. 10.41 of the emissions from livestock and manure management document (IPCC 2006). There, the CH<sub>4</sub> emission factor is estimated as:

**EQUATION 10.23**  
**CH<sub>4</sub> EMISSION FACTOR FROM MANURE MANAGEMENT**

$$EF_{(T)} = (VS_{(T)} \cdot 365) \cdot \left[ B_{o(T)} \cdot 0.67 \text{ kg / m}^3 \cdot \sum_{S,k} \frac{MCF_{S,k}}{100} \cdot MS_{(T,S,k)} \right]$$

where

$EF_{(T)}$  = annual CH<sub>4</sub> emission factor for livestock category  $T$  (kg CH<sub>4</sub> head<sup>-1</sup> year<sup>-1</sup>)

$VS_{(T)}$  = daily volatile solids excreted for livestock category  $T$  (kg dry matter head<sup>-1</sup> d<sup>-1</sup>)

365 = basis for calculating annual VS production (d year<sup>-1</sup>)

$B_{o(T)}$  = maximum CH<sub>4</sub>-producing capacity for manure produced by livestock category  $T$  (m<sup>3</sup> CH<sub>4</sub> kg<sup>-1</sup> VS excreted). From: *Volume 4: Agriculture, Forestry and Other Land Use*, p. 10.42 of 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

$MCF_{S,k}$  = CH<sub>4</sub> conversion factors for manure management system  $S$  by climate region  $k$  (%)

$MS_{(T,S,k)}$  = fraction of livestock category  $T$ 's manure handled using manure management system  $S$  in climate region  $k$  (dimensionless)

As can be seen, the IPCC's Tier 2 methodology is conceptually similar to the current NZ methodology (Equation 1) in that CH<sub>4</sub> emissions are calculated based on the amount of solids treated in ponds ( $VS \times 365 \times MS$  in the IPCC's Tier 2 methodology, versus  $FDM \times MMS$  in the NZ methodology), multiplied by a field methane yield ( $B_o \times 0.67 \times MCF/100$  in the IPCC's Tier 2 methodology, versus  $(W/1000/d) \times Y_m$  in the NZ methodology). Only the IPCC's equation, however, delivers results in the correct units.

An advantage of the IPCC's Tier 2 methodology is that it can account for differences in treatment systems ( $MCF$ ), operation ( $MS$ ), and climate ( $k$ ); these parameters can be modified with time to reflect changes in practice. Its main disadvantage is that it requires considerable amounts of inputs, some of which are currently unavailable. A particular concern is that the default  $MCF$  values provided by the IPCC are likely erroneous (Lory et al. 2010).

Chung et al. (2013) applied the IPCC equation for the year 2009, for NZ's lactating dairy cattle herd (livestock category  $T = DC$ ). They varied the factors  $VS_{(DC)}$  and  $MCF_{(DC)}$  over some ranges around the IPCC default values, to account for their uncertainty, while  $B_{o(DC)}$  was taken as the IPCC default value. For  $MS_{(DC)}$  they used MPI's value of 0.06. With that approach, Chung et al. (2013) predicted annual  $CH_4$  emissions of 27.6 to 43.7 Gg  $CH_4$  (equivalent to 579 to 918 Gg  $CO_2$ -e based on a GWP factor of 21 kg  $CO_2$ -e/kg  $CH_4$ ). This is 1.84 to 2.91 times larger than the current inventory value of 15.01 Gg  $CH_4$  year<sup>-1</sup> (Table 8), confirming that there is a need for revision of the inventory methodology.

#### 6.4.3 NZ-specific adaptation of the IPCC Tier 2 equation for $CH_4$ emissions from FDE ponds

The IPCC provides default values but generally recommends the use of country-specific factors. It should be noted that the IPCC does not consider emissions from milk and supplementary feed wastes. This represents a critical area for improvement both nationally and internationally. In the following, it is suggested how each factor of the Tier 2 equation from IPCC (2006) can be adapted for FDE from NZ's dairy farms, to arrive at an improved equation that could be implemented in the inventory with data already available (Equation 2).

With regards to manure production, the default value for a cow of 500 kg weight in Oceania is given as 3.47 kg VS head<sup>-1</sup> d<sup>-1</sup> (IPCC 1996) or 3.5 kg VS head<sup>-1</sup> d<sup>-1</sup> (IPCC 2006). Adjusting the latter value for a NZ-average dairy cow of 459 kg gives 3.21 kg VS head<sup>-1</sup> d<sup>-1</sup>. The IPCC (2006) guidelines also provide a methodology for calculating VS output based on DM intake, digestibility and other parameters (applied, e.g. by Luo et al. (2013), to model GHG emissions from dairy farms with different winter practices). Alternatively, NZ's GHG inventory already provides FDM data accounting for monthly variations in the number of cows milked and their dietary requirements (Table 8). If, as recommended by Dämmgen et al. (2011), the contribution of urine as a substrate of methane biosynthesis is disregarded, then the FDM data can be converted using a VS/FDM ratio of 0.82 (Alvarez et al. 2006), assuming faeces and manure have a similar ash content. The monthly VS figures for 2009 estimated with this approach are shown in Table 9. When converted to a daily time basis, they range from 0.80 to 2.75 kg VS head<sup>-1</sup> d<sup>-1</sup> (minimum in May, maximum in December) with an annual average of 1.94 kg VS head<sup>-1</sup> d<sup>-1</sup>, which is only 60% of the IPCC default value. Here, we do not discuss reasons for the discrepancy between IPCC and NZ values, but note that any error in the FDM input data would affect FDE pond emissions calculated on an actual VS basis in proportion to enteric  $CH_4$  emissions from dairy cattle computed elsewhere in the inventory.

By obtaining monthly VS input data, and by switching from an annual to a monthly calculation basis (as below), the factor 365 days/year in the IPCC equation becomes obsolete.

Since only the manure of lactating cows is managed in FDE ponds in NZ, the lactating population should be the basis for the inventory calculations (Chung et al. 2013), as is already current practice. However, since FDE is produced at the milking shed only while cows are milked, the milking season rather than the full year should be the basis for emission calculations. This can easily be implemented by applying the equation for the emission factor for dairy cows,  $EF_{(DC)}$ , on a monthly basis, multiplying it with the dairy cow population for

that month (Table 8) and then adding the results for all months of the milking season (where transition months between milking and “dry” seasons can be included with suitable time fraction factors). Manure from feed pads and stand-off pads is produced with a different seasonal pattern, is stored differently (Luo et al. 2013), and where it is treated in FDE ponds, it must therefore be accounted for separately.

With regards to the ultimate biochemical CH<sub>4</sub> potential ( $B_o$ ), data from the full-year record of representative farms across NZ by Pratt et al. (2012) are available. These suggest a value  $B_{o(DC)} = 0.147 \text{ kg CH}_4 \text{ kg}^{-1} \text{ VS added}$ , which is 8% lower than the IPCC default value. (This value already includes the conversion factor  $0.67 \text{ kg CH}_4 \text{ m}^{-3} \text{ CH}_4$ , which is thus obsolete.)

There are currently no  $MCF$  values associated with FDE treatment in NZ. Past and current monitoring efforts may provide the data necessary to calculate locally relevant  $MCF$  values, and predictive models could be applied to account for the impacts of climate, design and operation. Also, since the existing experimental data were all obtained using floating covers, which covered only a small fraction of the total pond area, it would be useful to corroborate them with methods integrating over the whole pond surface, e.g. micrometeorological methods (Park et al. 2010; VanderZaag et al. 2011; Laubach et al. 2013a). In the absence of NZ-specific data, IPCC defaults could be used. The default value in IPCC (1996) for “anaerobic lagoons”, in all climates, is 90%, but was criticised as incorrectly derived by Lory et al. (2010). These authors gave a range of 66 to 79% for climates across the USA. More in line with this range is the IPCC (2006) default value of 71% for an average annual temperature of 13°C. This value was used in the following calculations.

The factor  $MS$ , the fraction of total FDE generated and stored or treated in ponds, is variable and difficult to determine. For the below example calculation, a recovery factor of 15% of total waste at the cow shed was used (weighted estimate of farms with no feed pad, including twice-a-day and once-a-day milking), multiplied by a factor of 73.5% (Pratt et al. 2012), representing the weighted fraction of farms employing ponds, to arrive at  $MS_{(DC)} = 0.11$ .

These modifications result in Equation 2 for the annual national CH<sub>4</sub> emissions from FDE ponds,  $M_{(DC)}$ . As can be seen, expressing the  $EF$  values is not strictly necessary but provides data for discussing seasonal changes independently from population size.

#### Equation 2: Suggested GHG inventory equation for CH<sub>4</sub> emissions from NZ’s FDEponds

$$M_{(DC)} = \sum [Q_{(DC)} \times EF_{(DC)}] \quad (\text{summation over all months of milk production})$$

with

$$EF_{(DC)} = VS_{(DC)} \times B_{o(DC)} \times (MCF_{(DC)}/100) \times MS_{(DC)}$$

where

$M_{(DC)}$  = CH<sub>4</sub> emission from FDE ponds (kg CH<sub>4</sub> year<sup>-1</sup>)

$Q_{(DC)}$  = monthly population of lactating dairy cows (head), e.g. see Table 8 for 2009 year



$EF_{(DC)}$  = monthly CH<sub>4</sub> emission factor for dairy cows (kg CH<sub>4</sub> head<sup>-1</sup> month<sup>-1</sup>)

$VS_{(DC)}$  = monthly amount of VS generated = 0.82 × monthly amount of FDM (kg)

$B_{o(DC)}$  = maximum CH<sub>4</sub>-producing capacity for FDE in NZ = 0.147 kg CH<sub>4</sub> kg<sup>-1</sup> VS, following Pratt et al. (2012)

$MCF_{(DC)}$  = CH<sub>4</sub> conversion factor for FDE (%), in absence of NZ-specific data taken as IPCC (2006) default, 71% (at 13°C average annual temperature)

$MS_{(DC)}$  = 0.11 (fraction of total manure mass handled, based on average time fraction per day spent by cows on the milking platform)

#### 6.4.4 Revised estimates for CH<sub>4</sub> pond emissions using the improved equation

The results of applying Equation 2 for 2009 data (same inputs as in Table 8) are shown in Table 9. With this methodology, the emission factor for annual CH<sub>4</sub> emissions from FDE ponds is obtained as  $EF_{(DC)} = 7.125$  kg CH<sub>4</sub> head<sup>-1</sup> year<sup>-1</sup>. This value is higher than the EF values derived from McGrath and Mason (2004) and Craggs et al. (2008), which are 6.56 and 6.4 kg CH<sub>4</sub> head<sup>-1</sup> year<sup>-1</sup>, respectively, probably due to a higher fraction of FDE being now anaerobically treated at the farm level. It is, however, lower than the value determined by Pratt et al. (2012) from three FDE ponds representing different climatic regions of NZ: correcting their experimentally recorded EF of 13.45 kg CH<sub>4</sub> head<sup>-1</sup> year<sup>-1</sup> to account for the nationwide usage rate of 73.5% in 2010 yields 9.89 kg CH<sub>4</sub> head<sup>-1</sup> year<sup>-1</sup>. The higher value of Pratt et al. (2012) could be explained by the relatively significant contribution of waste milk, as demonstrated in Section 6.4.5, or the fact that the 15% manure recovery factor assumed for these calculations will vary greatly between farms.

If, in Table 9,  $MS_{(DC)}$  was assumed as 0.06 for each month, including April to June, then the annual total  $M_{(DC)}$  would be obtained as 20.087 Gg CH<sub>4</sub>, which is 33.8% larger than the 15.014 Gg obtained in Table 8. Hence, replacing the flawed inventory equation by Equation 2 and keeping the manure input estimates of the current inventory would yield 33.8% larger CH<sub>4</sub> emissions from FDE ponds than currently reported. A further discrepancy of 60% (32.319 Gg CH<sub>4</sub> from Table 9 compared to 20.087 Gg CH<sub>4</sub>) originates from the different assumptions for  $MS_{(DC)}$ .

Using Equation 2 and  $MS_{(DC)}$  as specified in Table 9, an estimate for the total pond emissions of  $M_{(DC)} = 32.32$  Gg CH<sub>4</sub> year<sup>-1</sup> is thus obtained for 2009, equivalent to<sup>3</sup> 679 Gg CO<sub>2</sub>-e year<sup>-1</sup>. This estimate is larger by a factor of 2.15 than the current inventory value of 15.01 Gg CH<sub>4</sub> year<sup>-1</sup> (Table 8). It falls into the range given by Chung et al. (2013); however, it should be noted that these authors used the lower value of  $MS_{(DC)} = 0.06$ . If, for consistency, one changed  $MS_{(DC)}$  in the calculations of Chung et al. (2013) from 0.06 to 0.11 × (9/12), where the factor 9/12 accounted for the winter months not contributing to the

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<sup>3</sup> This conversion uses the current global-warming-potential (GWP) factor, 21 kg CO<sub>2</sub>-e/kg CH<sub>4</sub>. With the revised GWP value for future inventories, 25 kg CO<sub>2</sub>-e/kg CH<sub>4</sub>, the emissions would be 808 Gg CO<sub>2</sub>-e year<sup>-1</sup>.

manure load of FDE ponds, then their estimated range of the CH<sub>4</sub> emissions would increase to 796 to 1262 Gg CO<sub>2</sub>-e and the result from the refined Equation 2 would be outside this range. The main reason for this is the discrepancy in  $VS_{(DC)}$  values: Chung et al. (2013) used the IPCC default, with  $\pm 20\%$  variation, while the  $VS_{(DC)}$  data in Table 9 are equivalent to 40% less than the IPCC default on average, as noted before.

Overall, there is a more than two-fold variation in the emission estimates obtained with Tier 2 approaches in this section and Section 6.4.2, from 579 to 1262 Gg CO<sub>2</sub>-e. This range reflects the cumulative uncertainty of the factors in Equation 2 ( $B_o$ ,  $VS$ ,  $MCF$  and  $MS$ ). Several measures may be considered to improve accuracy: one could implement systems recording the actual amounts of collected manure; one could develop relationships predicting the time spent by cattle herds on the milking platform, based on herd size, number of milking bays and average time required to milk a cow (or amount of milk produced); or one could estimate the time on the milking platform from surveys of dairy farm staff. Further investigation of the discrepancy between the  $VS$  load derived from the DMI data in NZ's inventory and the IPCC default values is also recommended.

**Table 9: Annual CH<sub>4</sub> emissions from FDE ponds on NZ's dairy farms, for 2009, based on modified Tier 2 approach (Equation 2). FDM = faecal dry matter, VS = volatile solids. The factors from Equation 2 not tabulated here were set constant:  $B_{o(DC)} = 0.147$  kg CH<sub>4</sub> kg<sup>-1</sup> VS and  $MCF_{(DC)} = 0.71$ .**

Month	Population (head)	FDM generated (kg DM)	VS generated (kg VS)	$VS_{(DC)}$ (kg VS/head)	$MS_{(DC)}$	$EF_{(DC)}$ (kg CH <sub>4</sub> /head)	$M_{(DC)}$ (Gg CH <sub>4</sub> )
Jan	4,347,656	370,730,240	303,998,797	69.92	0.11	0.803	3.490
Feb	4,347,656	374,445,334	307,045,174	70.62	0.11	0.811	3.525
Mar	4,347,656	368,476,176	302,150,464	69.50	0.11	0.798	3.469
Apr	4,347,656	175,385,356	143,815,992	33.08	0.00	0.000	0.000
May	4,606,970	138,923,910	113,917,606	24.73	0.00	0.000	0.000
Jun	4,606,970	164,471,531	134,866,655	29.27	0.00	0.000	0.000
Jul	4,782,412	342,827,522	281,118,568	58.78	0.11	0.675	3.227
Aug	4,606,970	404,426,750	331,629,935	71.98	0.11	0.826	3.807
Sep	4,606,970	388,018,803	318,175,419	69.06	0.11	0.793	3.653
Oct	4,606,970	342,449,158	280,808,310	60.95	0.11	0.700	3.224
Nov	4,606,970	362,353,270	297,129,681	64.50	0.11	0.740	3.411
Dec	4,606,970	479,320,297	393,042,643	85.31	0.11	0.979	4.512
<b>2009 total</b>				<b>707.72</b>	<b>0.096</b>	<b>7.125</b>	<b>32.319</b>

#### 6.4.5 Revised estimates for CH<sub>4</sub> emissions from dairy manure including other substrate sources

While the suggested new inventory equation for FDE ponds, Equation 2, certainly represents an improvement over current practice, there remain several parameters that need to be resolved before a really robust approximation of pond CH<sub>4</sub> emissions can be established. One parameter that could considerably affect CH<sub>4</sub> emissions is the *type* of effluent pond in use on farms. In the report by Pratt et al. (2012), only the presence or absence of an FDE pond on a farm was recorded. If a pond was registered, it was considered to be anaerobic and a CH<sub>4</sub> source. However, there exist a wide variety of FDE pond types on dairy farms, and it is not well-established how CH<sub>4</sub> emissions from storage ponds depend on management practices and in which circumstances they approach the full emissions potential of anaerobic ponds.

Further, the estimates of CH<sub>4</sub> emissions in Table 9 are based solely on estimates of the amount of effluent collected from milking platforms into FDE ponds. Chung et al. (2013) demonstrated that other sources of material can add considerably to these emissions: effluent collected at feed pads and stand-off pads, waste milk, and supplementary feed waste. Chung et al. (2013) outlined how a methodology could be structured to collect better data for these sources, and showed that including these emission sources leads to total dairy manure emissions roughly twice as large as those from FDE collected at the milking platform alone.

In the following, the contribution of these additional substrate sources to manure emissions is investigated by computing CH<sub>4</sub> emissions from FDE and other dairy-related substrate sources across NZ for two scenarios. As detailed in Table 10, Scenario A is considered as realistic for 2012 based on the present study and Luo et al. (2013). Scenario B is intended to include future trends, thus highlighting the importance of rectifying the accounting of FDE and other dairy-waste emissions in future GHG inventories. It should be noted that the following calculations are merely intended to highlight the potential significance of currently ignored organic streams. There are currently not enough relevant data available for a more accurate assessment. The following methodology was used:

- Equation 2 was used together with data shown in Table 9 to compute the CH<sub>4</sub> emissions from FDE collected at milking platforms into effluent ponds. These data are for the year 2009. This causes some underestimation of present emissions, due to increases in dairy-cow population, cow weights and milk yields that have occurred since 2009.
- The contribution of milk waste was computed as described by Chung et al. (2013). In brief, the total amount of waste milk treated in effluent ponds was estimated as 67 (average of 49 to 85) L head<sup>-1</sup> year<sup>-1</sup> and the milk's ultimate BMP was estimated as 48.8 g CH<sub>4</sub> L<sup>-1</sup> milk anaerobically treated. The IPCC default *MCF* value of 71% was used as in the case of FDE. A more accurate assessment should account for monthly variations of waste milk generation.
- The contribution of feed was computed as described by Chung et al. (2013) with modifications according to Luo et al. (2013). In brief, feed waste generation was estimated as 0.22 (average of 0.134 to 0.305) kg VS head<sup>-1</sup> d<sup>-1</sup> and the ultimate BMP of the feed was estimated as 0.208 kg CH<sub>4</sub> kg<sup>-1</sup> VS anaerobically treated. When the feed waste was diverted to storage or separation (not stored or treated in an effluent pond), the IPCC default *MCF* value of 19% was used as done by Luo et al. (2013).
- The contribution of the manure recovered from feed pads, stand-off pads, animal shelters and wintering barns was computed using Equation 2, and again assuming an *MCF* value of 19%. Note that this is adequate only for farms where this manure is collected and

treated in solid form. There is a wide variety of manure treatment systems in present and future use in NZ, and on some farms, part or all of the manure is added to the FDE pond as a liquid fraction (John Scandrett, Dairy Green, pers. comm., 2014). For those farms, the correct *MCF* value would be 71%. The assumption made here that all feed-pad, stand-off-pad and housing manure is treated in solid form will therefore lead to an underestimate of the CH<sub>4</sub> emissions. In this regard the two scenarios must be considered as conservative.

**Table 10: Assumptions for two scenarios for CH<sub>4</sub> emissions from dairy manure treatment systems, and comments on their likelihood**

Assumptions on:	Scenario A (present)	Scenario B (future)	Comments
Farms without FDE ponds	26.5% of farms, no CH <sub>4</sub> emissions assumed to occur from these	No farms without FDE ponds	Even short retention times (one-day storage in a sump) may cause significant CH <sub>4</sub> emissions
Farms with FDE ponds	73.5% of farms, <i>MCF</i> = 71%	100% of farms, <i>MCF</i> = 71%	It is likely that most regional councils will make FDE ponds compulsory
Manure fraction treated in FDE ponds	15% of manure from farms with FDE ponds, resulting in $MS = 0.15 \times 0.735 = 0.11$ during the milking season ( $MS = 0$ during 3 winter months)	15% of manure from farms with FDE ponds, resulting in $MS = 0.15$ during the milking season ( $MS = 0$ during 3 winter months)	Manure volumes are calculated for 2009 animal numbers, have increased since
Feed-pad use	27% of farms (Luo et al. 2013), manure treated as solids with <i>MCF</i> = 19%	100% of farms, manure treated as solids with <i>MCF</i> = 19%	Assumption on <i>MCF</i> is a lower bound because it is ignored that some farms treat feed-pad manure as liquid in FDE ponds
Manure fraction from feed pads	Cows spend 2 hours/day on feed pads (Chung et al. 2013), resulting in $MS = 0.083 \times 0.27 = 0.023$ during the milking season ( $MS = 0$ during three winter months)	Cows spend 2 hours/day on feed pads, resulting in $MS = 0.083$ during the milking season ( $MS = 0$ during three winter months)	Growth is assumed only for usage of manure treatment technologies, but not for animal numbers (manure volume)
Use of stand-off pads, animal shelters and cow housing	24% of farms (Luo et al. 2013), manure treated as solids with <i>MCF</i> = 19%	40% of farms, manure treated as solids with <i>MCF</i> = 19%	Assumption on <i>MCF</i> is a lower bound because it is ignored that some farms treat stand-off-pad manure as liquid in FDE ponds
Manure fraction from stand-off pads, animal shelters and cow housing	Cows spend 16 hours/day in these (Luo et al. 2013), resulting in $MS = 0.667 \times 0.24 = 0.16$ during three winter months ( $MS = 0$ during milking season)	Cows spend 16 hours/day in these, resulting in $MS = 0.667 \times 0.40 = 0.267$ during three winter months ( $MS = 0$ during milking season)	Growth is assumed only for usage of manure treatment technologies, but not for animal numbers (manure volume)

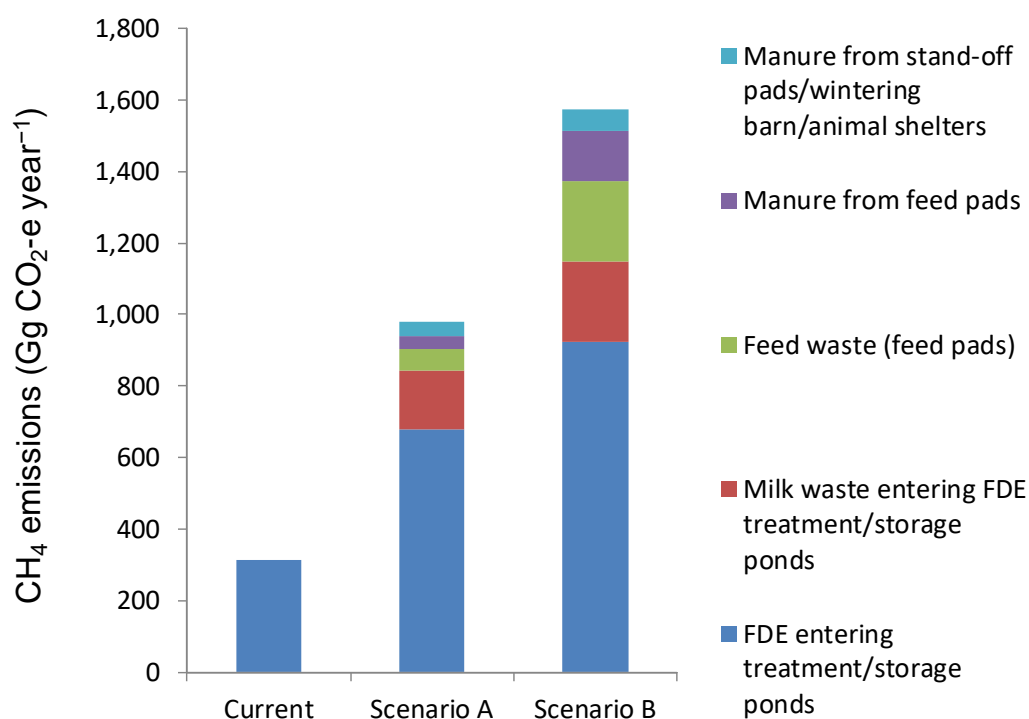


Figure 9: Methane emissions from NZ's dairy manure treatment systems for Scenarios A and B, with substrate sources indicated by different colours. Scenario A estimates 2012 conditions, Scenario B illustrates the effect of future growth trends in the usage of FDE ponds, feed pads and stand-off pads. See Table 10 and text above the table for details. The CH<sub>4</sub> emissions estimate in NZ's GHG Inventory for 2009 (MfE 2012) is given for comparison ("Current").

Figure 9 shows the emission estimates resulting for each scenario in comparison to NZ's GHG inventory for 2009 (MfE 2012). The emissions from FDE collected at milking only (blue column bases in Figure 9) are 679 Gg CO<sub>2</sub>-e for Scenario A (as obtained from Table 9) and 926 Gg CO<sub>2</sub>-e for Scenario B, exceeding the inventory estimate of 315 Gg CO<sub>2</sub>-e by factors of 2.2 and 2.9, respectively. The total emissions (whole columns in Figure 9) are 978 and 1573 Gg CO<sub>2</sub>-e for Scenarios A and B, respectively, exceeding the current inventory estimate by factors of 3.1 and 5.0, respectively. Despite considerable uncertainty associated with some scenario assumptions, the results indicate that for an accurate estimation of CH<sub>4</sub> emissions from dairy manure treatment in NZ, the relative importance of emissions caused by additional input streams needs to be adequately reflected, as well as their future growth. Crucial parameters are, firstly, how much time cattle spend on feed pads and stand-off pads and what fraction of the manure deposited on these pads is collected into management systems, in order to specify the *MS* values. Secondly, the results will strongly depend on the *MCF* values specified for each treatment and storage system.

Still not included in the discussed studies and scenarios are the effects of pre-treatment systems on CH<sub>4</sub> emissions, including the emissions from these systems themselves and the potential reduction of further emissions later in the treatment chain (i.e. from ponds). With regards to solids-separation systems, some studies report that emissions from mechanical

types of these are negligible (Hansen et al. 2006; Fanguero et al. 2008, see Table 2); whereas the IPCC (IPCC 2006) and preliminary results from Massey University Farm 4 (Guieysse et al., unpublished data) suggest significant CH<sub>4</sub> emissions. For weeping-wall systems, no data exist. It is therefore recommended that further research be undertaken into emissions from pond systems with weeping walls. Potentially, such systems could be investigated with micrometeorological methods, surrounding the emitting system with wind and concentration sensors (rather than having to deal with floating covers on two different surfaces, up- and downstream of the weeping wall). Available methods are the mass-budget method, as successfully applied to CH<sub>4</sub> tanks in Canada (Park et al. 2010; VanderZaag et al. 2011), as well as line concentration sensors in conjunction with a backward-Lagrangian flow model, as evaluated by Laubach et al. (2013a). Should CH<sub>4</sub> emissions from solid storage/separation systems indeed prove to be significant, then they should be included in future inventories. That would also require the specification of activity data on their use, which would probably have to be obtained by farm surveys.

## 6.5 Alternative inventory approach for N<sub>2</sub>O

The IPCC Tier 2 methodology recommends a default factor of 0 (zero) for direct N<sub>2</sub>O emissions from anaerobic ponds and lagoons. By contrast, indirect N<sub>2</sub>O emissions, via NH<sub>3</sub> volatilisation, can occur during any stage of the manure management. The default factor for indirect N<sub>2</sub>O emissions is 1% of volatilised nitrogen, regardless of where the volatilisation process is located. It would therefore make sense to develop a sub-inventory for NH<sub>3</sub> emissions, including all possible pathways (from collection, pre-treatment, storage and land application), compute NZ's total annual emissions of NH<sub>3</sub>, and then convert this number into the total indirect N<sub>2</sub>O emissions by applying the emission factor of 1%. The benefit of such a change to the inventory would be two-fold. Firstly, the national NH<sub>3</sub> emissions would be of interest outside the GHG context (e.g. for studies on aerosol formation affecting human health, or on the potential for eutrophication of waterways). Secondly, the conversion step from NH<sub>3</sub> to N<sub>2</sub>O would need to be implemented only once in the inventory worksheets, rather than in every affected emissions pathway.

Alternatively, the mass-balance approach of Dämmgen and Hutchings (2008), which includes the very small NO losses from liquid manure storage as well as losses of N<sub>2</sub> gas, should be considered as a potential method to improve the N<sub>2</sub>O emission accounting from dairy-farm manure management in NZ. Use of this methodology would address the identified omissions in the current methodology (e.g. milking parlour and holding yard NH<sub>3</sub> emissions and resulting indirect N<sub>2</sub>O emissions). It is also flexible enough to deal with new, or currently omitted, technology and emission pathways (such as emissions from pre-treatments like solids separators and weeping walls).

Such an approach would require the specification of more emission factors than in the present inventory. For some of these, in particular the N<sub>2</sub> emissions, data would be very sparse. To overcome this, Dämmgen and Hutchings (2008) suggest a wide range of useful and applicable default emission factors. These can be complemented with NZ-specific factors as more information becomes available. The approach would also require relevant activity data for the additional pathways, such as statistical information on the types of solids-separator systems used on NZ farms, which could be gathered by farm surveys. Since the combined direct and indirect N<sub>2</sub>O emissions are about one magnitude smaller (expressed as CO<sub>2</sub>-equivalents) than the CH<sub>4</sub> emissions, the relative uncertainties of emission factors and activity data for N<sub>2</sub>O are less critical than for CH<sub>4</sub>.

## 7 Mitigation options

### 7.1 Mitigation of methane emissions during FDE treatment and storage

Methane emissions during storage, in particular those from FDE ponds, are by far the largest contributions to the total GHG emissions from managed manure. Therefore, mitigation approaches that target CH<sub>4</sub> emissions from ponds would have the largest impact towards reducing emissions from manure. There has been comparatively little research conducted into such approaches, compared with enteric CH<sub>4</sub> abatement research. On the basis of emissions currently reported in NZ's GHG inventory this seemed justifiable, given the relative contributions of these sources to the country's agricultural GHG profile. However, now that it is evident that pond emissions are considerably higher than reported, it would appear worthwhile to re-evaluate the balance of investment into CH<sub>4</sub> mitigation research.

Some options for mitigating CH<sub>4</sub> emissions from ponds were outlined in a conference paper by Shilton et al. (2009). These are summarised in Table 11, modified from Pratt et al. (2014b). The options listed in Table 11 show that there are several alternatives by which potentially large reductions of CH<sub>4</sub> emissions from ponds could be achieved effectively. However, research is required to confirm many of the underlying mechanistic assumptions and refine the feasibility analysis before these options can be rolled-out at full-scale. For some options, it should also be investigated whether the CH<sub>4</sub> emissions reductions might be associated with increases in N<sub>2</sub>O, NH<sub>3</sub> or CO<sub>2</sub> emissions, i.e. the net GHG reduction efficiency would need to be determined.

Not included in Table 11 is one rather simple option to reduce the time that FDE is stored anaerobically. This is to avoid deferred irrigation as much as possible and irrigate FDE daily; however, such an approach to reduce CH<sub>4</sub> emissions would potentially conflict with the goal to minimise nitrate leaching, at least in the wetter parts of the country. It would also remove an opportunity to test and, where necessary, treat the FDE for presence of pathogens. Recommending daily irrigation would run contrary to present best-practice recommendations of the industry (Dairy Insight 2007a, b) and make it hard to justify investments into new FDE storage ponds.

Of the options listed in Table 11, the covering of FDE ponds and treating of the recovered CH<sub>4</sub> in bio-filters, combusting it in a flare or using it for on-site energy provision appear the most promising. For large dairy farms, or for farms that value energy-supply security highly, CH<sub>4</sub> capture with energy recovery can already be cost-effective based on the energy value alone (EECA dairy-farm biogas feasibility studies, unpublished). Biogas recovery systems will occasionally also be implemented where dairy farms have to deal with odour problems. While such systems are technically mature, mainstream uptake in NZ may be hindered because there is little awareness within the dairy industry of the CH<sub>4</sub> emissions problem and the already available solutions, and because of up-front investment costs. Future policies designed to overcome these obstacles could go a long way to improve the uptake of technologies that actively reduce GHG emissions from manure management on dairy farms.

Interestingly, compared with the targeted 30% reduction in enteric emissions over the next decade (Barber 2013), almost the same amount of CH<sub>4</sub> could be offset from ponds on dairy farms, with a more balanced research portfolio. Yet, currently there is no funding set aside by the NZ GHG Centre for researching the mitigation options listed in Table 11. We recommend that this imbalance be addressed in future allocations of research funding.

**Table 11: Mitigation options for pond CH<sub>4</sub> emissions, adapted from Pratt et al. (2014b)**

Mitigation approach	Achievable CH <sub>4</sub> emissions reduction (%)	Comments on confidence in effectiveness of technology
Covering of pond and flaring of CH <sub>4</sub> , with or without energy recovery	97 *)	Approach has been implemented successfully on other livestock farms (e.g., pig farms)
Bio-filter cover, comprising CH <sub>4</sub> -consuming bacteria, on pond surface	98 *)	Bio-filter cover design has not been field-tested. CH <sub>4</sub> reduction shown is based on preliminary study (Pratt et al. 2013)
Effluent acidification	Unknown	Potential to reduce NH <sub>3</sub> as well as CH <sub>4</sub> emissions. Chemical requirements (availability/cost) and potential for odour unclear.
Conversion of anaerobic to aerobic or facultative ponds	73 *)	CH <sub>4</sub> reduction based on theoretical calculations, field measurements required. In theory, easy to realise on farms with two-pond treatment systems, but may often not be practical because required area is too large (since depth for aerobic ponds should not exceed about 0.5 m).
Solids separation: physically preventing solids from entering anaerobic ponds and depositing them in aerobic conditions on pastures instead	40 for solids separation alone, 80 in combination with facultative pond *)	CH <sub>4</sub> reduction based on theoretical calculations, field measurements required. There is a risk of increased N <sub>2</sub> O with greater aeration, so it needs to be carefully considered whether there is a net reduction in combined GHG emissions.

\*) According to Shilton et al. (2009), Pratt et al. (2013)

## 7.2 Mitigation of emissions from land application

Mitigation of emissions from the application of FDE to pasture is more difficult to achieve than from pond systems (where the emission source is contained in a rather small, already controlled area). It is also perhaps of a lower priority because the emissions from land application, expressed in CO<sub>2</sub>-equivalent emissions, are significantly smaller than those from ponds. Methane emissions from land application are probably insignificant, which should be confirmed in the NZ context.

Nitrous oxide emissions tend to be smallest when soils are dry. Application of FDE during periods of saturated soils should thus be avoided. This is recommended best practice anyway in order to keep N leaching to a minimum. However, where the regular application of FDE is combined with regular irrigation of pasture, which, under good management, aims to apply sufficient water to avoid pasture stress while also avoiding soil saturation, N<sub>2</sub>O emissions may not be minimised. Whether timing and application rate of FDE on irrigated dairy farms can be optimised for low N<sub>2</sub>O emissions is an unstudied area.

An indirect way to reduce N<sub>2</sub>O emissions would be to remove as much as possible of the carbon that is easily available for N<sub>2</sub>O generation via denitrification. This would be achieved by near-complete anaerobic digestion of FDE prior to application, i.e. sufficiently long storage in ponds, where the ponds should be covered to avoid CH<sub>4</sub> emissions. Another possible approach to reduce N<sub>2</sub>O emissions might be to add nitrification inhibitors to FDE just before land application. The efficacy of this is so far unknown but is the subject of current trial work.



Ammonia emissions tend to be largest at high temperatures and at high pH. In principle they could be suppressed if the FDE could be kept acidic during and after application (that would also help to suppress the release of CH<sub>4</sub> trapped in the FDE that had previously been anaerobically formed). However, this is probably impractical. Avoiding hot dry conditions for application would be possible, but may conflict with the goal to reduce direct N<sub>2</sub>O emissions. The best approach to minimising NH<sub>3</sub> emissions is probably to keep the effluent as diluted as practical (which is desirable for smooth irrigator operation anyway), although this may increase N<sub>2</sub>O emissions depending on the application rate.

## 8 Conclusions and recommendations

### 8.1 Summary of knowledge about GHG emissions from FDE in NZ

The primary source of FDE is the milking parlour, where the cows' excreta are washed off with water and collected, either for direct land application, or into a storage pond for deferred FDE irrigation. The FDE is more dilute than manure stored and spread in many other countries, and therefore the number of studies relevant to NZ is limited. The number of farms with ponds has been rapidly increasing in recent years, surpassing 9,700 in 2011. According to data collated for this report (from regional councils, Dairy NZ, the Fertiliser Association and Fonterra), the average pond volume is about 1700 m<sup>3</sup>, representing about 3 months' storage capacity. In addition to the FDE from milking platforms, manure from other sources is increasingly being collected. These sources are feed pads (present on 27% of farms in 2010/11), stand-off pads (on 22% of farms), and winter shelters or housing (on 2% of farms), according to Luo et al. (2013). On some farms, this manure is added to FDE ponds, on others, it is anaerobically digested in solid form. Also increasing are pre-treatment practices, in particular solids separation and weeping walls.

For a complete accounting of GHG emissions, these would have to be quantified for the whole treatment chain, consisting of: emissions at source (milking sheds, yards, pads, winter housing), emissions during pre-treatment, emissions from ponds, and emissions from land application. Hence, emission factors and activity data would be required for each of these. Some of the available activity data are given in the previous paragraph and Tables 1, 8 and 9. Our knowledge about emission factors is summarised in the following.

There are at present no data for emissions at source. Based on the very short periods that the excreta stay at the sources, it can be speculated that emissions of CH<sub>4</sub> are N<sub>2</sub>O during these periods are very minor; however, NH<sub>3</sub> emissions are potentially considerable, of the order of 20% of TAN (Dämmgen and Hutchings 2008) because the co-deposition of urine and dung provides the substrate for rapid urea hydrolysis followed by NH<sub>3</sub> volatilisation.

For GHG emissions from FDE pre-treatment system in use in NZ, there are no published data to date with respect to CH<sub>4</sub>. Emissions of N<sub>2</sub>O from solids were found to range from 0.1 to 4.8% of total N internationally (studies with relevance indicated as "high" or "medium" in the first section of Table 3). Emissions of NH<sub>3</sub> from solids were found as 12 to 21% of TAN (Dinuccio et al. 2012; Moral et al. 2012, van der Weerden et al. 2014b).

From FDE ponds, three NZ studies have consistently found CH<sub>4</sub> emissions of about 0.20 m<sup>3</sup> CH<sub>4</sub> kg<sup>-1</sup> VS (Craggs et al. 2008; Pratt et al. 2012; Heubeck 2013<sup>4</sup>). Our review of the international literature suggests that N<sub>2</sub>O emissions from liquids and slurries with low DM contents are 0 to 0.1% of total N (eight relevant studies, second section of Table 3), even less than currently assumed for pond emissions in NZ's GHG inventory. By contrast, the emissions of NH<sub>3</sub> from liquids and slurries reported in the literature are substantial and highly variable, at 1 to 65% of TAN (six relevant studies, third section of Table 4), with pH differences probably being the most important contributor to the variability. The NH<sub>3</sub>

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<sup>4</sup> The reference is to a presentation of preliminary results; report to MPI in preparation.

emissions from FDE ponds in NZ are likely to represent *indirect* N<sub>2</sub>O emissions of a similar magnitude as the presently-reported *direct* N<sub>2</sub>O emissions.

No direct measurements of CH<sub>4</sub> emissions exist from the land application of FDE on NZ farms. Overseas studies on the effects of slurry spreading indicate that, apart from a brief spike on application (releasing already-formed CH<sub>4</sub> gas previously trapped within the slurry), such emissions are negligible. This would be expected because on application the FDE or slurry becomes aerobic, while methanogenic microbial activity is usually confined to anaerobic conditions. For N<sub>2</sub>O emissions from land application of FDE, the emission factors found in four NZ studies span from 0.01 to 4.9% of N (Table 5). This range includes that reported for N<sub>2</sub>O emissions from land-applied slurries, of < 0.1 to 3% (Chadwick et al. 2011). Land application of FDE caused NH<sub>3</sub> emissions of 0.05 to 3.1% of total N in three NZ studies (Di et al. 1998; Li et al. 2014a;b), which thus would give rise to only minor indirect N<sub>2</sub>O emissions.

Nitrate leaching occurs only from the last step of the FDE treatment chain, the land application. In three NZ studies, this was quantified as 1.6 to 4.7% of N (Di et al. 1998; Silva et al 1999; Di and Cameron 2002a). This is less than the factor of 7% currently used in the GHG inventory.

There is no doubt that the CH<sub>4</sub> emissions from ponds are the largest contributor to the total GHG emissions from FDE. N<sub>2</sub>O emissions from ponds are at least 20-fold smaller than these, when using internationally-agreed GWP factors for comparing the different gas emissions. The second-largest contribution to GHG emissions from FDE are the N<sub>2</sub>O emissions from land application, but these occur from all animal manure, regardless of whether the manure is collected and treated as FDE or deposited directly by excreting animals. Emissions from the other steps of the FDE treatment chain are poorly characterised, but will probably account only for a few percent of the total GHG emissions from FDE.

Crucially, the current inventory equation for CH<sub>4</sub> emissions from FDE ponds is flawed (Chung et al. 2013, Pratt et al. 2014a), and it is likely that as a consequence these emissions are underestimated. In addition, the amount of manure collected at milking sheds and anaerobically digested is probably underestimated. The combined effect of correcting the inventory equation and assuming an increased proportion for collected dairy manure is a doubling of the CH<sub>4</sub> emissions from FDE ponds compared to those reported in NZ's GHG inventory. Further adding to these emissions are waste streams from feed pads, stand-off pads and winter housing. Our scenario results (Figure 9) suggest that using a correctly-adapted Tier 2 equation and including all substrate sources will lead to 3 to 5 times larger CH<sub>4</sub> emissions from dairy manure than currently reported. Hence, instead of representing about 4% of the total CH<sub>4</sub> emissions from dairy farming, they are more likely to account for between 11 and 17% of these CH<sub>4</sub> emissions. Consequently, total emissions from the manure management category are more likely to represent about 4 to 7% of the total emissions from the agricultural sector, rather than 2% as presently reported (MfE 2013). Since emissions from ponds are far more amenable to capture and mitigation (Table 11) than enteric emissions, the potential for actual reduction of agricultural emissions in NZ by applying suitable measures at FDE ponds is larger than previously assumed.

## 8.2 Knowledge gaps

This work has identified several questions that need to be addressed in order to improve our understanding of emissions from FDE ponds and develop appropriate mitigation pathways:

- How do CH<sub>4</sub> emissions from storage ponds compare with those from treatment ponds? (NIWA is currently undertaking research to address this, see Heubeck 2013<sup>5</sup>).
- What are the *MCF* values associated with each treatment system and waste stream under relevant climatic conditions?
- How widely adopted is the practice of solids separation and how efficient is it at keeping solids aerobic upon removal? What amounts of digestible solids are reintroduced into the treatment pond?
- What are NH<sub>3</sub> emissions from FDE pond systems?
- What are CH<sub>4</sub>, NH<sub>3</sub> and N<sub>2</sub>O emissions from solids-separation systems (in particular from weeping walls where few published data exist)?
- What amounts of manure from feed pads and stand-off pads are handled as manure solids or as additional inputs to FDE ponds, respectively, and how can data on these amounts be kept up-to-date as management practices in the dairy industry evolve?
- In what respects do the soil and environmental factors that influence N<sub>2</sub>O and NH<sub>3</sub> emissions following land application of FDE differ from the factors that determine the N<sub>2</sub>O and NH<sub>3</sub> emissions following excreta deposition?

## 8.3 Recommendations

Based on the findings of this report, we recommend, with high priority, to:

- a) Improve NZ's GHG inventory methodology for estimating CH<sub>4</sub> emissions from dairy manure, by:
  - replacing the current, mathematically flawed, equation for emissions from FDE ponds with a correct equation, adapted from the IPCC guidelines (IPCC 2006), i.e. Equation 2 with numerical factors as specified there,
  - collecting data to improve the accuracy of the estimated fraction of manure entering effluent management systems.

These steps will provide a far more robust methodology than the approach currently used in the inventory.

Further, we recommend to:

- b) Fund research to quantify GHG emissions from pre-treatment technologies for FDE, in particular the increasingly common weeping walls. This could be approached using chamber-based studies, or by measuring emissions from whole FDE management systems using micrometeorological approaches that surround ponds and treatment facilities.

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<sup>5</sup> The reference is to a presentation of preliminary results; report to MPI in preparation.

- c) By adopting a Tier 2 approach, remove FDE ponds as a source of *direct* N<sub>2</sub>O emissions from the inventory calculations, but include them as a source of *indirect* N<sub>2</sub>O emissions (NH<sub>3</sub> volatilisation). Fund some targeted experiments to quantify NH<sub>3</sub> emissions from FDE ponds.
- d) At the least, include emission pathways for N<sub>2</sub>O and NH<sub>3</sub> in the inventory that have so far been neglected, such as for emissions from milking parlour and holding yards and from pre-treatment activities. The nitrogen lost in pre-treatment steps must then be subtracted from the substrate basis of the subsequent processing steps (storage and land application). Or, more comprehensively, consider moving to a full mass-balance accounting system for nitrogen, as developed by Dämmgen and Hutchings (2008).
- e) Include a question in the Agricultural Production Survey that addresses FDE pond design and operation. If this is not possible, then design a research plan on how to collect quantitative information on pond design and operation, rather than relying solely on dairy organisations that do not have the resources to keep detailed records of this information.
- f) Allocate more funding to research into mitigation options for emissions during storage, and develop policy on how the most economic options can be put into widespread practice.
- g) Support field demonstrations of systems that reduce GHG emissions, such as covered anaerobic ponds, and make results from these available to end-users via industry organisations.
- h) Continue to fund research into the relationships between soil moisture, nitrogen and carbon to better understand the processes controlling N<sub>2</sub>O emissions. This will allow the design of optimised schedules for application of FDE on irrigated dairy farms.

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### Title and Authors:

Review of gaseous emissions of methane, nitrous oxide and ammonia, and nitrate leaching to water, from farm dairy effluent storage and application to land

Final Report

Prepared for Gerald Rys, Principal Science Advisor, Science Policy, Strategy Systems and Science, MPI

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